

TNO report

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**Risks to Health and the Environment Related to the
Use of Lead in Products**

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1 Introduction

Lead is a widely used material in our society. Applications include car batteries, sealant in the building sector (e.g. chimney tops), various elements of electrotechnology and electronic goods (e.g. solder, glass for TV tubes), weights, lead shot, (professional) fishing and other applications, among others the use in lead stabilisers in PVC. Lead used to be applied in significant quantities as a scavenger in petrol, but with the introduction of lead-free petrol all over Europe this application has dropped dramatically. The same applies for the use of lead in applications such as water pipes.

The latter reductions of lead use caused a significant reduction of the direct emissions and exposure of humans to lead – unlike most other lead applications, the use in petrol in particular lead to emissions in a highly mobile form. However, there is still discussion about whether to strive for a further reduction of exposure to lead. There is uncertainty about the long-term risks of accumulation of lead in society and the potential for diffuse releases from various applications. In various EU member states there are calls to embark on a source-oriented policy, and for a further reduction of the uses of lead in products. Adherents of such a policy fear that, particularly during waste management, the lead may come available in a more mobile form, causing higher emissions in the long run¹.

Given this context, the Commission's Services have decided to launch a study into the risks of the present remaining uses of lead over time as an input to a discussion on the need for a further reduction of the use of this material. TNO was asked to perform this study. The following elements have been addressed:

- Effect assessment: derivation of NOAELs (human) and NOAECs (environment); see Chapter 2;
- An inventory of lead uses, trends and emissions in the EU, divided by product group; see Chapter 3;
- Preliminary risk assessment (see Chapter 4);
- Overall summary and conclusions (see Chapter 5).

It should be noted that the study is not meant as a comprehensive risk assessment. Its main goal is to map the current lead flows and emissions in the EU, and to give a preliminary insight into current and (due to time lags in dispersion) future potential risks. The main purpose of this study is therefore to indicate where questions with regard to risks are still outstanding rather than to unambiguously identify situations that could pose a risk. The main aim is to help the Commission's Services to decide if a more comprehensive, formal risk assessment on lead is necessary.

¹ An example is energy recovery processes for certain lead-containing waste products. Lead from for example solder and plastics most probably ends up in fly ashes and incineration slag, and might leach out on the long run.

2 Dose-Response Assessment

2.1 Introduction

This chapter provides the first building block in the preliminary risk assessment, i.e. the effect assessment of lead. In this chapter we will review the toxicological data known on lead, and indicate, in principle, which concentration limit values can be applied to assess human and ecotoxicological exposure. Given the confines of the study approach, the tender document indicated that this work could be based on an analysis of a number of key review studies on (eco)toxicological effects.

Sections 2.2 to 2.7 discuss human toxic effects. The information on human toxicity presented here is mainly based on the most recent ATSDR toxicological profile for lead (ATSDR, 1999). Two other reviews (WHO, 1995 and Davidson, 1994), and the recent recommendation for lead from the EU's Scientific Committee on Occupational Exposure Limits (SCOEL, 2000) have been used whenever they provided additional or conflicting information. Information contained in paragraphs without a literature reference originated from the ATSDR (1999) profile. For the ecotoxic effects, the review given in section 2.8 was mainly based on work by Janus et al (1999) and WHO (1989). The review concentrates on inorganic lead, as this is the predominant form of lead in the environment. Inorganic lead appears mainly in the environment as metal or as its salts, oxides or sulphides.

As for the human toxicity part, a few remarks have to be made. There is a very substantial database on human health effects of lead. These health effects are, in general, related to the absorbed dose of lead, as expressed in the Pb blood level, and are mostly a consequence of multi-route exposure. The toxic effects of lead are independent of the route of exposure, and are correlated with the internal exposure expressed in the Pb blood level. Human case studies are therefore not normally separated into different routes of exposure.

There is no absolutely equivalent animal model for the effects of lead on humans. This will be true for most substances. Animal studies are still used, as most of the time no human data will be available and because, when available, human data have some disadvantages over animal data: e.g. the nature, duration and level of exposure are usually not well known. In the case of lead, some of these disadvantages are reduced because the Pb blood level is a fair measure of the internal dose, and its toxic effects are independent of the route of exposure. Furthermore, to whichever species of inorganic lead a person is externally exposed, once absorbed and bio-available, there is no difference between these various forms of inorganic lead. In addition, human data have the advantage that the uncertainties inherent when extrapolating adverse effects found in animals to human beings are absent. Lastly, there is an ample database on human health effects of lead. Therefore, animal data are only mentioned in the summaries below when they represent added value in relation to the available human data.

Pb blood levels reflect relatively recent exposure, as the half-life of lead in human blood is 28-36 days. However, a single blood lead determination cannot distinguish between low-level intermediate or chronic exposure and high-level acute exposure². Both types of exposure could result in the same blood level because of recycling from bone. Therefore, it is not always possible to ascertain whether human health effects are the consequence of acute or repeated/chronic exposure. Observations from human data, which cannot be ascribed to one or the other, are not included in the summaries below.

2.2 Toxicokinetics of lead in humans

2.2.1 Introduction

The term 'toxicokinetics' is broadly used to describe the absorption, distribution, metabolism and elimination of a substance following exposure, and the kinetics of these processes, where appropriate.

ATSDR (1999) extensively discusses three pharmacokinetic models which may be used for risk assessments at hazardous waste sites: one physiology-based model (developed by O'Flaherty), and two classical multi-compartmental models: the IEUBK model (from the EPA) and the Leggett model. The IEUBK model only addresses children, since it does not adequately model skeletal growth past the age of six, while both other models can be used for children and adults. The O'Flaherty and IEUBK models are well validated, but validation of the Leggett model seems to be less elaborate. ATSDR itself has developed a simple framework to predict blood levels from data on lead concentrations at hazardous waste sites. According to ATSDR, preliminary tests of its predictive power have shown promise (data not published).

2.2.2 Kinetics and metabolism

The rate of disposition of particulate airborne lead in adult humans is approximately 30-50%, and is modified by factors such as particle size and ventilation rate. Once deposited in the lower respiratory tract, particulate, elemental lead is almost completely absorbed, and all chemical forms of lead also seem to be absorbed.

In humans, gastrointestinal absorption after ingestion is influenced by the physiological state of the exposed person (e.g. age, fasting, nutritional calcium and iron status) and the physical-chemical properties of the lead ingested (e.g. particle size, mineralogy, solubility, lead species) (ATSDR, 1999). A study of rats showed that relative to lead acetate (100%), lead carbonate was absorbed 164%; lead thallate 121%; lead sulphide, lead naphthenate, and lead octoate 62-67%; lead chromate 44%; and metallic lead 14% (Davidson, 1994). Oral absorption in children seems to be higher than in adults. Children up to eight years of age absorbed 40-50% of the ingested lead, according to estimates derived from a dietary balance study. In adults, oral absorption of ingested water-soluble lead compounds ranged from 20 to 70% in subjects that had fasted and from 3 to 15% in subjects that had been fed. Animal studies (rats, monkeys) have also shown that adult animals absorb less than young animals. There are indications that children aged six to eleven and their mothers absorb a similar percentage of ingested lead. Absorption of lead may increase during pregnancy, which may contribute to the increase in Pb blood levels observed during the latter half of pregnancy.

² More detailed tests can make this distinction, though.

The absorption of lead from ingested soil is less than that from dissolved lead. Individuals who have fasted absorbed 26% of the lead present in ingested soil, while only 2.5% of the same dose of soil lead was absorbed with a meal. There are no reported measurements of the absorption of soil-borne lead in infants and children. The reduced bioavailability of soil-borne lead has been confirmed by various animal studies (e.g. with immature swine, adult rats and mice). (ATSDR, 1999)

Very little information is available on absorption of inorganic lead after dermal exposure, be it in humans or in animals. Dermal absorption is reported to be much less significant than absorption by inhalation or ingestion. However, no meaningful figures can be given for the percentage of dermal absorption of inorganic lead (ATSDR, 1999). WHO (1995) concluded that dermal absorption of inorganic lead is considered minimal. In one human volunteer study cited by WHO (1995), only very small quantities of lead acetate (0-0.3% of the applied dose) were absorbed from two hair-darkening cosmetics applied to skin.

Transplacental transfer of lead has been demonstrated in humans. The foetal/maternal Pb blood concentration ratio was approximately 0.9. One of the sources of maternal lead is the bones, which are catabolised for the production of the foetal skeleton; another may be the increased absorption during pregnancy (see above). Transfer of maternal lead to offspring can also occur during breast-feeding, as indicated by studies with mice and rats. The mother animals were exposed prior to or during lactation, and around 25-33% of the maternal dose was transferred to the sucklings (ATSDR, 1999). In humans, the concentration of lead in breast milk was 10-30% of the maternal Pb blood level (WHO, 1995).

Inorganic lead is not known to be metabolised, but it does form complexes with a variety of protein and non-protein ligands. Once absorbed, inorganic lead appears to be distributed in the same way, irrespective of the port of entrance. In humans, lead present in blood is almost exclusively found in the erythrocytes, half of it bound to haemoglobin. Most of the total body burden of lead is found in the bones. The distribution of lead in adults and children appears to be similar, although in adults 94% of the body burden is found in bones, while in children this is 73%. Based on a study of 32 deceased smelter workers, the major soft-tissue organs with lead accumulation were, in descending order, liver, kidney, lungs and brain. A similar pattern of distribution was found in animal studies with rats and mice. After cessation of exposure, tissue levels declined, except in the bones (ATSDR, 1999). The half-life of lead in human blood and soft tissue is somewhat more than one month; in bone it is more than 25 years, depending on the bone type. Lead can be mobilised from bone to blood, and thus be once again bioavailable. This seems to be the case especially with pregnant women and the elderly (WHO, 1995). It may form a major source of blood lead in persons with previous exposures (SCOEL, 2000).

Not many recent data on inorganic lead excretion were evaluated by ATSDR (1999) and WHO (1995); most studies reviewed were from the 1970s. Human data suggest that 50-60% of absorbed lead was excreted on a short-term basis when in a steady-state condition with respect to intake and output (ATSDR, 1999). One study reported that 60% of absorbed lead is retained in the body and 40% is excreted (WHO, 1995). It seems that children retain more absorbed lead than adults: infants from birth to two years of age retained 32% of the absorbed lead, while adults only retained 1%. The lead that is not retained is either excreted by the kidneys or through biliary clearance, some

in the form of glutathione conjugates. In humans as well as in animals, more lead is excreted in faeces than in urine (ATSDR, 1999). However, Davidson (1994) states that approximately 75% of inorganic lead absorbed into the body is excreted in urine and less than 25% is excreted in faeces. This discrepancy probably arises because ATSDR referred to applied dose, instead of absorbed dose.

2.3 Acute toxicity/irritation

2.3.1 Introduction

The term 'acute toxicity' is used to describe the adverse effects on health which may result from a single exposure to a substance via ingestion, dermal contact or inhalation. An aspect of these acute effects is the skin and eye irritation and the dermal and respiratory sensitisation that may be caused by a substance. A sensitiser is a substance which, after first-time exposure, sensitises the immune system in such a way that, on subsequent exposure, a hypersensitivity reaction is elicited.

2.3.2 Acute effects of lead on humans and animals

Animal studies

No studies of acute animal toxicity after inhalatory and dermal exposure were encountered.

No oral LD₅₀ values were found by ATSDR (1999) in the available literature. Some LD_{LO} values were encountered. An LD_{LO} is defined as the lowest dose of a substance, given over any given period of time in one or more divided portions, reported to have caused death. The values listed by ATSDR ranged from 191 to 1366 mg lead/kg bw in dogs, and from 313 to 20,500 mg lead/kg bw in guinea pigs, depending on the inorganic lead compound. Comparisons of LD_{LO}s between compounds and species are difficult because these values are not statistically derived.

Human case reports

No studies regarding ocular and dermal effects in humans are available.

Acute nephrotoxic effects of lead in humans have been observed. These effects seem to be reversible at Pb blood levels ranging from 35-150 µg/dL. The SCOEL (2000) concluded that there is no evidence of nephrotoxic and/or gastrointestinal toxicity at Pb blood levels of 40 µg/dl and below.

Short-term exposure to high levels of Pb causes colic. It typically occurs at Pb blood levels of 100-200 µg/dL in occupationally exposed individuals, although the symptoms of colic have sometimes been noted at levels of 40-60 µg/dL. For children, the EPA has identified a LOAEL of approximately 60-100 µg/dL, based on symptoms such as colic, severe constipation, anorexia and intermittent vomiting observed in patients. Acute lead poisoning with severe gastrointestinal symptoms and/or signs of encephalopathy may occur in some adults at Pb blood levels of 50 to >300 µg/dL, but the data are somewhat ambiguous.

In children, signs of encephalopathy and even death were observed at Pb blood levels of 90-800 µg/dL, while other symptoms of acute lead poisoning were observed at levels of 60-450 µg/dL. In the most susceptible children acute encephalopathy may occur at blood levels of 80-100 µg/dL. The acute encephalopathy caused by lead leads to a greatly increased incidence of permanent neurological and cognitive impairments. Lead poisoning without symptoms of encephalopathy (Pb blood levels >80-100 µg/dL) also have increased incidences of lasting neurological and behavioural damage.

In children, acute lead exposure inhibits the synthesis of cytochrome P-450. No Pb blood levels were reported.

Review

Acute toxic effects of lead observed in humans are nephrotoxicity, colic, encephalopathy, and, in children, even death. Such effects are not to be expected at Pb blood levels of 40 µg/dl and below.

2.4 Repeated/chronic toxicity

2.4.1 Introduction

Repeated dose toxicity comprises the adverse general toxicological effects (i.e. excluding reproductive, genotoxic or carcinogenic effects) occurring as a result of repeated daily exposure to a substance for a part of the expected life span (sub-acute or sub-chronic exposure) or, in the case of chronic exposure, for the whole life span (or the major part of the life span).

For inorganic lead, there are many human studies available, both case studies and epidemiological studies. The large available database gives a good impression of the health effects of lead exposure. Therefore, animal data are only cited when they have added value in respect of the available human data.

Quite a number of human studies are available in which the duration of exposure is relatively well known and the level of exposure is reasonably well represented by measured Pb blood levels. When health effects of lead exposure are derived from studies in occupationally exposed persons, exposure is relevant for both the inhalatory route and ingestion. Non-occupationally exposed persons are mainly exposed by the oral route, with some possible inhalatory exposure. The latter mainly involves children. Therefore, the route of exposure is also fairly well known, although it may to a certain extent be mixed in nature.

2.4.2 Subacute, subchronic and chronic effects of lead on humans and animals

For the sake of completeness we discuss below the results of both animal studies and human case reports. It has to be noted though that, where possible, evaluations based on human case reports are preferable due to the differences in, for instance, *a/o*. bioavailability between experimental animals and humans.

Animal studies

No studies on the effects of dermal exposure to lead were available. A 28-day inhalatory exposure of mice to a concentration of 1.6 mg/m³ lead nitrate provoked an irritative response in the lungs, illustrated by slightly elevated lung weights and a haemorrhagic appearance of these organs. These effects were not observed after 14 days of exposure. Both periods of exposure led to a significant increase in liver weight and to significantly increased lead concentrations in the liver, spleen, thymus, lungs and kidneys. No functional or histopathological observations of the liver were executed. Signs of immunosuppression were observed, probably only local and not systemic in nature. All the effects mentioned here increase with exposure duration. In the rabbit, inhalatory exposure to particulate PbO at 0.028 mg Pb/m³ for three hours over four days led to decreased lung macrophage functioning and lung cell damage. The Pb blood level in lead-exposed rabbits remained near control levels (1-2 µg/dL).

Mice orally exposed to 0.5% lead acetate in drinking water (equal to 605 mg Pb/kg bw/day) during a three-generation study showed an increased mortality rate. The most recent chronic exposure studies, at doses otherwise non-toxic, indicate that lead ingestion is associated with increased blood pressure, which is sustained over a considerable portion of the animal's life span. ATSDR presents two (rat) studies in which NOAELs for this effect were established: 0.03 mg Pb/kg bw/day (five-month study) and 1.4 mg Pb/kg bw/day (18-month study). Both studies were conducted with Pb acetate. The latter study seems to be more relevant, according to ATSDR.

A number of rat studies were performed to determine the effect of oral lead exposure on bone growth and metabolism. Most studies were subchronic in nature; one chronic study was available. Lead exposure was observed to impair bone growth. No NOAELs were reported by ATSDR. LOAELs were 145, 1 and 7.5 mg Pb/kg bw/day in a 26-day, 50-day and 12-month study respectively. All these studies were performed with lead acetate. In the latter study, the LOAEL of 7.5 mg/kg bw corresponded to a Pb blood level of 20-30 µg/dL.

Oral exposure of rats to lead provoked hepatic effects like increased liver weight, pycnosis of Kupfer cells, reduced activity of some liver enzymes, decreased cholesterol, increased serum triglycerides and increased hepatic lipid peroxidation. ATSDR reported three NOAELs for hepatic effects in rats: 11.1 and 5.6 mg Pb/kg bw/day after 20 days and 18 months of exposure respectively.

A number of rat studies strongly suggest that oral exposure to lead during post-natal development has a selective detrimental effect on the rods of the retina in rodents. The LOAELs for the most serious effects (rod degeneration and alterations in rod photoreceptors) were 0.5 mg Pb/kg bw/day after 3 weeks of exposure from birth. Retinal effects have also been reported in other species exposed to lead, e.g. monkeys.

Low level oral exposure of rats to lead has resulted in adverse effects on both the humoral and cellular immune system. Pre-natal and post-natal exposure through lead acetate in drinking water up to an age of 35-45 days at a dose of 2.24 mg Pb/kg bw/day (mean Pb blood level 29.3 µg/dL) resulted in immunosuppression. Pregnant rats were exposed to lead acetate in drinking water during breeding and pregnancy. Immune function in pups and dams was assessed when the pups were 13 weeks of age.

In all dose groups, effects on immunological parameters were observed in the pups, but not in the dams. The lowest dose tested was 11 mg PBX/kg bow/day, which resulted in a PBX blood level of 39.4 µg/dl during pregnancy and lactation.

Human studies

There are no studies regarding dermal effects in humans, nor relevant data on respiratory effects of lead exposure.

Mortality studies available for occupationally lead-exposed workers do not permit firm conclusions as regards a minimum lethal dose, as the results show discrepancies and the studies are limited in design.

Exposure of humans to lead profoundly inhibits haem biosynthesis, leading among other things to increased urinary porphyrins, coproporphyrin and δ -aminolevulinic acid (ALA), increased blood and plasma ALA, increased erythrocyte protoporphyrin (EP) and zinc protoporphyrin (ZPP). Threshold Pb blood values in adults ranged from 20-50 µg/dL and 15-35 µg/dL in children (ATSDR, 1999). Frank anaemia, which is a result of reduced haemoglobin production and shortened life span of erythrocytes, is seen in adults at blood lead concentrations of 80 µg/dL and in children at concentrations of 70 µg/dL (Davidson, 1994). The SCOEL, however, indicated that the clinical significance of such biochemical changes at the lowest levels cited is uncertain. They concluded that subclinical changes in parameters of haem synthesis may occur below 40 µg/dl, but that these are not regarded as 'adverse' (SCOEL, 2000).

Lead exposure may cause growth retardation in children. In one study, children with Pb blood levels of 9-10 µg/dL (median level) were 1.2 cm shorter than children with Pb blood levels below the median. There are indications that the effect on growth is reversible after cessation of (high) exposure to lead.

Chronic exposure to lead provokes irreversible nephrotoxic effects such as progressive interstitial fibrosis, dilation of tubules, atrophy or hyperplasia of tubular epithelial cells, reduced glomerular filtration rate and azotemia. These symptoms occurred in workers with Pb blood levels from 60 to 100 µg/dL, who were exposed for several years (ATSDR, 1999). Most studies of nephrotoxic effects in the general population did not demonstrate any at Pb blood levels below 37.3 µg/dL (WHO, 1995).

Lead encephalopathy is the most severe neurological effect of lead in adults. Eventually this condition may be fatal. However, severe lead encephalopathy only occurs at extremely high Pb blood levels (>400 µg/dL). The lowest blood levels of Pb associated with overt signs of neurotoxicity in adults are 40-60 µg/dL.

At Pb blood levels of 40-80 µg/dL impaired behavioural functions were observed in workers that were exposed for a period of between five and 20 years. However, some measures of cumulative exposure may be better predictors of impaired performance in workers with current blood levels of <40 µg/dL. Symptoms noted were disturbances in oculomotor function, deficits in hand-eye co-ordination and reaction time, reduced cognitive performance (a/o. in IQ-tests), nervousness, mood and impaired coping ability. The most recent evaluation of the SCOEL (2000) concluded that for adults, consistent neurobehavioral effects which should be regarded as adverse appear in a multiplicity of studies at lead blood levels of 40 µg/dl and above. For children, Pb blood levels of 40-60 µg/dL are considered to be markedly elevated, and neurobehavioral

effects are distinct. However, neurobehavioral symptoms may occur at Pb blood levels as low as 11 µg/dL (ATSDR, 1999). There is some evidence of an association between decrease in IQ and Pb blood levels less than 10 µg/dL (WHO, 1995).

Nerve conduction velocities were impaired in workers with Pb blood levels <70 µg/dL and possibly as low as 30 µg/dL (ATSDR, 1999). According to the EPA (1986), the threshold for this effect is not clear (cited by Davidson, 1994). From threshold values of 20-30 µg Pb/dL blood, children have impaired nerve conduction velocities (ATSDR, 1999).

Cumulative exposure to lead may lead to (slightly) impaired postural equilibrium at Pb blood levels around 36 µg/dL. These negative effects seem to be reversible. In children, mean Pb blood levels of 12 µg/dL are associated with reduced performance in postural equilibrium tests. Chelation therapy to remove lead from the body seemed to improve performance in such tests.

In young adults, childhood exposure (20 years earlier) to lead in the environment was associated with worse performance in motor and cognitive function tests, and in some peripheral nerve function tests.

In 1986, the EPA concluded that there is no clear evidence of a blood lead level at which neurotoxic effects did not occur in children. This conclusion was reiterated in 1990 as more evidence became available (Davidson, 1994).

In children, at Pb blood levels of 12-120 µg/dL, conversion of vitamin D into its hormonal form was inhibited in a highly significant way whenif chronic exposure to lead was combined with chronic nutritional deficiency (ATSDR, 1999). In children without a deficient diet, Pb blood level of less than 20 µg/dL did not produce effects on calciotrope hormones, including parathyroid hormone, calcitonin and 1,25-dihydroxy-vitamin D. Pb blood levels above 20 µg/dL decreased 1,25-dihydroxy-vitamin D levels (WHO, 1995).

There is considerable scientific debate as to whether there is a causal relationship between repeated lead exposure and hypertension. If such a relationship exists, the correlation is a weak one.

There is evidence that workers with an average Pb blood level of ±56 µg/dL over a period of several years may experience depression of thyroid activity (ATSDR, 1999; WHO, 1995). Current Pb blood levels did not show such an association (WHO, 1995). If such a correlation exists, it will be a weak one. In children, there is no evidence of such an effect (ATSDR, 1999).

There are some indications of effects on the immune system in workers exposed to lead. However, the data are limited and inconsistent.

Review

In humans, children are more sensitive to the toxic effects of lead. In adults, inhibition of haem biosynthesis is associated with the lowest Pb blood levels (20-50 µg/dL), but below 40 µg Pb/dl blood the resulting subclinical changes tend not to be regarded as adverse. At slightly higher blood levels, signs of neurotoxicity have been observed (e.g. reduced cognitive performance, reduced peripheral nerve conduction velocity) as well

as nephrotoxicity. In children, toxic effects of lead like growth retardation, impaired postural equilibrium and other adverse neurobehavioral symptoms are associated by some authors with Pb blood levels of around 10 µg/dL. Haem biosynthesis is inhibited at slightly higher Pb blood levels. Some evidence indicates that even at levels lower than 10 µg/dL, neurobehavioural effects may occur in children. There are indications that reduction in some neuromotor and cognitive functions due to chronic childhood exposure continue 20 years after cessation of exposure. However, it should be stated that the debate in the scientific community on whether or not such low blood levels truly lead to effects giving cause for concern is not yet concluded. For hypertension, immunosuppression and thyroid activity depression no causal relationship with lead exposure in humans has been established, although some indications for a (weak) correlation do exist. Hypertension was found as an effect upon oral lead exposure during 18 months in rats, and a NOAEL of 1.4-mg Pb/kg bw/day was established. The lowest LOAEL observed in animal studies was 0.5 mg Pb/kg bw/day for serious retina effects in rats after three weeks of oral exposure from birth.

2.5 Reproductive and developmental effects

2.5.1 *Introduction*

Reproductive and developmental toxicity of substances encompasses adverse effects on male and female reproductive performance, such as gonadal function, oestrous cycle, mating behaviour, conception, parturition, lactation and weaning, and the growth and development of the offspring.

2.5.2 *Reproductive and developmental effects of lead on humans and animals*

Animal studies

Studies of developmental and high-dose reproductive effects in animals are not reported here because there are adequate human data available, and because the animal data are in agreement with those from human studies. Low-dose reproductive effects observed in animal studies are reported below.

Female rats orally exposed to lead acetate for 30 days had irregular oestrus cycles at doses of 0.014 mg Pb/kg bw/day (Pb blood level 30 µg/dL) and higher. At a dose of 0.26-mg Pb/kg bw/day (Pb blood level 53 µg/dL) these females exhibited ovarian with follicular cysts with reduction in the number of corpora lutea. Their male counterparts had testicular damage at the higher level (Pb blood level 30 µg/dL) and increased prostatic weight at the lower level (Pb blood level 19 µg/dL).

In male rats treated during three months with ±64 mg Pb/kg bw/day (Pb blood level 12 µg/dL) spermatogenesis was adversely effected, and relative testis weights were lower. Other rat studies also indicated adverse effects on spermatogenesis and/or sperm cells after low oral doses of lead. One rat study was reported that did not observe significant effects on sperm count or morphology after nine months of oral exposure, resulting in Pb blood levels ranging from 4.5 to 67 µg/dL.

One recent lifetime study of male monkeys orally exposed to doses of lead provoking Pb blood levels of 6-20 µg/dL and 22-141 µg/dL showed changes in sperm chromatin structure. Another, similar, study by the same research group demonstrated disruption of the general architecture of the seminiferous epithelium. This effect was observed after lifetime exposure and after exposure during infancy only. The control group and

the group only exposed after infancy did not exhibit this effect. Pb blood levels of the lifetime and post-infancy exposed groups were 35 µg/dL, while the infancy-exposed and control groups had levels of less than 1.0 µg/dL. In both studies many other reproductive parameters were not affected, e.g. sperm count, viability, mobility and morphology, circulating sexual hormones and absolute testis weights.

Oral exposure of female monkeys to a dose of ±1 mg Pb/kg bw/day (Pb blood level ±35 µg/dL) for a period up to 10 years suppressed circulation of LH, FSH and oestradiol, but not of progesterone. Shorter exposure to higher doses (1.3 or 5 mg Pb/kg bw/day) leading to Pb blood levels of 70 µg/dL also resulted in longer and more variable menstrual cycles and shorter menstrual flow, but did not prevent ovulation.

Oral exposure of pregnant and lactating rats and their offspring to doses ≥3.5-mg Pb/kg bw/day delayed vaginal opening in females. The NOAEL for this effect was 0.7 mg Pb/kg bw/day. Similar effects were reported for dams exposed prior to breeding. Oral exposure of adult rats to 40 and 81 mg Pb/kg bw/day provoked reduced serum lactin and LH levels, but did not induce changes in serum testosterone, 17β-oestradiol, FSH, LH, prolactin, TSH and thyroid hormones, nor in the histopathology of the gonads and the thyroid gland.

Human studies

There are clear indications that high levels of lead cause adverse effects on both male and female reproductive functions. Several studies suggest that exposure of pregnant women may lead to an increased risk of miscarriages, stillbirths and low birth weight (SCOEL, 2000). Less is known about reproductive effects of chronic exposure to low levels. Pb blood levels in pregnant women of 10 µg/dL did not cause spontaneous abortions (ATSDR, 1999), implying that the NOAEL might be higher for this effect. Several studies reported that the length of gestation is clearly affected at Pb blood levels of 15 µg/dL and above (WHO, 1995; SCOEL, 2000), but other studies do not seem to support these observations (SCOEL, 2000). Effects on sperm may start to appear at blood levels of 40 µg/dL (ATSDR, 1999; WHO, 1995; SCOEL, 2000). The SCOEL (2000) refers to a Finnish study that observed a significant increase in the risk of spontaneous abortion among the wives of men whose Pb levels in blood was 30 µg/dl or higher during spermatogenesis.

Developmental effects that have been observed in humans following exposure to low levels of lead include reduced birth weight, reduced gestational age and neurobehavioral deficits or delays. Maternal Pb blood levels at which effects were found ranged from ±2.5 to <30 µg/dL. The SCOEL (2000) stated that a definitive threshold for the impairment of cognitive development in newborns and infants cannot be derived from the present literature database, and that there are uncertainties and inconsistencies in the present database. No evidence for major congenital malformations have been found, while one study reported an association with a collection of minor anomalies. Only the association between low levels of Pb exposure and neurobehavioral deficits or delays in offspring is relatively consistent. Prenatal exposure seems to be more predictive of neurobehavioral lead-related problems than post-natal exposure.

Review

Low levels of lead exposure clearly have developmental effects in humans. It is impossible to derive a definitive threshold for maternal blood levels for the impairment of cognitive development in newborns and infants from the present literature database. Other effects could be reduced birth weight, reduced gestational age and neurobehavioral deficits or delays in offspring. Pre-natal exposure seems to be more predictive of neurobehavioral lead-related problems than post-natal exposure. The effects of low exposure on human reproduction are less known. Rat studies indicate adverse effects on male and female reproduction after a 30-day exposure to lead, resulting in Pb blood levels of 19 µg/dL and higher. Long-term monkey studies show adverse effects on spermatogenesis and female sex hormone production at Pb blood levels of 35 µg/dL. From these data it seems that the unborn child is more sensitive to lead exposure than the mother. However, no evidence exists of major congenital malformations due to lead exposure, although one study reported an association with a collection of minor anomalies. At low levels of lead exposure, effects on human male and female reproduction may exist. The limited human data and the animal data seem to indicate that, should they exist, it will be at levels of lead exposure comparable to those that cause other health problems (see paragraph 2.4.2).

2.6 Carcinogenic effects of lead

2.6.1 Introduction

Substances or preparations are defined as carcinogenic if they induce cancer or increase its incidence after inhalation, ingestion or dermal absorption.

2.6.2 Carcinogenic effects of lead on humans and animals

Animal studies

Genotoxic effects

In vivo studies with mice, rats and monkeys did not provide evidence for clastogenic properties of lead. However, one chronic study with monkeys showed that when given a cadmium-deficient diet, and only then, severe chromosome aberrations occurred as a consequence of lead exposure (ATSDR, 1999). WHO (1995) concluded, basing itself on the *in vivo* studies and also on *in vitro* studies, that lead is a weak mutagen in mammalian systems, but a strong mitogen³.

Carcinogenicity

The available data on carcinogenicity following ingestion of lead by laboratory animals indicate that lead acetate and phosphate are carcinogenic, and that the most common tumour site is the kidney. However, the extremely high cumulative doses of lead used in these studies are difficult to extrapolate to low-level exposure in humans, and do not provide sufficient basis for quantitative risk assessment. Moreover, the effects observed could be a consequence of non-specific tissue damage induced by the high lead doses, and not a direct effect of lead (ATSDR, 1999). In rats, no evidence of renal tumours was found at doses below 200 mg/l in drinking water (WHO, 1995). IARC (1987) and the EPA (2001) consider inorganic lead to be an animal carcinogen.

³ A mitogen is a substance that causes increased cell proliferation. An increase of cell proliferation activity may promote cancer in the presence of a genotoxic carcinogen.

Human studies

Genotoxic effects

Occupational exposure to lead is associated with increased mitotic activity in peripheral lymphocytes, increased rate of abnormal mitosis and increased incidence of chromosomal aberrations and sister chromatid exchange, at Pb blood levels ranging from 22 to 89 µg/dL. However, the data on chromosomal aberrations and sister chromatid exchange are contradictory, as other studies covering similar Pb blood level ranges did not demonstrate any association between these parameters and lead exposure as measured by Pb blood levels. A volunteer study in which men were orally exposed for 49 days to doses of lead that kept Pb blood levels at around 40 µg/dL also showed no increase in the frequency of chromosome aberrations nor in mitotic activity in lymphocytes.

Carcinogenicity

The available studies are not sufficient to determine the carcinogenicity of lead in humans, because of limitations in design and data reporting. However, one historical cohort mortality study reported increased incidences of renal cancer. The high level group showed a standard mortality rate (SMR) of 2.39 for this effect. Furthermore, two cases of renal cancer have been reported in workers exposed to lead with symptoms of lead poisoning and high Pb blood levels. In at least one of these cases, the tumour contained a high level of lead and had histopathological characteristics similar to those of kidney tumours provoked by lead in animals (ATSDR, 1999). According to WHO (1995), which used essentially the same study database, there is no association between renal cancer and lead exposure. A similar conclusion is reached by Davidson (1994). Both the EPA (2001) and IARC (1987) consider the database inadequate to evaluate carcinogenic risks of lead and inorganic lead compounds to humans. The former classifies lead and inorganic lead compounds as probably carcinogenic to humans, the latter as possibly carcinogenic (group 2B). At a recent IARC conference held in Gargnano, Italy, it was stated that lead should be regarded as a proven animal carcinogen, and that recent data on cancer risks of workers exposed to lead would probably justify a re-evaluation in the near future (SCOEL, 2000).

Review

Lead is considered an animal carcinogen, and may be a human carcinogen. WHO classifies lead as a weak mutagen and a strong mitogen in mammals. The mutagenic properties of lead may be enhanced by a cadmium-deficient diet, as shown by a chronic monkey study. A possible genotoxic mechanism underlying lead carcinogenicity cannot be excluded, but does not seem likely, at least not under normal dietary conditions.

2.7 (Possible) human health risk standard with regard to exposure to lead

2.7.1 Introduction

Occupational exposure to lead used to be primarily through inhalation, though with the low current OELs ingestion now also becomes relatively relevant. For the public in general the reverse is true: most exposure is through ingestion, with some exposure through inhalation. In any case, the toxic effects of lead are independent of the route of exposure, and correlated with the absorbed dose as expressed in the Pb blood level.

According to the EPA (2001), the degree of uncertainty regarding the health effects of lead is very low. The critical effects that occur as a result of exposure to lead (changes in levels of certain blood enzymes, elevation of blood pressure, and neurobehavioral deficits in children) occur at exposure levels (measured as blood lead) so low as to be essentially without a threshold. Therefore, the EPA's Reference Dose (RfD) Work Group considers it inappropriate to develop a RfD for inorganic lead. It should be noted that the most recent revision of this assessment was in 1991.

ATSDR (1999) has decided not to derive maximum residue levels (MRLs) because no clear threshold for some more sensitive effects in humans has been identified. In addition, deriving a MRL would ignore the significant body of literature on lead health effects in relation to Pb blood levels, as MRLs only address external levels of exposure. ATSDR has developed an approach which uses site-specific exposure data to estimate internal doses as measured by Pb blood levels.

Other regulatory authorities and institutions did establish limit values, some of which are summarised in section 2.7.5. The motivation behind a few limit values is dealt with in sections 2.7.2 to 2.7.4.

2.7.2 *Chronic/long-term exposure (general population)*

Lead was evaluated by a WHO Working Group developing air quality guidelines for Europe (WHO, 1987). Based on the assumption that Pb blood levels in 98% of the population would be maintained below 0.96 $\mu\text{mol/l}$ (20 $\mu\text{g/dL}$), a guideline value in the range of 0.5 to 1.0 $\mu\text{g/m}^3$ (long-term average, such as annual mean) was recommended (WHO, 1995).

When deriving the guideline value for drinking water, WHO considered it appropriate to use the TDI approach, as there is evidence from human studies that adverse effects other than cancer may occur at very low lead levels, and because a guideline value thus derived would also be protective for carcinogenic effects. WHO used the provisional tolerable weekly intake (PTWI) set by JECFA. In 1986, the latter organisation had established a PTWI of 25 μg of lead per kg of body weight (equivalent to 3.5 $\mu\text{g/kg}$ of body weight per day) for infants and children, which took account of the fact that lead is a cumulative poison so that any increase in the body burden of lead should be avoided. The PTWI was based on metabolic studies in infants showing that a mean daily intake of 3-4 $\mu\text{g/kg}$ of body weight was not associated with an increase in blood lead levels or in the body burden of lead, whereas an intake of 5 $\mu\text{g/kg}$ of body weight or more resulted in lead retention. This PTWI was confirmed by JECFA in 1993 and extended to all age groups. Based on the JECFA PTWI and the assumption of a 50% allocation to drinking water for a 5 kg bottle-fed infant consuming 0.75 litres of drinking-water per day, the guideline value established by the WHO for drinking water is 0.01 mg/l. As infants are considered to be the most sensitive subgroup of the population, this guideline value will also be protective for other age groups (WHO, 1996).

The JECFA PTWI of 25 μg of lead per kg of body weight has been adopted by the RIVM and the Health Council of the Netherlands as a human limit value for the general population. Based on this, RIVM derived a maximum permissible risk level of 3.6 $\mu\text{g/kg}$ of body weight per day (Baars et al, 2001).

2.7.3 *Short-term exposure (general population)*

No short-term exposure limits for the general population have been encountered.

2.7.4 *Short-term and long-term occupational limits*

Only two short-term occupational limits were encountered: a recommendation for lead and its inorganic compounds (except lead arsenate and chromate) not to exceed a concentration of 1 mg Pb/m³ in air for more than 30 minutes in an 8-hour period, set by DFG (DFG, 2000), and a ceiling value for lead arsenate of 0.04 mg Pb/m³ in air, set by NIOSH (ACGIH, 1999). Long-term occupational limits found for lead and inorganic compounds in air range from 0.05 to 0.15 mg Pb/m³. NIOSH sets a clearly different limit in air for lead arsenate and chromate compared with other inorganic lead compounds: 0.008 and 0.004 mg Pb/m³ respectively, instead of <0.1 mg Pb/m³. Occupational limit values for Pb blood levels set by different regulatory authorities for the worker in general range from 30-60 µg/dL. Some regulatory authorities indicate women of reproductive age and young persons as groups at risk among the worker population, and set lower values for them (25-30 µg/dL for women and 40-50 µg/dL for young persons).

2.7.5 *Overview of limit values*

On the basis of the information in the preceding sections, various regulatory authorities have derived limit values for lead. They are reflected in table 2.1. As for blood levels, the US Centre for Disease Control and Prevention (CDC) uses a 'level of concern' of 10 µg/dl. for young children, mainly because of possible effects like growth retardation and a possible relation between decrease in IQ and lead blood levels at these low concentrations (section 2.4.2). As for adults, the blood level values reported in 2.7.4 of 30 µg Pb/dl blood for males/females might be used (cf. SCOEL, 2000).

Table 2.1: Limit values for lead

Medium	Lead species	Limit name	Limit value	Regulatory authority	Source
<i>General population</i>					
Air	Lead	guideline value	0.5-1.0 µg/m ³	WHO	WHO, 1995
Drinking water	Lead	guideline value	0.01 mg/l	WHO	WHO, 1996
n/a	Lead	PTWI	25 µg/kg bw	JECFA	WHO, 1996
<i>Workers</i>					
Air	Lead	OEL-TWA	0.15 mg/m ³	EU	EU, 2001
Air	Lead & compounds (not being lead alkyls)	OES-TWA	0.15 mg/m ³	HSE	HSE, 1999
Air	Lead & inorganic compounds	TLV-TWA	0.05 mg/m ³	ACGIH	ACGIH, 1999 and 2000
Air	Lead & inorganic compounds	PEL-TWA	0.05 mg/m ³	OSHA	ACGIH, 1999
Air	Lead & inorganic compounds	REL-TWA	<0.1 mg/m ³	NIOSH	ACGIH, 1999
Air	Lead & inorganic compounds	ELV-TWA	0.05 mg/m ³	Arbejdstilsynet (Denmark)	Arbejdstilsynet, 1996
Air	Lead & inorganic compounds	MAC-TWA	0.15 mg/m ³	SZW	SZW, 2000
Air	Lead & inorganic compounds	OELV-TWA	0.1 mg/m ³	SNBOSH	SNBOSH, 1996
Air	Lead & inorganic compounds (except lead arsenate and chromate)	(Proposed) MAK- 'peak' (max 30 minutes in an 8 hr. period)	1 mg/m ³	DFG	DFG, 2000

Medium	Lead species	Limit name	Limit value	Regulatory authority	Source
Air	Lead & inorganic compounds (except lead arsenate and chromate)	(Proposed) MAK-TWA	0.1 mg/m ³	DFG	DFG, 2000
Air	Lead arsenate	TLV-TWA	0.15 mg/m ³	ACGIH	ACGIH, 1999 and 2000
Air	Lead arsenate	PEL-TWA	0.04 mg/m ³	OSHA	ACGIH, 1999
Air	Lead arsenate	CEIL	0.008 mg/m ³	NIOSH	ACGIH, 1999
Air	Lead chromate	TLV-TWA	0.05 mg/m ³	ACGIH	ACGIH, 1999 and 2000
Air	Lead chromate	REL-TWA	0.004 mg/m ³	NIOSH	ACGIH, 1999 and 2000
Air	Lead phosphate	TLV-TWA	0.05 mg/m ³	ACGIH	ACGIH, 1999
Air	Lead phosphate	PEL-TWA	0.05 mg/m ³	OSHA	ACGIH, 1999
Air	Lead phosphate	REL-TWA	<0.1 mg/m ³	NIOSH	ACGIH, 1999
Blood	Lead	(Proposed) BAT (women <45 years)	30 µg/dL	DFG	DFG, 2000
Blood	Lead	(Proposed) BAT (other workers)	40 µg/dL	DFG	DFG, 2000
Blood	Lead & compounds (not being lead alkyls)	Action level (women of repro. age)	25 µg/dL	HSE	HSE, 1999
Blood	Lead & compounds (not being lead alkyls)	Action level (persons under 18)	40 µg/dL	HSE	HSE, 1999
Blood	Lead & compounds (not being lead alkyls)	Action level (any other workers)	50 µg/dL	HSE	HSE, 1999
Blood	Lead & compounds (not being lead alkyls)	Suspension level (women of repro. age)	30 µg/dL	HSE	HSE, 1999
Blood	Lead & compounds (not being lead alkyls)	Suspension level (persons under 18)	50 µg/dL	HSE	HSE, 1999
Blood	Lead & compounds (not being lead alkyls)	Suspension level (any other workers)	60 µg/dL	HSE	HSE, 1999
Blood	Lead & inorganic compounds	BEI	30 µg/dL	ACGIH	ACGIH, 2000
Blood	Lead & inorganic compounds	REL-TWA	<0.06 mg/100 g whole blood	NIOSH	ACGIH, 1999
Blood	Lead phosphate	REL-TWA	<0.06 mg/100 g whole blood	NIOSH	ACGIH, 1999

Note: for some compounds, such as lead arsenate and chromate, standards are related to the anions rather than lead

2.8 Ecotoxicity of lead

2.8.1 Introduction

As indicated before, this review is limited to inorganic lead as this is the predominant form of lead in the environment. Inorganic lead appears mainly in the environment as metal or as its salts, oxides or sulphides (ATSDR, 1999). For the Dutch environment, RIVM concludes that alkyl leads (the major organic form of lead) is of less ecotoxicological importance than inorganic lead (Janus et al, 1999). Most of the information is derived from a review by WHO (1989).

2.8.2 *Model ecosystems*

In aquatic and aquatic/terrestrial model ecosystems, uptake by primary producers and consumers seems to be determined by the bioavailability of lead. Bioavailability is generally much lower whenever organic material, sediment or mineral particles (e.g. clay) are present. In many organisms, it is unclear whether lead is adsorbed onto the organism or actually taken up. Consumers take up lead from their contaminated food, often to high concentrations, but this is without biomagnification. (WHO, 1989)

2.8.3 *Aquatic organisms*

Uptake, loss and accumulation in aquatic organisms

The uptake and accumulation of lead by aquatic organisms from water and sediment are influenced by various environmental factors such as temperature, salinity and pH, as well as humic and alginic acid content. In contaminated aquatic systems, almost all of the lead is tightly bound. Only a minor fraction is dissolved in water, even in the interstitial water of sediments. (WHO, 1989)

Plants accumulate lead in industrially (and naturally) contaminated environments. Uptake can occur from both water and sediment, although uptake from sediment usually predominates. Lead levels decrease with distance from the source. (WHO, 1989)

In shellfish, lead concentrations are higher in the calcium-rich shell than in the soft tissue; they relate to concentrations in sediment. The lead uptake by fish reaches equilibrium only after a number of weeks of exposure. Lead is accumulated mostly in gill, liver, kidney and bone. Fish eggs show increasing lead levels with increased exposure concentration, and there are indications that lead is present on the egg surface but not accumulated in the embryo. In dolphins, lead is transferred from mothers to offspring during foetal development and lactation (WHO, 1989). As in terrestrial mammals, such as humans and rats, this is probably related to mobilisation of calcium from maternal bone during pregnancy (see section 2.4). (WHO, 1989)

Biomagnification of inorganic lead in the aquatic food chain is not probable, as the levels of Pb as well as the bioconcentration factors decrease as the trophic level rises. This is partly explained by the fact that, in vertebrates, lead is mainly stored in bone, which reduces the risk of lead transmission to other organisms in the food chain (Janus et al, 1999).

Toxicity to aquatic organisms

One of the most important factors that influences the aquatic toxicity of lead is the free ionic concentration, which affects the bioavailability of lead for organisms. The toxicity of lead salts is strongly dependent on environmental conditions, such as water hardness, pH, dissolved organic carbon (DOC) and salinity, a fact that has not been adequately considered in most toxicity studies. Results of toxicity tests should be treated with caution unless dissolved lead has been measured. (WHO, 1989)

There is little evidence for effects of lead on aquatic plants at concentrations below 1 to 15 mg/L. Many studies of aquatic plants have been made in sediment-free systems. However, the addition of uncontaminated sediment reduces the toxicity of lead to aquatic plants by reducing its availability (WHO, 1989). The results of experiments on the toxicity of lead salts to aquatic invertebrates are difficult to interpret due to the variations in experimental conditions and the lack of a standardised method for

determining lead concentrations in water. In the form of simple salts, lead is acutely toxic to aquatic invertebrates at concentrations above 0.1 to >40 mg/L for freshwater-organisms and above 2.5 to >500 mg/L for marine organisms. In communities, some populations of organisms are more sensitive than others, and community structure may be adversely affected by lead contamination. However, populations from polluted areas can show more tolerance to lead than those from non-polluted areas. In other organisms, adaptation to hypoxic conditions can be hindered by high lead concentrations (WHO, 1989). In fish, the 96-h LC₅₀ values for inorganic lead range from 1 to 27 mg dissolved lead/L; nominal concentrations being up to 100 times higher in hard water. Long-term exposure of adult fish to inorganic lead induces sub-lethal effects on morphology, amino levulinic acid dehydrates (delta-ALAD) and other enzyme activities, and avoidance behaviour at available lead concentrations of 10-100 mg/L. Juvenile stages are generally more sensitive than adults. Typical stages of lead toxicity include spinal deformity and blackening of the caudal region. Eggs are often less sensitive because lead is adsorbed onto the egg surface and excluded from the embryo. The maximum acceptable toxicant limit (MATC) for inorganic lead has been determined for several species under different conditions, and results range from 0.04 to 0.198 mg/L (WHO, 1989)

There is evidence that frog and toad eggs are sensitive to nominal lead concentrations of less than 1.0 mg/L in standing water and 0.04 mg/L in flow-through systems; arrested development and delayed hatching have been observed. For adult frogs, there are no significant effects below 5 mg/L in aqueous solution, but lead in the diet at 10 mg/kg food has some biochemical effect. (WHO, 1989)

Table 2.2 Overview of chronic NOECs for freshwater organisms

Taxonomic group	NOEC (µg Pb/L)		Toxicological endpoints
	Mean ± SD (n)	Range	
Bacteria	1,183 ± 683 (3)	450-1,800	Growth
Unicellular algae	20,055 ± 55,744 (15)	10-200,000	Growth
Multicellular algae	1,033 ± 945 (3)	300-2,100	Growth
Protozoas (fresh water)	403 ± 604 (4)	20-1,300	Growth, reproduction
Molluscs	204 ± 317 (3)	12-570	Hatching, survival
Crustaceans	502 ± 913 (8)	1-2,500	Reproduction, survival, growth
Fish	77 ± 74 (17)	7-250	Reproduction, survival, growth, abnormalities, development, hatching

Data summarised from Janus (2000).

Table 2.3 Overview of chronic NOECs for saltwater organisms

Taxonomic group	NOEC (µg Pb/L)		Toxicological endpoints
	Mean ± SD (n)	Range	
Algae	23 ± 32 (3)	0.1-60	Growth, reproduction
Protozoas	150 (1)	--	Population density
Coelenterates	300 (1)	--	Growth
Annelids	3,833 ± 5,346 (3)	500-10,000	Growth, reproduction
Molluscs	1,400 ± 2,400 (4)	200-5,000	Survival, growth
Crustaceans	269 ± 487 (4)	10-1,000	Growth, embryonic development, development, reproduction

Data summarised from Janus (2000).

2.8.4 *Terrestrial organisms*

Uptake, loss and accumulation in terrestrial organisms

In higher plants, the majority of lead is associated with the cell wall. Some lead that passes into the plant root cell can be combined with new cell wall material and subsequently removed from the cytoplasm to the cell wall. Of the lead remaining in the root cell, there is evidence of very little translocation to other parts of the plant because the concentration of lead in shoot and leaf tissue is usually much lower than in root. Foliar uptake of lead occurs, but only to a very limited extent. Plants accumulate lead in industrially contaminated areas. Levels decrease with distance from the source, and are lowest during the active growing season. Mosses accumulate lead from the atmosphere and are often used as biological monitors of airborne lead. (WHO, 1989)

In animals, there is a positive correlation between tissue and dietary concentrations, although tissue concentrations are almost always lower. The distribution of lead within animals is closely associated with calcium metabolism (WHO, 1989). A more detailed description of the kinetics and metabolism of lead in mammals, with an emphasis on man, can be found in section 2.4 and further. Lead shot is typically trapped in the gizzard of birds where it is slowly ground down resulting in the release of lead, which causes severe lead contamination: in organs, high levels of lead are found in blood, kidney, liver and bone. Elevated lead levels are found in terrestrial invertebrates and vertebrates from contaminated areas (WHO, 1989).

Accumulation of lead in the terrestrial food chain does not seem to be of great importance, for reasons similar to those brought forward for the aquatic food chain. An exception is the transmission of lead shot from birds that have ingested this lead species to raptors. In the USA this has been a major problem, principally in areas of large-scale hunting of waterfowl. For this reason, the US government has banned the use of lead shot (Janus et al, 1999).

Toxicity to terrestrial organisms

The tendency of inorganic lead to form highly insoluble salts and complexes with various anions, together with its tight binding to soils, drastically reduces its bioavailability to terrestrial plants via the roots. Translocation of the ion in plants is limited, and most bound lead stays at root or leaf surfaces. As a result, in most experimental studies on lead toxicity, high lead concentrations in the range of 100 to 1000 mg/kg soil are needed to cause visible toxic effects on photosynthesis, growth or other parameters. Thus, lead is only likely to affect plants at sites of very high environmental concentrations (WHO, 1989).

Ingestion of lead-contaminated bacteria and fungi by nematodes leads to impaired reproduction. Woodlice seem unusually tolerant to lead, since prolonged exposure to soil or grass litter containing externally added lead salts had no effect. Caterpillars maintained on a diet containing lead salts show symptoms of toxicity leading to impaired development and reproduction. The information available is too meagre to quantify the risks to invertebrates during decomposition of lead-contaminated litter.

Lead salts are only toxic to birds at a high dietary dosage (100 mg/kg or more). Almost all the experimental work is on chickens and other gallinaceous birds. Exposure of quail from hatching up to reproductive age resulted in effects on egg production at dietary levels of 10 mg/kg food. Although a variety of effects have been reported at high dosage, most can be explained as a primary effect of food consumption. Diarrhoea and lack of appetite, leading to anorexia and weight loss, are the primary effects of lead salts. Since there is no experimental evidence to assess effects on other bird species, it is necessary to assume a comparable sensitivity. If this is so, then it is highly improbable that environmental exposure would cause adverse effects. Metallic lead is not toxic to birds except at very high dosage, when administered in the form of powder. It is highly toxic to birds when given as lead shot; ingestion of a single pellet of lead shot can be fatal for some birds. The sensitivity varies between species and is dependent on diet. Since birds have been found in the wild with large numbers of lead shot in the gizzard (20 is not unusual), this poses a major hazard to those species feeding on river margins and in fields where a great deal of shot has accumulated. (WHO, 1989)

There are many reports of lead levels in wild mammals but few reports of toxic effects of the metal in the wild or in non-laboratory species. Wild rats showed similar effects to their laboratory counterparts. For a review on lead toxicity in laboratory mammals and men, reference is made to section 2.2 and further.

Table 2.4 Overview of chronic NOECs for soil microbe-mediated processes

Toxicological endpoint	NOEC (mg Pb/kg d.w.) [*]	
	Mean ± SD (n)	Range
C-mineralisation	1,123 ± 1,534 (11)	15-5,200
N-mineralisation	781 ± 473 (10)	180-1,500
Enzyme activities	1,062 ± 1,773 (28)	49-7,700

Data summarised from Janus (2000).

* For standard soil.

Table 2.5 Overview of chronic NOECs for terrestrial organisms

Taxonomic group	NOEC (mg Pb/kg d.w.) [*]		Toxicological endpoints
	Mean ± SD (n)	Range	
Plants	878 ± 538 (12)	120-1,500	Growth, yield
Oligochaetes	815 ± 663 (6)	170-2.00	Growth, reproduction
Gastropods	1,000 (1)	--	Survival, food consumption, growth (weight)
Crustaceans	40 (1)	--	Unspecified
Insects	1,100 (1)	--	Reproduction, growth
Mites	430 (1)	--	Reproduction, growth, survival

Data summarised from Janus (2000).

* For standard soil.

2.8.5

Target and intervention values for soil sanitation established by the Dutch government

As one of the few authorities in the world, the Dutch government establishes target and intervention values for soil sanitation. For the sake of completeness we discuss them here to give some insight on how target values for soil quality can be derived. The soil sanitation intervention value is defined as the concentration level of a substance above which there is a case of serious contamination. When this level is exceeded, the functional properties that soil possesses for humans, plants or animals are considered to

be seriously affected, or threatened with this. The intervention values are established for soil/sediment and groundwater, and are valid for land and water soils. The Dutch government also sets target values for soil/sediment and groundwater. These are concentration levels at or below which the soil or groundwater is considered clean with respect to the substance concerned (VROM, 2000).

At the request of the ministry (VROM), RIVM has evaluated the intervention values for lead. RIVM proposed an intervention value for soil/sediment of 450-575 mg Pb/kg dry matter. This value is based on an ecotoxicological serious soil contamination concentration (ECOTOX SCC) of 450-575 mg Pb/kg dry matter and a human-toxicological serious soil contamination concentration (HUM-TOX SCC) of 450-670 mg Pb/kg dry matter. The exact outcome within the ranges presented depends on choices made for certain default parameters (e.g. soil type correction factor), on whether or not an added risk-approach is used, and the extent to which new data are included. The ECOTOX SCC was derived using toxicity data for soil organisms' biological processes selected from literature, and normalisation of the data used with respect to soil type. The HUM-TOX SCC was derived from the human-toxicological maximal tolerable risk (human MTR) of 3.6 µg/kg bw/day⁴, a proposed preliminary relative bioavailability factor for lead in soil of 0.6, and estimates of possible lead exposure of children (the population at risk) by soil ingestion and food consumption. RIVM proposed an intervention value for groundwater of 32-41 µg/L, based on the intervention value for soil/sediment and the soil/water partition coefficient for lead proposed by it. (Lijzen et al, 1999)

Based on this RIVM advice, VROM decided not to change the present intervention values for lead: 530 mg Pb/kg dry matter for soil and 75 µg dissolved Pb/L for groundwater. The target value for lead is set at 85 mg Pb/kg dry matter (equal to the country wide background value) for soil, at 15 µg dissolved Pb/L for shallow groundwater and 1.7 µg dissolved Pb/L for deep groundwater. (VROM, 2000)

No other documentation on intervention values was found in the available literature and databases.

⁴ Equal to 1/7 of the PWI established by JECFA. WHO used this value as a TDI to calculate the drinking water limit (WHO, 1996; see also section 2.7).

3 Lead Uses, Trends and Emissions

3.1 Introduction

This chapter presents estimates of present and future emissions of lead into the environment. The emissions are related to different groups of products including both intentional applications of lead in products and products containing lead as a trace contaminant. Emissions will be specified for the different life-cycle stages of the product groups, i.e. production, use and waste management.

In section 3.2, a short description of the consumption of lead in the world and the European Union is given. The section ends with a classification of lead applications that will be used in this report. In Section 3.3, the assumptions used to estimate trends in consumption, the disposal of products and the emissions during the different life-cycle stages are stated. In fact, this section describes the methodological approach towards the inventory of uses, trends and emissions. Sections 3.4-3.11 present detailed results for trends in consumption and disposal of the different product groups, and the accompanying emissions. Section 3.12 summarises these results in the form of emission figures for 2000 and 2030.

3.2 Lead applications and uses: an overview

3.2.1 Introduction

This section gives a generic overview of lead applications and some initial data on uses. We make a distinction between (1) functional applications of lead, and (2) products containing lead as a contaminant.

The functional applications of lead make use of its unique properties like flexibility, corrosion resistance and high density. Its flexibility and low melting point makes it easy to handle and fashion. Its high resistance to corrosion makes it suitable for weatherproofing buildings (lead sheet, lead paint) and for the equipment used in the manufacture of acids. Its high density makes it appropriate as a shield against radiation and noise. However the most important use of lead today is in the use of lead-acid batteries to store electrical power.

Besides the intentional use of lead in products, lead is also present in products as a contaminant. Products may contain lead because of the natural presence of lead in ores, as is the case for example in (phosphate) fertilisers and fossil fuels, especially coal. Other lead contamination in products has an anthropogenic origin, such as lead in sewage sludge, manure and ashes and slags of the waste incineration and steel industry. All of these products represent waste materials at the end of the chain, which may be reused in agriculture (i.e. the use of sewage sludge and manure as fertiliser), the building sector (i.e. the use of fly ash in concrete and other building materials) and road construction (i.e. the use of slag in road construction).

Below, we will discuss in somewhat more detail the functional and non-intentional uses of lead. The data given are meant as a general background; the specific emission data used further on in this project will be discussed in the next sections. We have chosen to use the statistics of the International Lead and Zinc Study Group (ILZSG) as a basis. This source gives the most comprehensive public consumption and production data related to lead. Sometimes, other organisations also publish data on consumption and use of lead for individual applications, and their values may not always be exactly the same as those of the ILZSG⁵. However, in general such differences were not more than 20-30 % compared to the ILZSG data we used, which can be considered of minor importance given the usual uncertainties in the type of SFA we performed here⁶. Since we needed data that were consistent for different product groups and for the whole of Europe we decided to stick to the ILZSG data.

Unfortunately, the ILZSG does not give data on lead use, let alone lead use by product category, for the accession countries Poland, the Czech Republic and Slovenia. From our literature research for this study, which was meant to concentrate on evaluating and combining information from a number of major review documents, no such information could be obtained either. A crosscheck with the main industry organisations indicated they had no such information either⁷.

3.2.2 *Functional use of lead*

Over the last ten years the consumption of lead has increased by 23%. Table 3.1 shows the world consumption of lead in today's society (ILZSG, 2001). The consumption in Europe is about 33% of global consumption. Table 3.2 shows the major intentional applications of lead on a worldwide scale (1997). By far the largest application of lead is the use of lead in lead-acid batteries, at 73 % of global consumption. The second largest application is the use of lead as a pigment and in other compounds (e.g. as a stabiliser in plastics and rubber), 11% of global consumption. The third largest application is the use of rolled and extruded lead, 6% of global consumption. About 80% of this rolled and extruded lead is used as building materials. Production in the West represents 82% of the total global production. Consumption in the West represents 87% of the total global consumption. In 1998 the lead production was based on 1980 kilotonnes primary lead and 2970 kilotonnes secondary lead (ILZSG, 1999).

Table 3.1 Global consumption of lead (kilotonnes) (ILZSG, 2001)

	1996	1997	1998	1999
Europe	1942	1968	1952	1999
Africa	120	121	132	127
America	2056	2085	2179	2251
Asia	1795	1770	1673	1810
Oceania	74	70	64	64
World total	5987	6014	6000	6251

⁵ For instance, the European Lead Sheet Industry Association (ELSIA) and the EUROMETEAUX 2000 annual reports might give somewhat different values than those used here.

⁶ Typical reasons why differences in data might occur: different definitions of product groups, different definitions of industry sectors, reporting of production or consumption data, etc. As we will explain later, the mass flows will be used to estimate future emissions making use of emission factors per industry sector. The uncertainty in the emission factors is considerably higher than the 20-30 % uncertainty in production/use data of lead from different sources.

Table 3.2 Applications of lead and the relative lead demand (worldwide scale, 1997) (ILZSG, 1999)

	%
Batteries	72.5
Rolled and extruded	6.1
Pigments and other compounds	10.9
Shot/ammunition	2.4
Alloys	2.4
Cable sheathing	1.7
Petrol additives	0.9
Miscellaneous	3.1

Table 3.3 shows the long-term consumption of lead by industry in the Western world, OECD⁸ countries, from 1970 to 1990 (OECD, 1993). Table 3.4 shows an estimate of the consumption of lead by industry in the European Union (EU15⁹) over the past 10 years (ILZSG, 2000). For the European Union it is assumed that these production figures of products also resemble the consumption of these products in the EU15, unless otherwise indicated.

3.2.3 Non-intentional use of lead

Lead may occur as an element in other (non-lead) ores. The lead in waste materials such as sewage sludge, compost and manure originates from many different sources 'upstream' of the lead chain. In some studies the primary sources are traced back into the chain. For example, in the Netherlands the lead in sewage sludge in 1990 was caused by the corrosion of lead from lead sheets in the building sector (58%), industrial emissions (8%), the corrosion of lead from water pipes (4%), faeces and urine (4%), the deposition of lead from air (petrol 4%, others 4%) and unknown sources (17%) (Annema et al, 1995). For Europe, however, the origins of lead in sewage sludge might be different from the Dutch situation. In the Netherlands the connection to sewer systems and the use of lead sheet in the building sector is much higher than the European average, and relatively little lead is still used in water pipes. For a proper origin analysis the total lead chain should be modelled for the European Union. Such an analysis is, however, beyond the scope of this project.

The lead content in manure is a result of the internal agricultural cycle. Lead is introduced into the agricultural sector through the use of fertiliser, by atmospheric deposition and by the 'recycling' of compost and sewage sludge (see above). Due to the process of closed-loop recycling (grass / cattle / manure / grass) the concentration of lead in manure will slowly increase, as is shown for the Dutch situation by Van der Voet et al.

⁷ Personal communication with Dr. A. Bush and D. Wilson, Lead Development Association, London, and Prof. Ian Thornton, Imperial College of Science, Technology and Medicine, London, July 2001

⁸ OECD countries are EU15 and Iceland, Norway, Switzerland, Finland, Turkey, Canada, United States, Japan, Australia and New Zealand.

⁹ Belgium, Denmark, Germany, Greece, Spain, France, Ireland, Italy, Luxembourg, the Netherlands, Austria, Portugal, Finland, Sweden, and the United Kingdom

Table 3.3 Principle uses of lead in OECD (production figures) (tonnes Pb) (OECD, 1993)

	1970	1975	1980	1985	1990	% per group
Batteries	1182500	1261000	1518000	1838000	2120000	
-SLI (car batteries)	936000	1000000	1200000	1430000	1644000	77
- Motive power	150000	160000	203000	254000	293000	14
- Industrial	96000	100000	114000	145000	165000	8
- Consumer	500	1000	1000	9000	18000	1
Rolled and extruded products	359400	304000	280600	261900	302300	
- Radiation shielding materials	37000	30000	28000	26000	29500	10
- Soundproofing materials	6500	5600	7100	7400	7700	3
- Building materials	182000	188000	191000	193000	243000	80
- Chemical applications	39700	28800	29000	18700	14600	5
- Collapsible tubes	22900	9200	4700	2200	100	0
- Wine bottle capsules	15000	15000	15300	13300	6000	2
- Pipes	56300	28000	6100	3100	1000	0
Pigments and other compounds						
Glass pigments					224000	
- Cathode ray tube	63000	70000	100000	125000	145000	65
- Crystal	50000	45000	60000	50000	55000	25
- Speciality glass/optical	15000	17000	18000	15000	14000	6
- Light bulbs	18000	16000	14000	12000	10000	4
Other pigments and compounds					139000	
- Plastic additives	58000	60000	83000	75000	84000	60
- Glazes	30000	33000	50000	30000	33000	24
- Paints	67000	53000	46000	33000	16000	12
- Ceramics	15000	13000	10000	8000	6000	4
Shot and ammunition	111200	126400	86500	101000	99500	
- Sporting shot	71200	80100	58700	73700	80500	81
- Steel malding shot	40000	46300	27800	27300	19000	19
Alloys						
Solder					80000	
- Electronics	36000	31000	33000	36000	47000	59
- Plumbing	31000	24000	21000	13000	12000	15
- Car radiators	21000	19000	12000	10000	9000	11
- Car body	31000	24000	19000	10000	8000	10
- Food cans	28000	17000	17000	10000	4000	5
Other alloys					38000	
- Brass and bronze	36000	30000	27000	25000	23000	61
- Bearings and bushings	25000	19000	13000	10000	8000	21
- Terne plate	8000	9000	7000	7000	7000	18
Cable Sheathing	384700	298900	243500	157200	153800	
Petrol additives	322000	283000	207000	135000	72000	
Miscellaneous	147000	121000	131500	117500	126000	
- Type metal	40300	27000	17000	10500	6500	5
- Wheel weights	20000	22000	24300	29000	35000	28
- Miscellaneous cast	67700	53000	70000	56000	67800	54
- Galvanising	6700	7300	8100	7600	7200	6
- Annealing	7600	6300	5600	6300	6000	5
- Plating	2400	2400	2800	3000	3500	3

Table 3.4 Principal uses of lead in EU15 (production figures) (kilotonnes Pb) (ILZSG, 2000)

	1993	1994	1995	1996	1997	1998	% 1998
Batteries	743	822	869	870	888	919	58
Rolled and extruded products	214	218	234	221	222	233	15
Pigments and other compounds	203	206	211	196	201	193	12
Shot and ammunition	53	52	55	56	57	59	4
Alloys	37	40	40	36	33	32	2
Cable Sheathing	91	80	78	61	42	35	2
Petrol additives	50	48	47	43	36	38	2
Miscellaneous	72	68	69	69	62	65	4
Total	1463	1533	1603	1551	1540	1574	

(eds., 2000)¹⁰. This issue will be elaborated to some extent in Chapter 4 of this report. Yet again, a comprehensive and detailed analysis of the effects of closed-loop accumulation would require dynamic modelling of the total EU lead chain, which is beyond the scope of this project.

3.2.4 Classification of lead products and uses of products used in this project

In principle, various classifications of products and uses of products can be used in this project. In various national studies, different classifications have been used (e.g. Lassen and Hansen, 1996). However, this project has to cover the EU15 and if possible also some accession countries. For this purpose, in this project of limited scope a pragmatic choice had to be made that allows consistent discussions of all mass flows in the EU. Hence, in view of data availability we decided to stick to the classification of ILZSG (2000), since this source gives consistent use data for the EU over a number of years. Additionally, we added a number of pragmatically chosen categories of non-intentional applications of lead. This implied choosing the following categorisation:

- Batteries;
- Rolled and extruded products;
- Pigments and other compounds;
- Shot and ammunition;
- Alloys;
- Cable Sheathing;
- Petrol additives;
- Miscellaneous;
- Non-intentional use in the agricultural sector (manure, sewage sludge and compost).

¹⁰ However, other sources also come to this conclusion. Examples include Moolenaar (1998), Annema et al. (1995) and the Danish EPA (1998). The discussion does not seem to be *whether* a slow increase takes place, but rather *how rapidly* it takes place (e.g. concentration doublings in soil occurring over hundreds, or thousands, of years; see also Chapter 4).

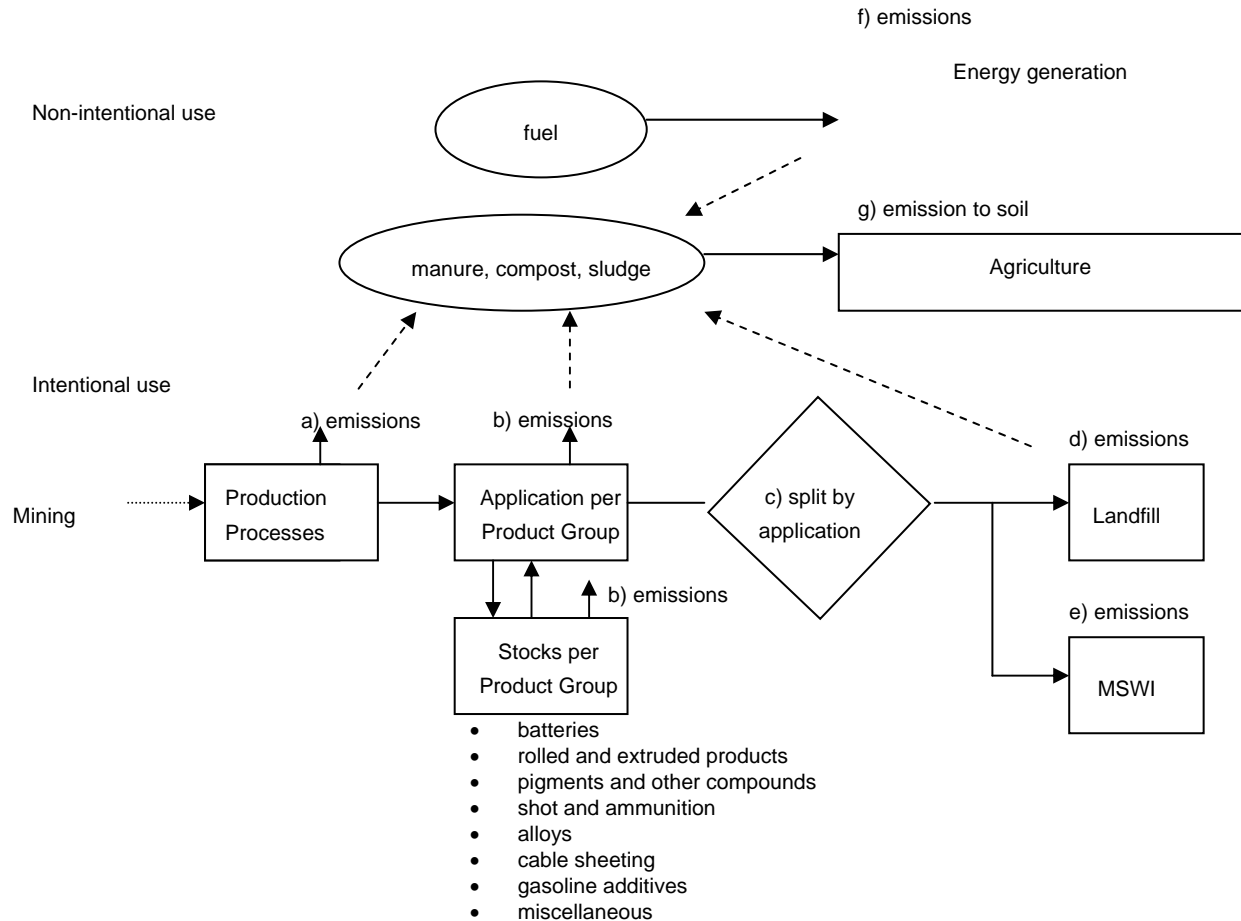
3.3 Methodology for estimating present and future emissions of lead

3.3.1 Introduction

This section discusses the general methodological approach behind the emission inventory and the general assumptions valid for a number of product groups. Details are given for each product group in sections 3.4 to 3.12. All data given were elaborated with the help of a relational spreadsheet model.

The basic structure of the model used is presented in Figure 3.1. For the intentional uses, the model is set up as a Substance Flow Analysis (SFA). For each product group, a distinction is made between:

- production processes;
- applications and related stock formation in society (e.g. new battery use and the number of batteries in use);
- waste flows and the division of waste flows to landfill and incinerators.



- a) Based on EMEP. Allocated to product groups based on volume Pb.
 b) Gasoline: based on EMEP. Other: function of stock and emission factor
 c) Volume per application to waste: based on stock, past production volumes and application lifetime. Split landfill/MSWI: informed assumption.
 d) Function of stock and emission factor.
 e) Based on EMEP. Allocated to product groups based on input volume Pb.
 f) Based on EMEP.
 g) Based on mass balance agriculture.

Figure 3.1: Structure of the emission inventory

For non-intentional use, the model concentrates on emissions from energy-generation processes and mass flows into agriculture. In principle, feedback loops exist between these elements (e.g. emissions from intentional use leading to contamination of sludge), but due to the limitations of this project these links could not be quantified in the model.

The basic approach behind the study was first to make an inventory of current mass flows and stocks. The future mass flows and stocks can be estimated on the basis of trends in uses. Here we chose to 2030 (one generation) as the time horizon. Such a time horizon is quite usual in SFA. Of course, the drawback is that the projections become more uncertain than if one had chosen a year closer in time, e.g. 2010. Yet, since an important part of the discussion related to lead concerns long-term aspects such as fear of accumulation of lead in certain societal flows and soil, for example, we felt such a longer-term perspective was appropriate for this study. The results of such (uncertain) projects for 2030 have to be interpreted in the following way. It gives a *possible* future emission situation under assumptions that *potentially* might be true, but that could well be influenced by policies, or external, unexpected developments. We will take this perspective into account in our discussion and evaluation of the results.

As a second step, the current emissions are inventoried. For direct emissions sources they are in general based on EMEP (a co-operative programme for monitoring and evaluation of the long-range transmission of air pollutants in Europe). An emission factor is derived for production processes, MSWIs and energy generation processes, among others, by dividing the known emissions and mass flow. For some other (diffuse) sources, like lead applications and landfills, the emission can be calculated as a function of an emission factor and the lead stock. In several cases, it is likely that emission factors and lead flows in 2030 will differ from the situation in 2000. We adapted the emission factor where the literature showed there were clear indications this would happen. The 2030 emission estimates are basically calculated by multiplying the new lead flows for 2030 with the existing emission factors from 1998 for some product groups, or where appropriate new emission factors. Below, these elements are discussed in some more detail.

3.3.2 *Calculation of mass flows and stocks*

In this section a general description is given of the methods and assumptions used to estimate the current mass flows and trends. All assumptions and calculations will be given in more detail in the sections on the different product groups.

Consumption and trends

Consumption in a specific year is defined as the flow of newly made products to the consumers in that year. Unless otherwise indicated it is assumed that the production figures of the lead industry in the EU15 (table 3.4) equal the consumption of these products in the EU15. This is of course not true, since imports and exports are neglected, but it will be an adequate approach in many cases. In specific cases, where it is clear that imports or exports are important, separate estimates are made.

Past trends are extrapolated to the future. As a consequence, the consumption of some lead applications will increase, such as batteries and lead sheet. Some applications will remain at a constant consumption level, like extruded products (radiation shielding materials, chemical applications), pigments (glazes, ceramics, paints), ammunition, alloys and miscellaneous; and finally, some will decrease and fade away, like water pipes, lead stabiliser in PVC, cable sheathing, petrol additives and probably in the long

run, cathode ray tubes due to replacement by flat screen displays. For some applications, like the use of lead in solder, the future is not clear, and an informed assumption has to be made.

Disposal of products: accumulation in stocks on landfill sites and in the environment

Estimates of lead waste flows (recycling, landfill and incineration) are based on the inflow of the products into the stock in the past. For this purpose a lifetime is defined for each application. In order to estimate present waste flows, information on past consumption of lead in different applications is required. For example, the current waste flow related to an application with an average lifetime of 10 years is estimated as equal to the use of that product 10 years ago. We base past uses (inflows) on the trend given for the OECD countries (table 3.3). The percentage of recycling also has to be defined for each product group; we made product-specific estimates that will be discussed in sections 3.4-3.12. For products going to the final waste treatment it is assumed that 80% is landfilled and 20% is incinerated, which was the situation in 2000. It is especially difficult to estimate what the future situation in the EU will be like on this aspect. Several countries will put more emphasis on incineration (like the Netherlands, Denmark, Germany and Belgium), but in other countries landfill is likely to remain an important ultimate destination for (final) waste¹¹. During our literature search we have not encountered well-founded, EU-wide projections in this field. To account for the likely shifts to incineration in some EU member states, we have taken an amount of 70% landfilled and 30% incinerated in 2030, with a proportional increase between 2000 and 2030. Table 3.5 summarises the assumed lifetimes of lead products and recycling rates. In general, we have taken quite optimistic recycling rates, assuming that appropriate waste management systems are in place¹². For each product group, the disposal of the product, the flows of lead to the final waste treatment and the accumulation of lead in landfill sites is described in more detail in sections 3.4 - 3.12.

Note that the stock of lead in landfill sites is only based on consumption of lead products from 1970 onwards. In fact, these stocks may be much larger due to lead landfilled before 1970. Stock estimates therefore are expected to be lower than the real situation, and therefore the emissions from landfill sites will probably be underestimated as well.

¹¹ Of course the EU Landfill Directive and various measures aimed at supporting recycling will ensure that the amount of waste landfilled without any pre-treatment will decrease considerably from what it is now. However, since there is no incentive for steering waste to incineration it is unlikely that countries that currently rely on relatively inexpensive landfill (which will stay relatively inexpensive even after adapting landfills to the demands of the EU Landfill Directive), and who have enough space, will start incinerating waste in high volumes.

¹² For instance, even in countries with sophisticated collection systems like the Netherlands, experience shows that the high values of 95 % for LCI batteries and 90 % for consumer batteries are difficult to meet (and sometimes are still not met). Our data here are much higher than given by the LDA in a report valid for the mid-1980s (Thornton, 2001: 96). Though with the advent of the Directive on Waste from Electrical and Electronic Equipment (WEEE) recycling targets for electronics will be set in future, it seems to us very unlikely that recycling will concentrate on the very small fraction of solder in such products. Hence the assumption of 0% recycling; lead will most probably end up in a residual fraction that is incinerated or landfilled after all. The recycling of cathode ray tubes and glass has been arbitrarily set at 50%. The PVC industry has committed itself to recycling roughly 50 % of the main PVC product categories, a value used here but which still has to prove itself.

Table 3.5 Lifetime of products and the relative fate of the discarded products

	Life time of product	Remains in environment	To waste treatment (present situation)		
	Years	%	%	Disposal Landfill % (2000/2030)	Incineration % (2000 /2030)
Batteries					
SLI Batteries (car batteries)	5		95	4-3.5	1-1.5
Motive power battery (elec. vehicles)	5		95	4-3.5	1-1.5
Standby energy supply in industry	10		100		
Consumer batteries	5		90	8-7	2-3
Cable sheathing Rolled and extruded products	25	50	50		
Lead sheets	50		90	8-7	2-3
Pipes	50		90	8-7	2-3
Other extr. products	15		50	40-35	10-15
Miscellaneous Ammunition Petrol additives Pigments and other compounds	15		50	40-35	10-15
Cathode ray tubes	15		50	40-35	10-15
Other pigments in glass	15		0	80-70	20-30
PVC cables, pipes, profiles	50	50	25	20-17.5	5-7.5
Other pigments and compounds	15	0	0	80-70	20-30
Alloys					
Solder in electronics	15		0	80-70	20-30
Other solder	15		100		
Other alloys	15		0	80-70	20-30

3.3.3 Calculation of emissions and emission factors

Emissions from point sources and traffic

As a basis for calculating emission factors, we used emissions of lead to air taken from EMEP (co-operative programme for monitoring and evaluation of the long range transmission of air pollutants in Europe). Figure 3.1 shows the emission of lead in the EU15 over the period 1990-1998. The sharp drop in emissions to air in this period is mainly caused by the introduction of lead-free petrol. Table 3.6 shows the contribution of the different sectors to the emissions to air in 1998. In the table, the individual emission categories are rearranged into groups relevant for this project (see Figure 3.2 and section 3.2.4).

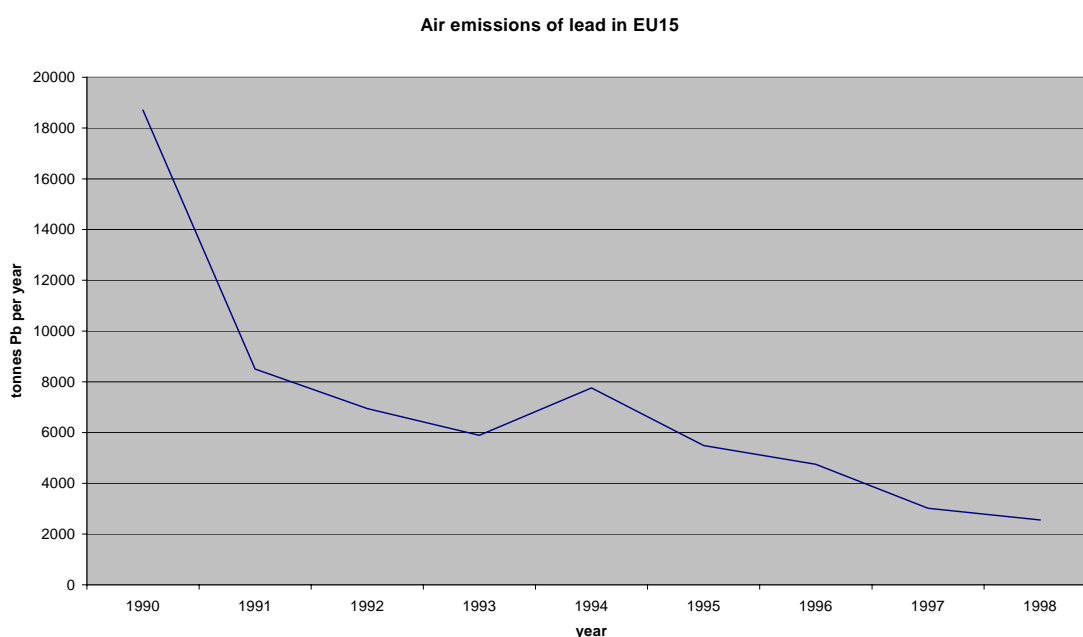


Figure 3.2 Emissions of lead to air in the EU15 (tonnes / year; based on EMEP)

Table 3.6 Sector emissions of lead to air in the EU15 in the year 1998 (based on EMEP).

Economic sector	Air emission (tonnes)	%
Energy generation	480.86	
- Combustion in energy and transformation industries	19.97	1
- Combustion in manufacturing industry	440.91	17
- Extraction & distribution of fossil fuels and geothermal energy	0.00	0
- Non-industrial combustion plants	19.98	1
Production processes	358.14	14
Road transport (petrol additives)	1559.84	61
Waste treatment (MSWIs)	124.19	5
Other (not included)	35.23	
- Other mobile sources and machinery	34.00	1
- Other sources and sinks	0.02	0
- Solvent and other product use	1.10	0
- Agriculture	0.11	0
	2558.25	

The EMEP data are the most recent source available for the EU. But it has to be noted that there are other, somewhat older, sources that give much higher emission values. For instance, in a rather extensive forecast even for 2010, TNO suggests total emissions of 5400 tpa lead in Western Europe (2400 tpa excluding road transport), whereas Pacyna gives values that are at least a factor 2 higher (TNO and Pacyna as cited in Thornton, 2001:60-63)¹³. It has to be noted as well that many sources who report a sharp decline of lead emissions in Europe over the last decade and predict further sharp declines in future, do so mainly in relation to the sharp decline in the use of leaded petrol. The cited TNO study indicates that in industrial sectors like public power, industrial combustion, production processes, and waste treatment, lead emissions will not or will only slightly decline between 1990 and 2010.

Combustion emissions of lead to air are due to the contamination of fossil fuels like coal and oil. For the future situation, for the sake of simplicity they have been kept constant. The total emissions given in EMEP for 1998 correspond fairly well with an estimate by TNO for 1990 and 2010. TNO's projection for 2010 is in the same range as the value used here for 2000, and just 5-10 % lower than TNO's value for 1990 (TNO as cited in Thornton, 2001:60). Hence, this literature confirms our assumption.

Process emissions to air from industry are attributed to the different lead products based on the relative consumption of lead in the lead industry (see table 3.4). We introduced a minor inconsistency here, since a small part of these process emissions are related to the steel industry rather than the lead industry. EMEP gives no separate values for these two industrial sectors. However, an analysis by Pacyna (cited in Thornton, 2001: 62) suggests that in 2000 some 25 % of the process emissions of lead from industry come from steel production, and 75 % from the non-ferrous industry. Given the inherent uncertainties in emission factors, we felt that a further specification of the EMEP data on this basis would have little added value¹⁴. With a total emission of 358 tpa in 1998, and a new lead use of 1574 kt/a, this gives an emission per tonne of lead of 0.227 kg (or 0.02 %). This number has been used to estimate future emissions on the basis of future mass flows. In various literature sources one can find arguments that the emission factor will stay the same in future, but also that there might be a decline. A report by TNO (cited in Thornton, 2001:60) into long-range transboundary air pollution backs our choice not to assume major reductions of this emission factor. TNO estimated that the emissions between 1990 and 2010 in Western Europe from production processes would diminish by just some 6 %, which is marginal given the uncertainties in the emission factor used. For the US, an emission factor of 0.03 % for lead recycling has been reported (Lave et al, 1995). This is well in line with our 0.02 %. However, for this specific process of lead recycling, more recent data suggest an emission factor that might be a factor 5 lower (Socolow and Thomas, 1997). Yet, at the same time it has to be acknowledged that lead recycling is a process with relatively low emissions in comparison to, for example, primary lead production. All in all, we feel that it is probably the safest option to keep on using the emission factor of 0.02 % for the future as well, but to acknowledge that this might give an overestimation should more

¹³ The TNO study was published in 1998. The Pacyna study was published in 1996. It has to be noted that the Pacyna study does not cover all of Western Europe, but does include some Eastern European countries, which could be the reason for the relatively high emission values. At the same time, Pacyna's estimated emission values from the lead smelters in the EU15 alone amount to over 2100 tpa., or almost the total amount we used on the basis of EMEP for all sectors together.

¹⁴ An additional argument is the fact that the EMEP data differ highly from the Pacyna data, and we would face the danger that using the Pacyna data to split up the EMEP data would simply introduce another inconsistency.

comprehensive emission reduction measures be implemented, for example, as predicted in the cited TNO study.¹⁵

Emissions from road transport are mainly caused by lead additives in petrol. Lead additives in petrol are still the largest source of emissions to air (60%). However, the sharp decline in the consumption of leaded petrol over the past 10 years is assumed to mean that in the year 2010 leaded petrol will no longer be used in the EU15. We refer further to the specific section on lead in petrol (3.10).

The emissions of lead from waste incinerators are attributed to the different products based on the estimated streams of products to the final waste treatment. Estimates for the waste stream are based on assumptions of the lifetime of products and recycling percentages given in table 3.5. It is also assumed that the present waste treatment and recycling percentages will still be valid in 2030 (see also paragraphs 3.4-3.12). As will be shown in these sections, the total lead input into Municipal Solid Waste Incinerators (MSWIs) in 1998 is estimated at 71 ktpa. With an emission of 124 tpa, this results in an emission factor of 0.175 kg per tonne of lead input in MSWIs. The emission factor we calculated in this way is equal to or lower than the emission factor usually applied by LCA experts for the most modern MSWIs that comply with the most stringent legislation in the Netherlands and Germany¹⁶. Hence, it is likely that for the present situation we underestimated the emissions from MSWIs since MSWIs do not yet meet such standards throughout Europe¹⁷. There is certainly no objective reason to assume a lower emission factor for the future situation than the 0.175 kg per tonne we used here¹⁸. In reality, the relationship between lead present in the different applications and the emission to air in incinerators will not be linear. This means that some lead applications, like pigments and stabilisers, will contribute proportionately more to the emission to air than metallic applications like batteries and alloys. However, in this study the air emissions are attributed to the different products in proportion to the amount of products in the waste stream. Physio-chemical properties are not taken into account. An exception is made for lead in glass (cathode ray tubes, crystal, optical

¹⁵ An emission factor of 227 g/tonne of lead implies a loss of just 0.02 % to air. The processes in the lead industry are of course not totally comparable with Municipal Solid Waste Incineration (MSWI), but this is already an emission factor which is lower than for MSWIs equipped with flue gas cleaning complying with the most stringent legal standards in the Netherlands and Germany (cf. AOO, 1995). On our interim report the LDA commented that BAT could reduce the emissions to 50 g/tonne of lead processed in the lead industry, and that at least in the UK a relatively high part of the reported lead emissions actually come from the steel industry. Though we accept the suggestion that BAT might lead to emission reductions in future, we feel it is highly difficult to predict which emission factor can be used in the end. For instance, both TNO (1300 tpa) and Pacyna (over 3000 tpa) (as cited in Thornton, 2001) give much *higher* total emissions for 2000 from the lead industry than we used on the basis of EMEP. Had we used the former data, we would have started with at least a four-fold emission factor in the first place. Therefore, we decided to stick with the 227g/tonne as well for the future. We see this whole discussion as additional proof of the uncertainties of such emission factor estimates in general.

¹⁶ The Dutch Waste Management Council (AOO) applies an emission factor of about 0.07 % in its LCAs (e.g. AOO, 1995).

¹⁷ It might be that EMEP underestimates the current emissions from waste management. For instance, TNO (as cited in Thornton, 2001:60) suggests a total emission of 430 tpa in 1990 and 500 tpa in 2010).

¹⁸ The fact that in some countries lead emissions from incinerators have fallen over 90 % since 1980/1985 due to implementation of measures like electrostatic filters does not mean that we can indefinitely extrapolate such reductions to the future. The simple fact is that in order to comply with the EU's incineration directive, an incinerator has to be equipped with an electrofilter and certain washing steps, and such a configuration leads typically to the emission factor we used. We can safely assume that further reductions will not be strived for in the next 10-20 years at least, given the tremendous technical and financial demands already posed by the EU's incineration directive.

glass), which is considered stable and so does not contribute to the emissions to air in incinerators.

Emissions from diffuse sources

Several other applications lead to emissions due to corrosion and slow leaching of lead from various applications and stocks of lead. Of course, the use of somewhat different assumptions can change all these emission factors, and we have no illusion that any of the values presented here will have a lower uncertainty than 100 %. They are in general based on a defensible logic, but it is strongly advised that additional research is done to lower the uncertainty ranges for the most crucial lead applications. In sum, we applied the following emission factors:

- ***Corrosion of lead from building materials***

Many authors use a factor of 5 g/m².yr for the corrosion of lead (Bentum, Verstappen & Wagemaker, 1996; Coppoolse et al., 1993, Annema et al., 1993). For the Netherlands, several studies assume that roughly 25% of the total amount of lead sheet in buildings is exposed to the atmosphere and rainwater leading to a total yearly emission of lead in the Netherlands of 120 – 150 tonnes (Bentum, Verstappen & Wagemaker, 1996; Coppoolse et al., 1993, Annema et al., 1993). Given the annual new use of lead sheet for building applications (see table 3.10) and the number of newly built houses, an average use of lead per house was calculated for the Netherlands. Multiplying this by the total number of houses in the Netherlands, a total stock of lead sheet in building applications of 1448 kilotonnes was estimated. With annual emissions from building applications of between 120 and 150 tpa, this implies an emission factor between 0.008 and 0.01 % per year from lead in buildings. We decided to use the lower value of 0.008 %, mainly because the Lead Development Association (LDA) argued that the emissions calculated by the above-mentioned authors may be slightly overestimated¹⁹. It is unclear if the run-off will end up in soil or in a sewage system. We have assumed that 50 % will emit directly to soil and 50 % will be trapped by a sewage system. Assuming that all lead is removed in a waste water treatment plant, the remainder will end up in sewage sludge. Over half of the lead sheet in the EU is applied in the UK, where some 50 % of the sewage sludge is applied on land as fertiliser. If we apply this value for the whole of Europe²⁰, this implies an additional emission of 25 % to soil, or with the direct emission a total of 75% of the 0.008% to soil. The net effect is an emission of 0.006% to soil.

- ***Corrosion of lead from water pipes***

In literature no estimates are found for the total amount of water pipes made of lead in the EU15 or any of its member states. Nor is any corrosion factor available. However, for the Netherlands emissions of lead to the sewer were estimated based on measurements in residential areas with water pipes made of lead (Bentum,

¹⁹ The emission figures from the cited literature sources can be influenced by making different assumptions about the percentage of lead exposed to the atmosphere and the thickness of lead sheet. The LDA criticised the emission figures in literature since they might be based on thinner lead sheet than is sold on average on the market. Obviously, with a fixed emission factor of 5 g/m²/year, the thicker the sheet, the lower the emission as a percentage of the stock. A point that the LDA also brought into the discussion is that the corrosion might decline in time due to the formation of metal oxides and sulphides. For some other metals such processes seem to reduce the emission considerably. However, no practical research has been done yet to make any quantification of this (potential) influence and it must be stressed that for lead this argument is therefore entirely hypothetical at this stage, and can only be used if hard proof is delivered. Hence, we used the 5 g/m²/year which is generally mentioned in literature as a starting point, and would suggest that practical research into the actual emission of lead during the lifetime is conducted.

²⁰ In the Netherlands, another important user of lead sheet, sewage sludge is mainly landfilled or incinerated. We have assumed that in other EU countries landfill of sewage sludge is somewhat more dominant.

Verstappen & Wagemaker, 1996.) These emissions were converted to emission factors by dividing the emissions by the total population (table 3.7). Note that the emission factors represent the Dutch situation. In the Netherlands it was assumed that in 1990 that 15% of the houses still contained water pipes made of lead. For the European Union this percentage might very well be higher.

Table 3.7 Emission factors of lead from water pipes in the Netherlands (mg/capita per year).

	1980 ¹	1985 ²	1990 ³	1993 ³
Background concentration from water station		40		
Corrosion from water pipes	1069	660	1014	906

1 Comber & Gunn, 1994, Foundation for Water Research (FWR), Diffuse sources of heavy metals to sewers;

2 Coppoolse et al, 1993; SPEED document; Heavy metals in surface water, sources and measures;

3 Bentum, Verstappen & Wagemaker, 1996; RIZA report; Watersysteemverkenningen, Een analyse van de problematiek in aquatisch milieu, WSV- doelgroepstudie bouwmaterialen.

- **Corrosion of ammunition**

Little is known about the corrosion of lead from ammunition to the soil. However, it is generally agreed that the corrosion will increase in acid environments, which is the case in the humus layer of the soil. The yearly corrosion of lead from ammunition is assumed to be 1% of lead from the stock of ammunition in the environment (Stoop & Rennen, 1990). We have assumed this to be an emission to soil. One could argue of course that the lead that is corroded might not be immobilised in soil again by the formation of lead oxides, etc.

- **Corrosion of other products**

For other products like alloys and cable sheathing, no corrosion factors are given in literature. As a rough guess, for all metallic applications of lead a corrosion factor is assumed of 0.008% per year, which is the same as the corrosion factor for lead sheet in buildings.

- **Leakage from landfill**

Little is known about the leakage of substances from landfill sites. The leakage depends on many parameters, like the stock of lead (composition and amount) in the waste on the site, the weather and hydrological conditions and the way the site is controlled (covered or not covered, treatment of waste-water from the site, etc). As a rough guess, for lead compounds a leakage percentage is used of 0.23% per year (Roorda, 1998). At first glance this might seem a relatively high value, but we have applied it for 'free' lead only. For most lead inflows to landfill, we have additionally taken into account a corrosion factor that indicates which fraction of the lead in landfill might be mobilised. The corrosion factor multiplied by 0.23% is the actual emission factor for this lead product group, and in general is much lower than the 0.23%. For the leakage of lead metallic applications, a corrosion factor of 0.008% a year has been used. The leakage of lead compounds from ceramics also depends on a wear factor, set at 0.1% per year. For some applications the lead is considered to be stable, meaning no leakage occurs from lead compounds in glass (crystal, cathode ray tubes) and lead in PVC. All the ash and slag from waste incinerators is assumed to be dumped in the landfill sites. The lead stabiliser of the *incinerated* PVC (20% in 2000 and 30% in 2030) is considered to be free, and leakage is assumed

to be 0.23% per year. It should be noted that the leakage of lead from landfill sites is based on many assumptions, with a high degree of uncertainty in all parameters. Only the stock in landfill sites due to the accumulation of discarded products since 1970 is estimated. The estimated amount and composition of the stock in landfill is based on a model using the lifetime of products and recycling rates as uncertain parameters. The wear and corrosion factors used for different products also introduce uncertainty. And finally, the leakage factor of lead compounds from the site to the soil is a rough guess. The estimated emission from landfill might easily be a factor 10 higher or lower than the emission presented in this study²¹.

3.3.4 *Conclusion*

The above approach describes in outline how the current and future mass flows, stocks and emissions have been calculated. The next sections describe in more detail all calculations for each product group. For each product group, the emissions upstream (lead production) and downstream (lead waste management) have been included. Section 3.12 gives a final overview of emissions.

3.4 **Batteries**

3.4.1 *Uses of lead in batteries*

By far the largest application of lead is its use in batteries. In the EU15 in 1998, about 919 kilotonnes of lead were used in the manufacture of batteries. That is nearly 60% of the total lead consumption. In the West, the consumption for batteries in 1998 was 3534 kilotonnes, nearly 75% of the total lead consumption (ILZSG, 2000). Lead in batteries is applied in a variety of products:

- SLI batteries for vehicles with petrol and diesel engines,
- traction batteries to power electric vehicles,
- standby and storage batteries to provide emergency electricity in the event of power failure, and
- consumer batteries in small electrical products.

3.4.2 *Trends in uses of batteries*

Figure 3.3 shows the consumption of lead in batteries in the EU15. The figures for the European Union in 1970-1990 are based on the trend of the consumption of lead in batteries in the OECD countries (OECD, 1993). Consumption of lead in batteries has increased by a rate of 3% per year. For the future consumption in the EU15 it is assumed that this annual increase will continue.

²¹ Indeed, differences in assumptions for emission from landfill vary dramatically, as can be shown with some examples from the LCA-community. Some experts tend to use a time horizon of some 100 years, and on the basis of certain leachate tests they estimate the losses of metals from a well-managed landfill in this period to be 0.1-1 % at most (compare AOO, 1995), or lower (Finnveden, 1995). However, other experts use much longer time horizons, and argue that instead of a classical leachate test, a total availability leachate test should be used as a proxy to estimate the total losses via landfill. Such a test uses a relatively high pH, and losses of several dozen percent for metals are not uncommon (Hellweg, 2001). This discussion shows clearly that science has not yet reached consensus on the question of whether in the long run landfills will contain all hazardous materials, or if they will eventually release most of their hazardous contents into the environment.

Table 3.8 shows the consumption of lead for the manufacture of batteries for different countries in the EU15. In this study it is assumed that the consumption of batteries in the EU15 equals the production of batteries.

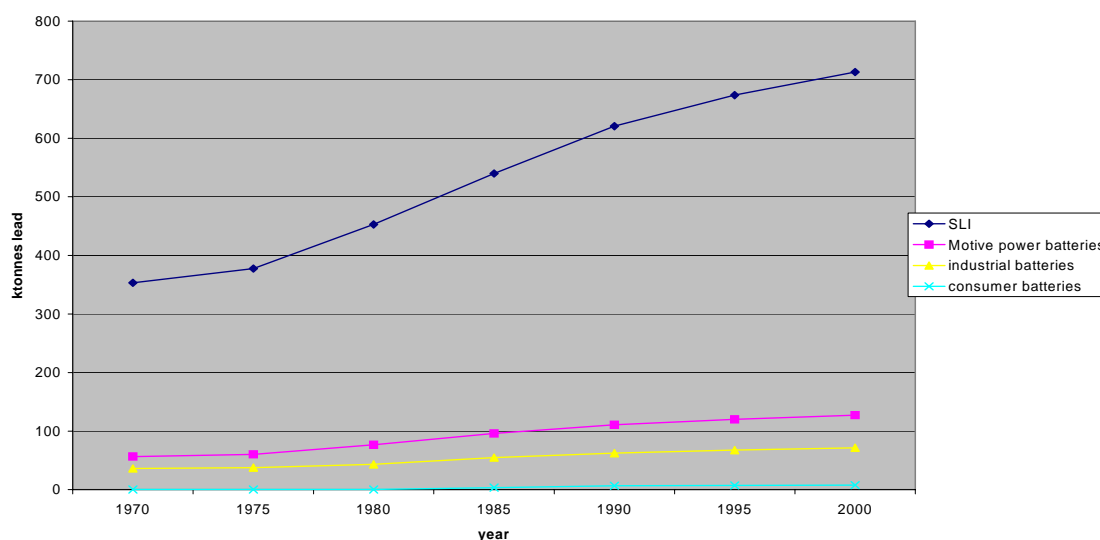


Figure 3.3 The consumption of lead in batteries in the EU15 in 1970-2000.

Table 3.8 The consumption of lead (kilotonnes) for the manufacture of batteries in the EU15 in 1993-1998. (ILZSG, 2000)

	1993	1994	1995	1996	1997	1998
Austria	41.50	42.40	43.50	39.00	42.00	46.00
Belgium*	21.40	19.80	19.80	14.90	15.50	14.00
Finland	3.00	3.50	3.50	3.50	3.50	3.00
France	155.80	170.40	189.90	190.90	188.20	199.00
Germany	204.10	215.70	206.90	192.70	178.40	181.70
Italy	109.00	132.00	149.40	158.40	165.60	174.70
Netherlands*	18.00	26.00	29.00	25.00	25.00	24.00
Scandinavia	20.00	27.00	30.00	35.00	32.00	30.00
Spain	67.30	78.00	88.00	103.00	127.00	141.00
United Kingdom	102.60	107.60	108.80	107.20	110.80	106.00
TOTAL	742.70	822.40	868.80	869.60	888.00	919.40

* Includes Cable Sheathing from 1996 onwards

3.4.3 Emission factors for lead in batteries

In the literature no emissions during the use of lead-acid batteries are reported. The most important leakage to the environment will probably take place in the disposal phase of the lead-acid batteries. Although the recycling percentage of lead-acid batteries is very high, the batteries still make a large contribution to the lead load of municipal solid waste due to the large consumption of lead in batteries. The collection rate of

batteries is high in most of Western Europe. More recently, an EU batteries directive obliges member states to ensure a high rate of battery collection. Different countries have approached this in various ways. Stated collection targets range from 100% in France and 99.9% in Denmark, to 75% in Portugal. Some member states like the UK have not set up any formal system, as a high (estimated at 90%) recycling rate is already achieved by existing methods using scrap dealers (Scoullos, Vonkeman & Thorton, 2001).

In this study it is assumed that, in general, batteries are recycled in at a rate of 95%; for industrial applications 100 % has been taken, and for consumer applications 90 %. For 2000 and earlier, a split between landfill and incineration has been used of 80-20%.;for 2030 this split has been set at 70-30%, with a proportional increase in between. The remaining batteries will probably end up in part in the municipal solid waste, leading to air emissions during waste incineration, leakage from landfill (solid waste, and slag and ash) and leakage from building materials (fly ash in concrete, slag in road construction, etc). The lifetime of batteries is four to five years. For industrial applications the lifetime is about 10 years (Scoullos, Vonkeman & Thorton, 2001). Table 3.9 shows the estimated emissions of lead during the production and waste disposal of batteries in the EU15; detailed calculations specified by type of batteries are given in Annex 1.1. The emission during production is based on the air emissions from industry of 358 tonnes in 1998 (EMEP, 2001) attributed to batteries proportional to the consumption of lead in products in the EU15 (see table 3.4). The air emission of waste incinerators is based on the emission of 124 tonnes in 1998 (EMEP, 2001) attributed to the different products proportional to the estimated waste streams from discarded products (see table 3.9).

Table 3.9 Emissions of lead during production, consumption and disposal of batteries in the EU15 (in kilotonnes Pb unless stated otherwise)

Year	Production	Consumption			Waste treatment							
		Air emission	Inflow of Pb	Outflow of Pb	Emission	Remains in environment	Recycling	Landfill	Incineration	Emission from incineration to air	Stock on landfill	Emission from landfill to soil
						90-95%	8/7% or 4/3.5%	2/3% or 1/1.5%				0.23 % leakage 0.008% corrosion
1965												
1970	p.m.	446	0	0	0	0	0	0	p.m.	0	0.000	
1975	p.m.	476	410	0	0	390	16	4	p.m.	103	0.002	
1980	p.m.	573	474	0	0	453	18	4	p.m.	212	0.004	
1985	p.m.	694	568	0	0	541	21	5	p.m.	345	0.006	
1990	p.m.	800	682	0	0	650	26	6	p.m.	505	0.009	
1995	p.m.	869	793	0	0	755	30	7	p.m.	692	0.013	
2000	0.21	919	863	0	0	823	32	8	0.03	894	0.016	
2005	0.21	947	915	0	0	873	33	9	0.03	1108	0.020	
2010	0.22	975	945	0	0	901	34	10	0.03	1328	0.024	
2015	0.22	1005	973	0	0	928	34	11	0.04	1555	0.029	
2020	0.23	1035	1002	0	0	956	34	13	0.04	1789	0.033	
2025	0.24	1066	1032	0	0	984	34	14	0.05	2029	0.037	
2030	0.25	1098	1063	0	0	1014	35	15	0.05	2277	0.042	

3.5 Rolled and extruded products

3.5.1 *Uses of lead in rolled and extruded products*

The second-largest application of lead in the EU15 is the use of lead in rolled and extruded products. In 1998 in the EU15 the consumption of lead for rolled and extruded products was about 233 kilotonnes (15% of the total lead consumption in the EU15). In the Western world the consumption in 1998 was 290 kilotonnes (6% of total consumption) (ILZSG, 2000). Rolled and extruded applications represent a variety of products, but by far the largest application is the use of lead sheets in buildings (80% of the rolled and extruded products). Other applications are radiation shielding materials (10% of the rolled and extruded products), sound-proofing materials (3% of the rolled and extruded products) and chemical applications (5% of the rolled and extruded products). In the past, lead pipes were also used to transport drinking water. However new lead pipe has not been used for domestic water supplies for over 30 years in many countries. In some countries, though, a considerable amount of lead piping is still in service, particularly connecting the street mains supply to individual houses.

3.5.2 *Trends in uses of rolled and extruded products*

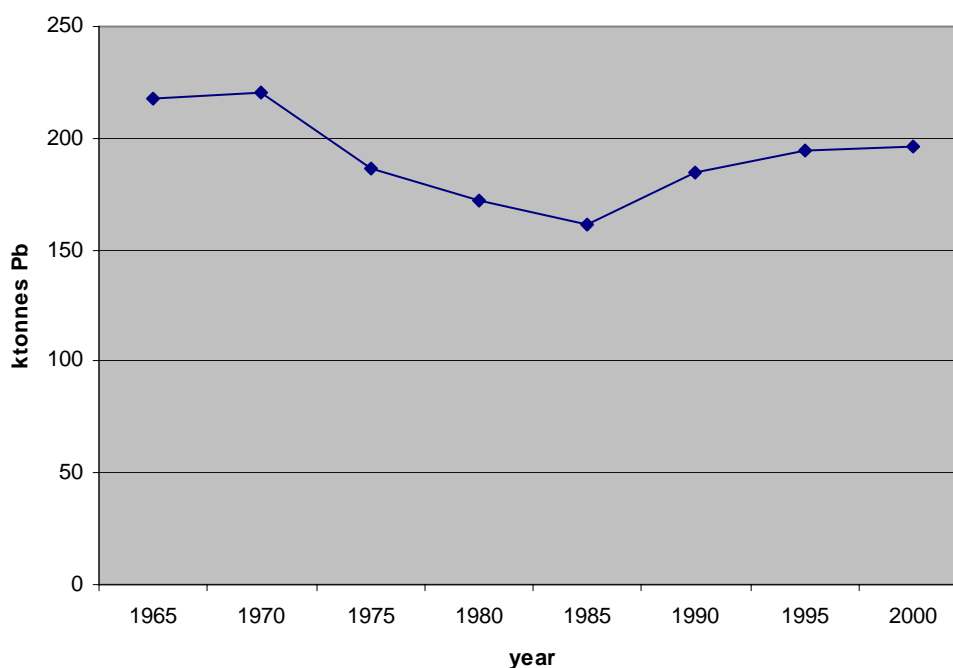
Lead sheets

Figure 3.4 shows the consumption of lead in lead sheet in the EU15. Between 1990 and 2000, ILSZG data formed the basis (see table 3.10). Figures for the EU15 between 1995-1990 were based on the use according to ILSZG in 1990 and the trend in consumption of lead in lead sheet in the OECD countries (OECD, 1993). For the future it is assumed that the demand for lead sheet in the EU15 will continue to grow due to the growing population and the resulting housing demand. In relation to the population growth, the stock of lead in buildings is assumed to grow by 0.3% annually. In table 3.10 the consumption of lead sheet in the European Union is given for the period 1993-1997; table 3.11 gives a more comprehensive view over the period 1965-2030.

Water pipes and other rolled and extruded products of lead

During the period 1970-1990 most of the applications of rolled and extruded products, like pipes, chemical applications, wine bottle capsules and collapsible tubes declined (see table 3.3) (OECD, 1993). The use of lead in sound-proofing materials and radiation shielding has remained more or less constant. For the future it is assumed that the annual consumption of lead in other rolled and extruded products will remain at the same level: about 44 kilotonnes lead per year. In order to avoid any confusion: the ILZSG data given in table 3.10 are for lead sheet only. Including the 44 ktpa for these other rolled and extruded products, our total estimate for 1997 for all rolled and extruded products together is hence some 240 ktpa.

Figure 3.4 The consumption of lead in lead sheets in the EU15 in 1970-2000 (in ktpa)

Table 3.10 Consumption of lead sheet in the EU15 (kilotonnes lead) (ILZSG, 1999)^{a 22}

	1991	1992	1993	1994	1995	1996	1997
Belgium	8.7	11.0	14.4	12.7	11.1	16.6	21.2
France	16.0	14.6	15.7	15.3	15.2	15.4	15.8
Germany	24.1	21.7	19.6	26.1	21.0	18.3	19.0
Greece ^b	2.4	2.4	2.4	2.4	2.4	2.4	2.4
Ireland	14.6	14.6	7.0	5.6	5.4	8.8	9.8
Italy	15.0	13.1	11.6	10.4	11.1	10.3	10.7
Netherlands	22.6	22.6	23.4	22.1	21.0	21.9	21.8
Spain ¹	5.6	3.9	3.8	4.0	4.0	4.0	4.0
UK	75.9	74.6	76.5	75.1	92.3	84.1	89.5
TOTAL	184.9	178.3	174.2	173.7	183.7	181.8	194.2

a: ILSZG (1999) gives data for all rolled and extruded products. Lead sheet is about 80 % of this volume. Hence, the consumption of lead sheet in building was estimated by multiplying the ILSZG data by 0.8

b: production figures, not consumption figures of lead sheet

²² It should be noted that the ILZSG data are roughly 20% higher than the data generated by the European Lead Sheet Industry Association. LDA suggested that ELSIA data only refer to milled lead sheet, and do not include sheet produced via another route. As indicated in the introduction of this chapter, for reasons of consistency we decided to stick to the ILZSG data.

3.5.3 *Emission factors for rolled and extruded products*

Lead sheets

The use of lead sheet in the building sector leads to an emission of lead to the environment due to a slow corrosion of lead. In the literature, corrosion of lead of 5 g/m².year is usually used to estimate the emission of lead from lead sheets (Bentum, Verstappen & Wagemaker, 1996; Coppoolse et al, 1993, Annema et al, 1993). As indicated in section 3.3.2, this implies an emission factor of roughly 0.008 % of the lead stock per year; we have estimated that some 75% of this amount will end up as emission to soil.

In order to estimate the corrosion from lead sheet in the European Union, the stock of lead sheets in the present buildings (houses and utility buildings) should be estimated. Various methods can be used here. One approach is to use the annual application of lead sheet in the building sector, roughly some 200 ktpa since 1965, and to multiply this value by the average lifetime of lead sheet in the building sector. Here, 50 years seems a minimum, suggesting a lead stock of at least 50 times 200, or some 10,000 kilotonnes. With longer lifetimes, say 100 years, this value might even double. Another approach is to calculate the average amount of lead used per house and to multiply this value by the total number of houses in the EU15. Based on the present inflow (consumption) of lead sheet in the stock (see table 3.5) and the increase in the number of households in the European Union, the average amount of lead sheet in houses is estimated to be 90 kg/house (note that the amount of lead sheet per house differs between the different EU member states – this value is an average for the EU)²³. Based on the total number of households in the EU15, the stock of lead sheet in houses is estimated to be 10715 kilotonnes in 1991. Since we calculated the 90 kg/house by assuming the housing sector would use only half of the inflow given in table 3.10, the remainder being applied in utility buildings, we have to correct by multiplying by a factor of 2 to include the utility sector. This implies a total stock of lead sheet in buildings in the EU15 of 21430 kilotonnes in 1991. This might be an upper limit, since the 90 kg was calculated including countries with a high lead use in buildings (the UK, the Netherlands) and excluding some countries which probably have relatively low use. At the same time, the 10,000 kilotonnes calculated on the basis of inflow and lifetime seems to be a lower limit. Taking the average of these two data, we used a stock in the EU15 in building applications of 15,000 kilotonnes in 1990.

The changes in stock and emissions from stock from 1990 to 2030 (table 3.11) can now be estimated using the following assumptions:

- the stock in 1990 is set at 15,000 kilotonnes;
- the change in the stock in the following year is the difference between new use/inflow and waste/outflow²⁴;

²³ The 90 kg per dwelling was derived by dividing the inflow to the building sector given in table 3.10 by the number of authorised dwelling permits in a country in a year as given by Eurostat. In calculating the 90 kg, we assumed that only half the amount of lead given in table 3.10 is used for dwellings, the other half going to utility buildings. This explains the correction factor of 2 in the main text in calculating the total stock. We included the following countries to calculate the 90 kg: Belgium, the Netherlands, the UK, Ireland, Greece, Spain, Germany, France, Italy, i.e. countries for which we had specific use data.

²⁴ Table 3.11 is actually presented in 'blocks' of 5 years; in that case we simply calculated the difference in inflow/outflow over five years assuming that the inflow/outflow data for the base year were valid for the block of 5 years. As indicated, values for before 1970 were extrapolated making use of OECD data and data after 1997 were estimated as a 0.3 % growth in use per year in relation to population growth.

- given the assumed lifetime of 50 years, the outflow is assumed to be equal to the inflow of 50 years earlier. Since we have no inflow data from before 1965, we assumed this to be 200 ktpa;
- the annual corrosion of lead from stock is $8E-5$ kg Pb per kg lead sheet, of which 75% ends up as an emission to soil.

Though there is no formal system in place, most metals are stripped from old buildings before demolition and sold on to appropriate scrap merchants. The recovery rate is not known but estimated to be very high (Scoullos, Vonkeman & Thorton, 2001)²⁵. In this study it is assumed that 90% of the lead sheet is collected for recycling. The remaining 10% is dumped in landfill sites (80%) or is incinerated (20%). For 2030, we assumed a split of 70-30% between landfill and incineration, with a proportional increase in between. The lifetime of lead sheet is assumed to be about 50 years.

Table 3.11 shows the estimated emissions of lead during production, use and waste disposal of lead sheet in the EU15. The emission during production is based on air emissions from industry of 358 tonnes in 1998 (EMEP, 2001) attributed to lead sheet proportional to the consumption of lead in products in the EU15 (see 3.4). The air emissions of waste incinerators is based on the emission of 124 tonnes in 1998 (EMEP, 2001) attributed to the different products proportional to the estimated waste streams from discarded products (see table 3.7).

Lead in water pipes and other rolled and extruded products

Due to health concerns, the use of lead for drinking water systems has been gradually phased out in Europe. It is assumed that at present lead pipes are no longer being used in new buildings and are being replaced by copper pipes or plastic pipes (PE). However, a large amount of lead pipes are still present in 'old' buildings and distribution systems. Because the application is phased out it is very difficult to estimate this present old stock of lead pipes. However, as an approximation, the corrosion of lead from lead pipes for the EU15 is estimated using Dutch emission factors (table 3.12).

Table 3.12 Emission factors of lead from water pipes in the Netherlands (mg/capita per year).

	1980 ¹	1985 ²	1990 ³	1993 ³
Background concentration from water stations		40		
Corrosion from water pipes	1069	660	1014	906

¹ Comber & Gunn, 1994, Foundation for Water Research (FWR), Diffuse sources of heavy metals to sewers;

² Coppoolse et al, 1993; SPEED document; Heavy metals in surface water, sources and measures;

³ Bentum, Verstappen & Wagemaker, 1996; RIZA report; Watersysteemverkenningen, Een analyse van de problematiek in aquatisch milieu, WSV- doelgroepstudie bouwmaterialen.

²⁵ It should be noted that the source of Scoullos et al (2001) is oral information from the Lead Development Association. Since the assumed value is so crucial for the flow of lead to landfill, it is highly recommended that this number should be underpinned by scientific evidence, for example, through mass balance approaches.

Table 3.11 Emissions of lead during production, consumption and disposal of lead sheet in the EU15 (kilotonnes lead).

Year	Production	Consumption				Waste treatment						
	Emission to air	Inflow of Pb ¹	Outflow of Pb	Stock	Emission to soil	Remains in environment	Recycling	Landfill	Incineration	Emission from incineration to air	Stock on landfill	Emission from landfill to soil
In %					0.006 from stock	0	90	8 [2000] 7 [2030]	2 [2000] 3 [2000]			0.23 [leakage] 0.008 [corrosion]
1965		218										
1970		220				0	0	0	0		0	0.000
1975		186	200			0	180	16	4		100	0.002
1980		172	200			0	180	16	4		200	0.004
1985		161	200			0	180	16	4		300	0.006
1990		185	200	15000	0.9	0	180	16	4		400	0.007
1997		194	200	14888	0.9	0	180	16	4		500	0.009
2000	0.04	196	200	14868	0.9	0	180	16	4	0.01	600	0.011
2005	0.04	199	200	14843	0.9	0	180	16	4	0.02	700	0.013
2010	0.05	202	200	14834	0.9	0	180	15	5	0.02	800	0.015
2015	0.05	205	218	14839	0.9	0	196	16	5	0.02	909	0.017
2020	0.05	208	220	14770	0.9	0	198	16	6	0.02	1019	0.019
2025	0.05	211	186	14706	0.9	0	168	13	5	0.02	1112	0.020
2030	0.05	214	172	14825	0.9	0	155	12	5	0.02	1198	0.022

1 Figures for the years 1965-1990 are derived from 1990 figures for EU15 using the trend of rolled lead consumption in the OECD countries between 1965-1990 (OECD, 1993)

For the European Union (EU15), the corrosion of lead from water pipes (table 3.13) is estimated using the emission factors for 1990 and 1993 given by Bentum, Verstappen & Wagemaker (1996). The future emissions are based on the trend in emission reduction between 1990 and 1993. The trend is not based on population (growth) because the water pipes are assumed to be phased out. Note that the emission factors represent the Dutch situation. In the Netherlands it was assumed that in 1990, 15% of houses still contained water pipes made of lead. For the European Union this percentage might very well be higher.

Table 3.13 Corrosion of lead from water pipes in the EU15 (kilotonnes Pb)

	Corrosion
1997	0.29
2000	0.28
2005	0.27
2010	0.25
2015	0.24
2020	0.23
2025	0.23
2030	0.22

The total amount of lead pipes still present in the domestic water supply is not known. For this reason, it is not possible to estimate the amount of lead pipes which are discarded. However, it is likely that most of the pipes will not end up as final waste to be dumped or incinerated, but will be recycled.

Emissions of lead during the use of other extruded products, like radiation shielding and sound-proofing, are assumed to be negligible. So the largest leakage of lead will occur in the waste stage of these products. The recycling rates of these products are considered to be high and are assumed to be 90%. The remaining 10% is dumped in landfill sites (80%) or is incinerated (20%). For 2030, we assumed a split of 70-30% between landfill and incineration, with a proportional increase in between. The lifetime of the products is set at 15 years. In table 3.14 the emissions during production, use and waste disposal of the other rolled and extruded products is given. The emission during production is based on air emissions from industry of 358 tonnes in 1998 (EMEP, 2001) attributed to 'other extruded products' in proportion to the consumption of lead in products in the EU15 (see table 3.4). The air emissions of waste incinerators are based on the emission of 124 tonnes in 1998 (EMEP, 2001) attributed to the different products proportional to the estimated waste streams from discarded products (see table 3.7).

Table 3.14 Emissions (kilotonnes) of lead during production, consumption and disposal of lead in other rolled and extruded products in the EU15.

Year	Production Air emission	Consumption			Waste treatment						
		Inflow of Pb	Outflow of Pb	Emis- sion	Remains in enviro- nment	Recy- cling	Landfill	Inciner- ation	Emission from incin- eration to air	Stock on landfill	Emission from landfill to soil
						90%	8/7%	2/3%			0.23 %leakage 0.008% corrosion
1965											
1970		44									
1975		44									
1980		44									
1985		44	44	0	0	40	4	1		22	0.000
1990		44	44	0	0	40	4	1		44	0.001
1995		44	44	0	0	40	4	1		66	0.001
2000	0.01	44	44	0	0	40	4	1	0.00	89	0.002
2005	0.01	44	44	0	0	40	3	1	0.00	111	0.002
2010	0.01	44	44	0	0	40	3	1	0.00	133	0.002
2015	0.01	44	44	0	0	40	3	1	0.00	155	0.003
2020	0.01	44	44	0	0	40	3	1	0.00	177	0.003
2025	0.01	44	44	0	0	40	3	1	0.00	199	0.004
2030	0.01	44	44	0	0	40	3	1	0.00	222	0.004

3.6 Pigments and other compounds

3.6.1 Uses of lead as pigment and stabiliser

Another considerable use of lead is its application as a pigment and stabiliser of plastics (PVC). In 1998 the consumption of lead pigments or other compounds in the EU15 was about 193 kilotonnes (12% of the total lead consumption in the EU15). In the Western world the consumption was 437 kilotonnes (9% of the total lead consumption in the West). Lead is used as a compound in a wide range of products including:

- Cathode ray tubes: the tubes for television and computer monitors are constructed from a special leaded glass to protect them from harmful rays;
- Other glass applications, like lead crystal, radiation glass and optical glass;
- Stabiliser in PVC: lead compounds are added to PVC in order to prevent the plastics from degradation caused by heat and UV-light during manufacture and use. They are principally used in rigid PVC products, such as building profiles (window frames, guttering – 8% of the PVC applications), and pipes (33% of the PVC applications). Flexible PVC cables used for insulation of electric wires are made almost entirely from PVC (54% of the PVC applications). Foils and films (5% of the PVC applications) are also made of PVC stabilised with lead;
- Ceramic glazes, to provide a smooth, clear and scratch-free finish to a variety of ceramic products;

- Lead paints: although no longer used in paints for use by the public, lead is still an important constituent of special paints used for purposes such as corrosion protection of steel, road markings and warning signs.

The largest current application for lead compounds is in cathode ray tubes. The second most important application for compounds is the use of lead as stabiliser in PVC. Table 3.15 shows the relative use of lead compounds in the different products (derived from OECD figures for 1990 (OECD, 1993)).

Table 3.15 The relative consumption of lead pigments and compounds in products

Pigments and other compounds	%
Glass pigments	
Cathode ray tubes	40
Crystal	15
Speciality glass/optical	4
Light bulbs	3
Other pigments and compounds	
Plastic additives	23
Glazes	9
Paints	4
Ceramics	2

3.6.2 *Trends in uses of pigments and stabilisers*

Figure 3.5 shows the consumption of lead pigments and compounds in the EU15. The figures for the European Union in 1970-1990 are based on the trend of the consumption of lead in lead sheet in the OECD countries (OECD, 1993). The lead demand for cathode ray tubes reflects the increase in demand for television and computer terminals. The other applications of lead in glass have remained stable. The demand for lead glazes and other compounds in ceramics have been declining slowly as a result of technological changes. There has been a major decline in the use of inorganic lead compounds in paints. The EU countries no longer permit the use of lead paints in households. A small amount of lead in paint is used as corrosion inhibitor. The demand for lead in plastics increased by 70% during the 1970s and has since been relatively stable (OECD, 1993). For extrapolation of the use of lead compounds in the future, it is assumed that the present annual consumption for most of the applications will remain constant over the next 30 years. We have made two exceptions. First, the PVC industry is seriously considering reducing the use of lead stabilisers. We have assumed that by 2010 lead stabilisers will no longer be used. Furthermore, it is likely that the use of cathode ray tubes will decline as a result of the advent of flat screen displays. It is very hard to predict how fast such displays will totally remove cathode ray tubes from the market. We have assumed a decline to 25 % of the use in 2000 by 2030, with a proportional decline in the years in between. Annex 1.2 gives the detailed model calculations.

Table 3.16 shows the consumption of lead for pigments and compounds in different countries in the EU15 over the period 1993-1997. In this study it is assumed that the consumption of pigments and compounds in the EU15 equals the production.

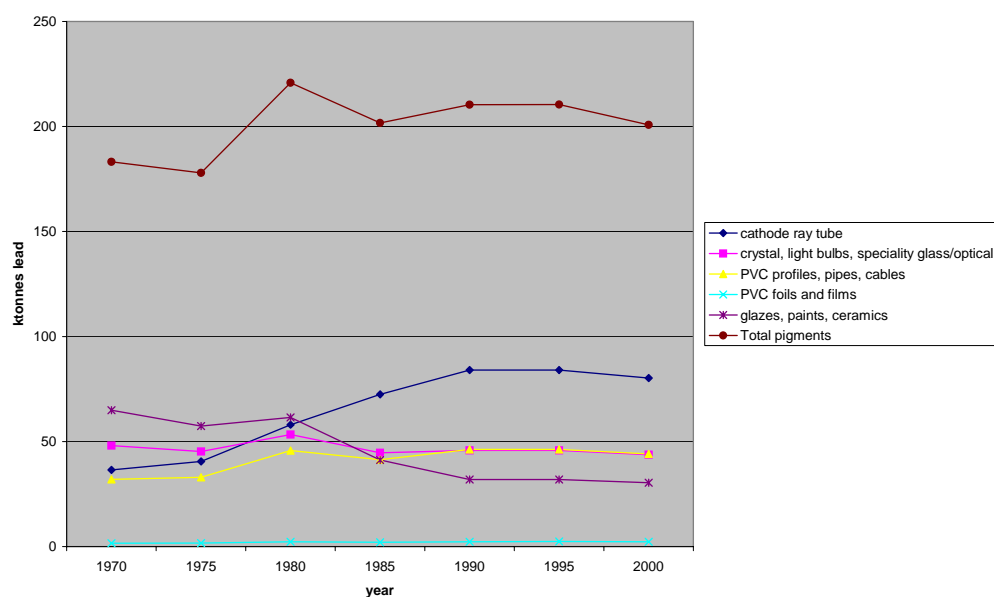


Figure 3.5 The consumption of lead pigments and compounds in the EU15 in 1970-2000.

Table 3.16 The consumption of lead (kilotonnes) for the manufacture of pigments and compounds in the EU15 in 1993-1998. (ILZSG, 2000)

	1993	1994	1995	1996	1997	1998
Austria	11.00	12.50	14.00	15.50	16.00	17.00
Belgium	4.10	4.20	3.30	2.00	1.60	1.20
France	23.80	24.00	27.30	21.30	18.30	18.00
Germany	79.80	77.20	73.20	68.90	73.50	70.20
Italy	41.90	40.10	42.00	39.60	37.10	33.40
Netherlands	10.00	12.00	12.00	10.00	13.00	10.00
Spain	15.30	17.00	20.00	21.00	23.00	25.00
United Kingdom	17.50	18.60	18.70	17.90	18.30	18.50
TOTAL	203.40	205.60	210.50	196.20	200.80	193.30

3.6.3 Emission factors for lead pigments and stabilisers

In the literature no emissions during the use of pigments and are reported. The use of lead in paints is assumed to be negligible. The most important leakage to the environment will probably take place in the disposal stage of the products containing pigments or stabiliser. Lead in glass is assumed to be immobile and will therefore not lead to emissions in incinerators and in landfill sites. The lead in PVC is also considered to be immobile²⁶. However, if PVC is incinerated it will lead to air emissions and leakage of lead from landfill.

None of the products containing pigments and stabiliser are assumed to be recycled. The main source of lead in cathode ray tubes is from TV sets and computer monitors.

²⁶ This might be a rather positive assumption, though. Some sources suggest that lead might be mobilised (e.g. Hjertberg and Gevert, 1995). The discussion is of minor importance given the likely phase-out of the use of lead as stabiliser in PVC.

Very little recycling is done in Europe at present. However, recycling rates for electrical and electronic equipment are predicted to increase in the future. Leaded glass could be returned to glass manufacturers for recycling. At present, the glass industry is not doing this because there is no economic incentive to do so (Scoullos, Vonkeman & Thorton, 2001). It is likely that the EU's proposed Directive on Waste for Electrical and Electronic Equipment (WEEE) will improve this situation. It is difficult to derive targets however. The current draft of the directive suggests that a collection rate of 4 kg/inhabitant for WEEE should be reached. The WEEE sets this absolute value rather than a collection percentage since the total amount of WEEE in the EU is not clear. However, in section 11.2.2 of the explanatory memorandum, a value of 6 million tonnes is suggested, which at a collection rate of 4 kg/EU inhabitant means that about 1.6 million tonnes (or 25 %) will be collected. Dutch data (134,000 tpa disposed (VROM, 2001), 16 million inhabitants) suggest an amount of 8.5 kg WEEE per inhabitant, implying that the EU target would mean a collection rate of some 50 %. The draft WEEE states that 75% of equipment with cathode ray tubes that have been collected should be recycled or recovered. With a collection rate of 25 % or 50 % this implies overall recycling rates of 19 to 37.5 %, assuming that average collection rates and recycling rates will be applicable for cathode ray tubes. Since Dutch data suggest that for the main sources of cathode ray tubes (TVs, monitors) very good collection rates can actually be achieved, we have assumed 66 % collection after 2005 (and hence with 75 % recycling an overall recycling rate of 50 %).

In some countries, like the Netherlands, there are some initiatives to recycle PVC building materials like profiles and pipes. Currently, only a small fraction of PVC products are recycled. Cables are sometimes recovered after use for the recycling of copper. However, it is often not economically feasible to collect old cables, as collection can be more expensive than the value of the metals. For the future, however, the PVC industry is trying to set up recycling schemes. Roughly, the aim is that some 50 % of the collectable waste will be recycled. We have assumed that some 50% of the applications analysed here (pipes, cables) will not be collected but left in the environment after their useful life has ended. Because of the long lifetime of PVC products (50 years or more), the products have not yet been discarded. However in the future, large amounts of PVC products will be discarded from the stock. In future, attention should be given to the waste treatment. Incineration of the PVC, in waste incinerators or during copper recycling of cables might lead to an increase of emissions to air.

In table 3.17 the emissions during production, use and waste disposal of the products containing pigments and stabilisers is given. Annex 1.2 gives more detailed calculations for this product group. The emission during production is based on the air emission from industry of 358 tonnes in 1998 (EMEP, 2001) attributed to 'pigments and other compounds' proportional to the consumption of lead in products in the EU15 (see table 3.5). The air emissions of waste incinerators is based on the emission of 124 tonnes in 1998 (EMEP, 2001) attributed to the different products proportional to the estimated waste streams from discarded products (see table 3.7).

Table 3.17 Emissions of lead (in kilotonnes) during production, consumption and disposal of products containing lead pigments and stabiliser in the EU15.

Year	Production	Consumption			Waste treatment							
	Air emission	Inflow of Pb	Outflow of Pb	Emission	Remains in environment	Recycling	Landfill	Incineration	Emission from incineration to air	Stock on landfill	Emission from landfill to soil	Stock in environment
						80%	20%				See Annex 1.	
1965												
1970		183	2	0	0	0	1	0		8	0.00	0
1975		178	2	0	0	0	1	0		16	0.01	0
1980		221	2	0	0	0	2	0		28	0.01	0
1985		202	104	0	0	0	83	21		545	0.09	0
1990		210	148	0	0	0	119	30		1288	0.16	0
1995		211	167	0	0	0	134	33		2123	0.24	0
2000	0.04	201	169	0	0	8	142	35	0.04	3010	0.31	80
2005	0.04	191	163	0	0	8	134	37	0.03	3865	0.37	160
2010	0.03	134	194	0	16	50	98	30	0.03	4503	0.43	240
2015	0.03	124	188	0	16	48	93	31	0.04	5125	0.48	320
2020	0.03	114	176	0	16	43	86	31	0.04	5711	0.53	400
2025	0.02	104	167	0	17	38	81	32	0.04	6274	0.59	482
2030	0.02	94	170	0	23	37	77	33	0.05	6827	0.65	597

3.7 Alloys

3.7.1 Uses of lead in alloys

In 1998 the consumption of lead for alloys in the EU15 was about 32 kilotonnes (2% of the total lead consumption in the EU15). In the Western world the consumption was 140 kilotonnes (3% of the total lead consumption in the West). Lead alloys are used in a wide range of products including:

- Solder, extensively used by the electronics industry, primarily to make electrical connections on printed circuit boards;
- Solders in cars, plumbing and food cans;
- Terne plate, iron and steel can be protected against corrosion by coating the surface with a lead-based alloy;
- Brass and bronze products, to reduce wear and friction in certain types of machinery.

3.7.2 Trends in uses of alloys

Figure 3.6 shows the consumption of lead for alloys in the EU15. The figures for the European Union in 1970-1990 are based on the trend in the consumption of lead for alloys in the OECD countries (OECD, 1993). The use of lead is declining for the majority of the applications, with the exception of electronics. However, the decline of these other applications is steadily decelerating. As for electronics, the future use will be mainly determined by the outcome of the current debate surrounding the possible replacement of lead solder with alternative alloys. In our projections, we have assumed that such measures will not be taken, and that the application of other alloys will stay at the level of 2000 (32 ktpa). Overall this might have given a slight overestimate of the future consumption.

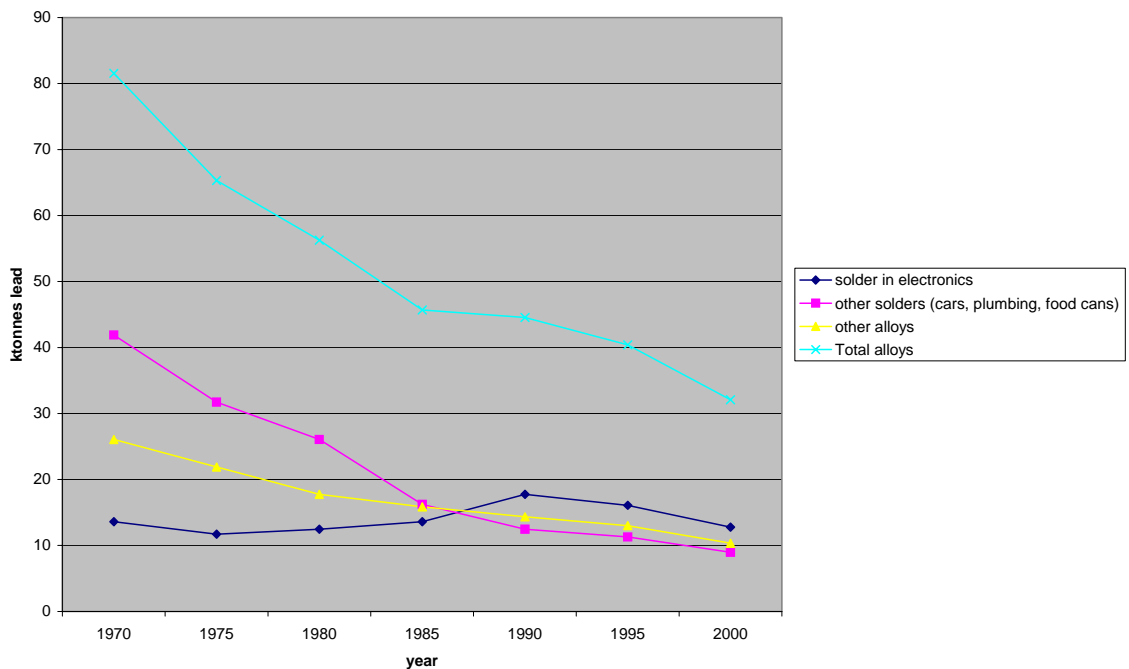


Figure 3.6 The consumption of lead for production of alloys in the EU15

Table 3.18 shows the consumption of lead for the manufacture of alloys for different countries in the EU15. In this study it is assumed that the consumption of alloys in the EU15 equals the production.

Table 3.18 The consumption of lead (kilotonnes) for the manufacture of alloys in the EU15 in 1993-1998. (ILZSG, 2000)

	1993	1994	1995	1996	1997	1998
Alloys						
France	4.20	4.90	5.20	5.30	4.40	4.40
Germany	8.20	9.40	8.30	7.50	8.20	7.60
Italy	3.00	3.00	3.60	3.50	3.40	3.30
United Kingdom	21.50	22.70	23.30	19.50	16.80	16.80
TOTAL	36.90	40.00	40.40	35.80	32.80	32.10

3.7.3 Emission factors for lead from alloys

The emissions of lead during use of alloys are considered to be negligible. Most emissions will occur during the disposal of the products. Except for the solders in cars, it is assumed that the alloys are not recycled. The EU's WEEE directive may in future ban the use of lead solder, but this is currently still a point of discussion. We have considered a lead ban for this application in our calculations. The EU's WEEE directive sets recycling targets for electronic waste, but in general does not set targets for specific components. We think it is unlikely that a tiny fraction like solder will be addressed in the recycling schemes that will be set up under this directive; it is most likely to end up in the 'grey' residual fraction after separation and recycling bound for incineration and landfill. Hence, we stayed on the cautious side and did not assume recycling for lead solder in WEEE.

Table 3.19 shows the emissions during production, use and waste disposal of the products containing alloys. For 2000 and earlier, a split between landfill and incineration has been used of 80-20% for waste disposal; for 2030 this split has been set at 70-30% with a proportional increase in between. Recycling has been set at 80 % for other alloys (bronze, etc.), 0% for solder in electronics, and 100 % for other solders used in applications that consist mainly of metals that can be assumed to be recycled (cars, tin cans, etc.) Detailed calculations per product sub-group are given in Annex 1.3. The emissions during production are based on the air emissions from industry of 358 tonnes in 1998 (EMEP, 2001) attributed to 'alloys' proportional to the consumption of lead in products in the EU15 (see table 3.4). The air emissions of waste incinerators is based on the emission of 124 tonnes in 1998 (EMEP, 2001), attributed to the different products proportional to the estimated waste streams from discarded products (see table 3.7).

Table 3.19 Emissions of lead during production, consumption and disposal of alloys in the EU15. (kilotonnes lead)

Year	Production Air emission	Consumption			Waste treatment						
		Inflow of Pb	Outflow of Pb	Emis- sion	Remains in enviro- nment	Recy- cling	Landfill	Inciner- ation	Emission from incin- eration to air	Stock on landfill	Emission from land-fill to soil
In %							80 [2000] 70 [2030]	20 [2000] 30 [2030]			0.23 [leakage] 0.008 [corros.]
1965											
1970		82	0	0	0	0	0	0		0	0.000
1975		65	0	0	0	0	0	0		0	0.000
1980		56	0	0	0	0	0	0		0	0.000
1985		46	82	0	0	63	15	4		94	0.002
1990		45	65	0	0	49	13	3		174	0.003
1995		40	56	0	0	40	13	3		254	0.005
2000	0.01	32	46	0	0	29	13	3	0.01	338	0.006
2005	0.01	32	45	0	0	24	16	4	0.01	441	0.008
2010	0.01	32	40	0	0	22	14	4	0.01	535	0.010
2015	0.01	32	32	0	0	17	11	4	0.01	609	0.011
2020	0.01	32	32	0	0	17	11	4	0.01	683	0.013
2025	0.01	32	32	0	0	17	11	4	0.01	758	0.014
2030	0.01	32	32	0	0	17	10	4	0.01	832	0.015

3.8 Shot and ammunition

3.8.1 *Uses of lead in ammunition*

In 1998 the consumption of lead for the manufacture of lead shot in the EU15 was about 59 kilotonnes (4% of the total lead consumption in the EU15). In the Western world the consumption was 115 kilotonnes (2% of the total lead consumption in the West). About 80% of the lead shot is used as ammunition; the other 20% is used as an additive in certain steel alloys to improve the steel machinability (OECD, 1993).

3.8.2 *Trends in uses of ammunition*

Table 3.20 shows the production of lead shot and ammunition for different countries in the EU15. The production of lead for ammunition in the future is assumed to be constant and is estimated to be about 44 kilotonnes per year (an average for the past years).

Table 3.20 The production of lead shot and ammunition in the EU15 (kilotonnes Pb) (ILZSG, 2000)

	1993	1994	1995	1996	1997	1998
France	7.90	8.00	6.50	8.00	7.00	7.50
Germany	13.20	14.40	14.70	14.00	14.60	13.60
Italy	23.50	21.90	23.70	23.60	24.60	26.40
Spain	4.50	4.00	5.00	5.00	5.00	5.50
United Kingdom	3.50	3.50	5.50	5.20	5.30	5.60
TOTAL	52.60	51.80	55.40	55.80	56.50	58.60

3.8.3 *Emission factors for lead from ammunition*

A large amount of lead particles is emitted to the soil annually due to the use of lead in ammunition in hunting and sport shooting. In table 3.21 some estimates are given for the annual emission of lead particles in the European Union based on the above production figures (table 3.20), and based on extrapolated emissions from the Netherlands (Booij et al, 1993; Eerd & Stiggelbout, 1992) and Denmark (Lassen & Hansen, 1996) using population ratios as extrapolation factor.

Table 3.21 Emission of lead particles to the soil (kilotonnes Pb)

Emission data derived from production figures in the EU15		Emission data based on extrapolated emissions			
Consumed ¹	Emission ²	Based on Dutch data ³		Based on Danish data ⁵	
45	23	5	14	14	19

1 Ammunition is assumed to be 80% of production of shot and ammunition (OECD, 1993); all ammunition produced in EU is considered to be used in EU.

2 Not all end use is considered as loss in the environment: in some shooting sports the bullets are collected, it is assumed that 50% of the use is emitted.

3 Source: Booij et al, 1993.

4 Source: Eerd & Stiggelbout, 1992.

5 Source: Lassen & Hansen, 1996.

Because lead particles can be consumed by fowl, the emission of lead particles to the soil can be regarded as toxic releases of metallic lead. However the corrosion of lead from particles to the soil, groundwater and surface water is also an exposure route of lead. Little is known of the corrosion of lead from lead particles in the soil. However, it is generally agreed that the corrosion will increase in acid environments, which is the case in the humus layer of the soil. The corrosion is assumed to be 1% per year (Stoop & Rennen, 1990). In order to estimate this corrosion from lead particles in the soil, it is necessary to estimate the stock caused by the use of lead as ammunition. In table 3.22, the corrosion from stock as accumulated over the past 100 years is estimated assuming different emissions of lead particles to the soil. In the overall comparison of the emissions due to products, in this study the loss of ammunition to the soil is estimated to be 14 kilotonnes per year.

Table 3.22 Stock and corrosion from stock of ammunition to the soil in the EU15 based on different constant emissions of particles (kilotonnes Pb)

Year	Constant inflow of particles (kilotonnes Pb)					
	22		14		5	
	stock	corrosion	stock	corrosion	stock	corrosion
2000	1414	14	880	9	328	3
2005	1412	14	905	9	337	3
2010	1410	14	928	9	346	3
2015	1408	14	951	10	354	4
2020	1406	14	972	10	362	4
2025	1405	14	992	10	369	4
2030	1403	14	1011	10	376	4

1 Stock in 2000 is the result of the constant inflow of particles over the last 100 years minus the emission during these years.

2 Corrosion is assumed to be 1% per year (Stoop & Rennen, 1990).

3.9 Cable sheathing

3.9.1 Uses of lead in cable sheathing

Lead can be extruded into a continuous covering to prevent water penetration of underground or underwater power and telecommunication cables. Although this use is recyclable, its demand has been declining because of the development of alternative technologies that replace paper-insulated lead cables with polyethylene and PVC. At present the remaining market is mainly for marine cables. For submarine cables, lead is superior to other cable sheathing materials. It is assumed that lead for cable sheathing is no longer used for underground cables on land. However large amounts of cables used in the past may still be lying underground not yet having been disposed of or collected.

3.9.2 Trends in uses of cable sheathing

In table 3.23 an estimate is given of the consumption of cable sheathing in the European Union extrapolated from the consumption in OECD countries (OECD, 1993) in the period 1965-1990, using population ratios. As indicated above, for the future situation we have assumed that lead will be used in cable sheathing only for applications at sea, and that this demand will be stable in the future.

Table 3.23 Production and consumption of lead in cable sheathing in the EU15 (kilotonnes)

	Produced	Consumption ¹	
		Continent	Sea ²
1965	167	109	58
1970	145	87	58
1975	113	55	58
1980	92	34	58
1985	59	1	58
1990	58		58

OECD, 1993; EU15 data derived from production in OECD countries

¹ All cables produced in EU15 are assumed to be used in EU15.

² In 1990 the market was mainly marine cables; it is assumed that in the previous years this market was constant.

3.9.3 Emission factors for lead from cable sheathing

These inflows in the past can now be used to estimate the stock of cables in the ground and the emissions from this stock in the future. Little is known about the corrosion of lead from cable sheathing. In this report, the same annual corrosion factor is used as for lead sheets in the building sector (8E-7 kg Pb / kg cable sheathing). Although it is stated that the cables are recyclable, several scenarios are calculated assuming different collection percentages after disposing of the cables (table 3.24). It is often not economically feasible to collect old cables, as collection can be more expensive than the value of the metals. The lead sheathing has a relatively low commercial value; cables are recovered primarily for their more valuable copper. Cables are more likely to be recovered if they are in accessible locations (e.g. in soil, rather than on the sea bed), and have high copper contents (for example, high-power cables) (Scullos, Vonkeman & Thorton, 2001). In the overall comparison of emissions from products, in this study the recycling of cable sheathing is assumed to be 50%. The remainder is assumed to be left in the ground. It is assumed that no new lead cable sheathing was applied in the EU15 after 1985; this implies that in the long run only a stock in soil will remain due to the fact that not all cable sheathing is recycled. Landfill and incineration are irrelevant for this product group.

3.10 Lead additives in petrol

3.10.1 Uses of lead additives in petrol

Lead tetraethyl and tetramethyl petrol additives improve engine performance. The use of lead as an additive has declined very much because of the introduction of catalytic converters in new vehicles to reduce SO₂, NO₂ and other gas emissions. Unleaded gas was developed for these vehicles. The second reason for the reduction was that governments reduced the permissible level of lead in petrol because of human health concerns. (OECD, 1993). In 1998 the production of lead as an additive for petrol in the EU15 was about 38 kilotonnes (2% of the total lead consumption in the EU15). In the Western world the consumption was 46 kilotonnes (1% of the total lead consumption in the West).

Table 3.24 Inflow, discard, outflow stock and emission of lead in cable sheathing in the EU15, based on four different recycling percentages. (kilotonnes)

Year	Consumption	Discarded ²	10% discarded cables remain in soil to recycling			25% discarded cables remain in soil to recycling			50% discarded cables remain in soil to recycling			100% discarded cables remain in soil to recycling		
			Stock ¹	Corrosion ³	Emission	Stock ¹	Corrosion ³	Emission	Stock ¹	Corrosion ³	Emission	Stock ¹	Corrosion ³	Emission
1965	109		0	545	0.04	0	545	0.04	0	545	0.04	0	545	0.04
1970	87		0	1090	0.08	0	1090	0.08	0	1090	0.08	0	1090	0.08
1975	55		0	1525	0.12	0	1525	0.12	0	1525	0.12	0	1525	0.12
1980	34		0	1799	0.14	0	1799	0.14	0	1799	0.14	0	1799	0.14
1985	1		0	1967	0.15	0	1967	0.15	0	1967	0.15	0	1967	0.15
1990	0	109	98	1973	0.15	82	1973	0.15	55	1973	0.15	0	1973	0.15
2000	0	87	78	1482	0.12	65	1563	0.12	44	1700	0.13	0	1972	0.15
2005	0	55	49	1089	0.08	41	1236	0.10	27	1481	0.12	0	1971	0.15
2010	0	34	30	842	0.07	25	1030	0.08	17	1344	0.10	0	1971	0.15
2015	0	1	1	689	0.05	1	903	0.07	1	1258	0.10	0	1970	0.15
2020	0	0	0	683	0.05	0	897	0.07	0	1255	0.10	0	1969	0.15
2025	0	0	0	683	0.05	0	897	0.07	0	1254	0.10	0	1968	0.15
2030	0	0	0	683	0.05	0	897	0.07	0	1254	0.10	0	1968	0.15

1 Stock in 1965 is the result of a constant inflow of 109 kilotonnes Pb over the period 1960-1965.

2 Lifetime of cables is assumed to be 25 years, so the cables from 1965 are discarded in 1990.

3 Corrosion is assumed to be 8E-5 kg Pb / kg cable sheathing.

3.10.2 *Trends in uses of lead additives in petrol*

In the European Union in 1990 the emission of lead to air due to lead additives in petrol was about 17 kilotonnes. In 1997 these emissions were reduced to 1.6 kilotonnes (EMEP, 2001). It is assumed that lead additives in petrol will gradually be phased out. Assuming a continuation of the trend between 1990 and 1997, the emission of lead due to lead additives in petrol is expected to be zero in 2010 (see table 3.25).

3.10.3 *Emission factors for lead additives in petrol*

Table 3.25 shows the emission of lead from lead additives in petrol in the EU15.

Table 3.25 The emission of lead from lead additives in petrol in the EU15 (in kilotonnes)

Emission to air	
1990	17.00
1997	1.60
2000	0.15
2005	0.01
2010	0.00
2015	0.00
2020	0.00
2025	0.00
2030	0.00

Note that the production of lead additives in the EU15 is very much higher than the consumption of the additives in petrol (table 3.26). Probably all the additives produced in the United Kingdom are exported to countries outside the EU15.

Table 3.26 The production of lead additives for petrol in the EU15. (kilotonnes)(ILZSG, 2000)

	1993	1994	1995	1996	1997	1998
France A	4.70	5.10	5.70	5.00	-	-
Germany A	2.50	2.50	2.50	-	-	-
Italy A	4.30	5.20	3.90	-	-	-
United Kingdom A	38.80	35.30	35.00	38.20	36.20	37.90
TOTAL	50.30	48.10	47.10	43.20	36.20	37.90

3.11 **Miscellaneous**

3.11.1 *Uses of lead in miscellaneous products*

There are several applications of lead with a small economic market in a variety of products, like yacht keels, counter balances, curtain weights, fishing sinkers, type metal, galvanising, annealing, plating and miscellaneous casting). In 1998 the total consumption of lead in the EU15 for the manufacture of these miscellaneous products was about 65 kilotonnes (4% of the total lead consumption in the EU15). In the Western world the consumption was 154 kilotonnes (3% of the total lead consumption in the West).

Lead is used as weight for the balancing of wheels in vehicles and the balancing of ships in keels. Lead-tin-antimony alloys are sometimes cast into pieces for typesetting in letterpress printing. Thin layers of lead can be electrically plated on to steel items for corrosion resistance. Lead is also used in manufacturing processes of other products. In galvanising, a layer of molten lead is used in the bottom of a molten zinc bath to assist the separation of impurities as zinc coatings are applied to steel for corrosion resistance. Baths of molten lead are also used in annealing processes for controlled cooling of steel products during manufacture. Lead is reused in these processes (OECD, 1993).

3.11.2 Trends in uses of miscellaneous products

Some uses of the products, like weights for balancing wheels, are increasing along with the growth in the number of vehicles in Europe. Some of the applications have declined, like type metal. Most of the applications show a small but stable demand for lead (OECD, 1993). Overall it is assumed that the future demand for lead for miscellaneous products will remain stable, at a level of 65 kilotonnes per year.

3.11.3 Emission factors for lead from miscellaneous products

Except for the loss of fishing sinkers into water, the use of the miscellaneous products are not expected to lead to emissions during the use of the products. So most of the emissions will occur during the waste treatment of the products. It is assumed that about 50% of the miscellaneous products are recycled. For the non-recycled part, a split between landfill and incineration has been used of 80-20% for 2000 and earlier; for 2030 this split has been set at 70-30% with a proportional increase in between. The average lifetime of the products is estimated to be 15 years. Table 3.27 shows the emissions during production, use and waste disposal of the miscellaneous products .

Table 3.27 Emissions of lead during production, consumption and disposal of miscellaneous products in the EU15. (kilotonnes lead)

Year	Production Air emission	Consumption			Waste treatment						
		Inflow of Pb	Outflow of Pb	Emis- sion	Remains in envir- onment	Recy- cling	Land- fill	Inciner- ation	Emission from incin- eration to air	Stock on landfill	Emission from landfill to soil
						50%	40%	10%			Leakage 0.23% Corros. 0.008%
1965		65									
1970		65		0	0	0	0	0		0	0.000
1975		65		0	0	0	0	0		0	0.000
1980		65	65	0	0	33	26	7		163	0.003
1985		65	65	0	0	33	26	7		326	0.006
1990		65	65	0	0	33	26	7		488	0.009
1995		65	65	0	0	33	26	7		651	0.012
2000	0.01	65	65	0	0	33	26	7	0.02	814	0.015
2005	0.01	65	65	0	0	33	25	7	0.02	977	0.018
2010	0.01	65	65	0	0	33	25	8	0.03	1139	0.021
2015	0.01	65	65	0	0	33	24	8	0.03	1302	0.024
2020	0.01	65	65	0	0	33	24	9	0.03	1465	0.027
2025	0.01	65	65	0	0	33	23	9	0.03	1628	0.030
2030	0.01	65	65	0	0	33	23	10	0.03	1790	0.033

The emissions during production are based on the air emissions from industry of 358 tonnes in 1998 (EMEP, 2001) attributed to ‘miscellaneous products’ proportional to the consumption of lead in products in the EU15 (see table 3.4). The air emissions of waste incinerators are based on the emission of 124 tonnes in 1998 (EMEP, 2001) attributed to the different products proportional to the estimated waste streams from discarded products (see section 3.3.3.).

During sport fishing on inland waters, emission of lead to the water and sediment occurs due to the loss of sinkers. No data are available for the consumption of lead for sinkers (part of the group miscellaneous). Data for the loss of sinkers are derived from data given for the Netherlands and Denmark using population ratios as extrapolation factor. In table 3.28, emissions for the European Union are given based on these Dutch (Van den Berg & De Poorter, 1994) and Danish data (Lassen & Hansen, 1996). The annual loss of lead in the environment is considered to be constant. No corrosion factors for lead in water are found in literature. Assuming a constant loss of sinkers to the environment of three kilotonnes per year, the stock of lost fishing sinkers in 2000 is estimated to be 338 kilotonnes. In 2030 the stock will be 439 kilotonnes. If a yearly corrosion rate of 0.008% is assumed, this will lead to a lead emission to the water/sediment of 0.03 kilotonnes per year.

Table 3.28 Loss of fishing sinkers to inland waters, sediment in the EU15 extrapolated from Dutch and Danish emission data (kilotonnes).

	Based on Dutch emission ¹	Based on Danish emission ²
1997	1	3
2000	1	3
2005	1	3
2010	1	3
2015	1	3
2020	1	3
2025	1	3
2030	1	3

1 Source: Van den Berg & De Poorter (1994).

2 Source: Lassen & Hansen (1996); The emission of lead to water is estimated to be 105-275 tonnes / yr. This includes both emissions to fresh water and seawater. The consumption of lead for angling in fresh water is considered to be 25% of the total consumption for fishing in fresh and sea water

Note: Figures represent emission of particles, not corrosion from particles to the water.

3.12 Non-intentional uses in the agricultural sector

The use of lead-contaminated phosphate fertiliser leads to a direct emission of lead to the agricultural soil. The other fertilisers used by the agricultural sector are sewage sludge and manure. Together with the deposition from air these are indirect sources of lead in the environment, which can be traced back to the original product emissions (corrosion lead sheets, corrosion water pipes, petrol additives, lead in coal, etc.). So the emissions to the agricultural soil due to the reuse of these ‘waste materials’ in the agricultural sector are **not** additional emissions to the environment but are caused by emissions all ready mentioned in the previous paragraphs, like air emissions from industry and waste incineration and water emissions by corrosion from water pipes, etc .

In table 3.29, the lead balance in the agricultural sector is given. The balance is based on the Dutch balance using different extrapolation methods. The load due to fertilisers was estimated using the fertiliser use in the Netherlands and the EU15 (FAO, 2001). The load by sewage sludge was based on population ratios between the Netherlands and the EU15 corrected for the percentage of households connected to the sewer. The deposition from air was estimated from the emission data for lead in the EU15 (EMEP, 2001) assuming that all the emitted lead in the EU15 is also deposited in the EU15 and that 50% is deposited on agricultural soil and 50% is deposited on non-agricultural soil.

Table 3.29 Load of lead in the agricultural soil in the EU15 (tonnes)

	1990	1991	1992	1993	1994	1995	1996	1997	1998
Crop production									
Total in:	10908	5766	4942	4427	5371	4229	3873	3009	2772
Fertiliser	350	332	299	301	312	310	322	313	307
Sewage sludge	511	513	515	518	520	522	523	525	526
Deposition from air	9355	4250	3472	2949	3879	2742	2375	1507	1279
Manure	692	671	655	659	660	655	653	664	660
Others									
Total out:	1025	1002	984	991	992	987	985	999	995
Fodder	805	781	762	768	768	763	760	773	768
Food crops	220	221	222	223	224	225	225	226	226
Animal production									
Total in:	805	781	762	768	768	763	760	773	768
Fodder	805	781	762	768	768	763	760	773	768
Total out:	805	781	762	768	768	763	760	773	768
Food animal products	113	110	107	108	108	107	107	109	108
Manure	692	671	655	659	660	655	653	664	660
Emission to soil									
Agricultural sector	9883	4764	3958	3436	4379	3242	2888	2010	1777

3.13 Review of waste flows and emissions per product group during production, use and waste treatment

3.13.1 Flows to waste treatment

On the basis of the former sections, table 3.30 and 3.31 give the total waste flows to incinerators and landfills in 2000 and 2030. Of course these data have to be interpreted with care, since the assumptions made can have a considerable influence on the outcome of the analysis. Having said this, the following can be concluded from the tables, taking into account the assumptions made:

Table 3.30: Lead flows to landfill(kilotonnes), estimates for 2000-2030

Year	Batteries	Extruded products			Pigments and compounds					Alloys	Miscellaneous	Total	Total incl. incineration residues
		Lead sheet	Water-pipes	Other extruded products	Stabiliser in PVC (pipes, cables and profiles)	Stabiliser in PVC (foils and films)	Glazes, ceramics	Cathode ray tube	Glass other				
2000	32	16	p.m.	4	6	2	33	58	43	13	26	233	291
2005	33	16	p.m.	3	6	2	25	66	35	16	25	228	291
2010	34	15	p.m.	3	6	0	24	32	35	14	25	190	247
2015	34	16	p.m.	3	6	0	23	30	34	11	24	182	243
2020	34	16	p.m.	3	6	0	22	26	32	11	24	174	238
2025	34	13	p.m.	3	6	0	22	22	31	11	23	166	231
2030	35	12	p.m.	3	8	0	21	18	31	10	23	160	229

Table 3.31: Lead flows to incinerators (kilotonnes), estimates for 2000-2030

Year	Batteries	Extruded products			Pigments and compounds					Alloys	Miscellaneous	Total
		Lead sheet	Water-pipes	Other extruded products	Stabiliser in PVC (pipes, cables and profiles)	Stabiliser in PVC (foils and films)	Glazes, ceramics	Cathode ray tube	Glass other			
2000	8	4	p.m.	1	2	0	8	14	11	3	7	58
2005	9	4	p.m.	1	2	1	7	18	10	4	7	63
2010	10	5	p.m.	1	2	0	7	10	11	4	8	58
2015	11	5	p.m.	1	2	0	8	10	11	4	8	61
2020	13	6	p.m.	1	2	0	8	9	12	4	9	63
2025	14	5	p.m.	1	2	0	9	9	12	4	9	66
2030	15	5	p.m.	1	3	0	9	8	13	4	10	69

- The highest flows to landfill and incineration are related to cathode ray tubes and glass. This is related to the low current recycling rates for these applications (Scoullos et al, 2001). For the future situation we have assumed a considerable replacement by flat display screens, and a net collection/recycling rate of 50 %, implying a much lower flow to waste. Since we assumed glass is inert, these assumptions have no influence on our forecasted emissions from landfill (see next sections);
- Despite our rather high assumptions concerning recycling (95% or more), batteries appear to be the next most important flow to landfill and incineration. The outflow of lead to landfill/incineration can only be lowered if even higher collection rates are achieved. However, experience shows that for small quantities of hazardous waste collection rates of 95% are already an outstanding achievement;
- The inflow of lead to landfill and incineration related to alloys is for some 80% caused by lead solder in electronics. This is all related to our assumption that this material will not be recycled. Of course, one could argue that in the future the WEEE directive will require considerable recycling of electrical and electronic waste. Yet, to our knowledge, the current draft directives will not set demands for recycling of the solder. On mass basis, solder is just a tiny fraction of an average piece of electronic equipment. In order to meet the recycling targets, it is most likely that the electronics industry will give priority to recycling the big parts of the equipment (e.g. plastics from housing, etc.). It is most likely that after mechanical or chemical recycling the solder will end up in a residual flow bound for incineration or landfill. Even in the case of incineration, the lead fraction will end up via slag or fly ash in landfill after all. Hence, unless specific proof is given that solder is likely to be recycled as such, we find it irresponsible to apply high recycling rates for this specific fraction of electrical and electronic equipment;
- For lead sheet, the flow to landfill is not negligible either, despite the recycling rate of 90 % we have assumed. We feel that this recycling rate is more likely to be overestimated than underestimated. Separating lead from building or demolition waste is quite laborious, and it can be expected that in quite some cases buildings will be demolished and the residues sent to landfill without that lead being separated;
- The lead in PVC might become a problem of the past, since the PVC industry will most likely cease the use of PVC in the next decade;
- The amount to landfill/incineration from ‘miscellaneous’ is strongly influenced by our assumed recycling rate of 50 %.

Of course it is possible to dispute some of the assumptions made above. One could also look at the analysis from a different perspective. If one wants to ensure that lead flows into the waste stage are minimised, the analysis above makes clear that the following conditions are helpful to achieving such a goal:

- Ensure recycling rates for batteries and lead sheet of 98 / 95 % and more;
- Ensure that the future WEEE directive leads to recycling schemes that steer the lead in electronics to a truly recycled fraction, and not a residual fraction that still needs landfill or incineration;
- Make a clear choice whether or not landfill of ceramic materials and glazes can be accepted in view of their (likely) inertness. If it is decided that landfill of such material is not desirable, ensure high recycling rates (90% or more) are achieved for these flows.

3.13.2 *Emissions to the environment*

In tables 3.32 and 3.33, a summary is given of the emissions during the production, use and disposal of the lead products for the years 2000 and 2030. The products and their accompanying emissions have been taken from the detailed discussions in section 3.4-3.12.

On the basis of our analysis, some decline of emissions to air can be expected, but the differences don't seem dramatic. This general conclusion corresponds well with an earlier analysis by TNO, that suggested that apart from phasing out lead from petrol, the lead emissions to air in Europe would only marginally decline between 1990-2010 (TNO as cited in Thornton, 2001). A few comments can be made with regard to table 3.32 and 3.33, though:

- For 2000, the largest share of the emissions to air is due to the combustion of fossil fuels (natural contamination of fossil fuels). We found no clear reason to assume that the emission factor for such processes would be lower in 2030 than in 2000.
- In 2030 the use of lead as additive in petrol is assumed to have been phased out. However, since the use was already low in 2000, this lowers the total emission by just 150 tpa.
- Another important source of emissions to air is the emission of lead during production processes, especially from smelters. Smelter emissions are allocated to the products based on the relative production of the products. Therefore the largest share is for production of batteries. Since battery production is assumed to increase based on trend extrapolation, the air emission during production increases as well. As discussed extensively in Chapter 3.3.3, we saw no good reason to assume a major improvement of emission factors in the next 30 years (cf. TNO as cited in Thornton, 2001). Yet, if this assumption is incorrect, it is clear that the emissions in 2030 will be lower rather than higher.
- An important exposure route of lead for humans is the corrosion of lead from water pipes. In 2030 the emissions will be reduced because the stock of water pipes will slowly diminish as a result of the assumption that lead pipes are no longer applied in new buildings. Yet, since there is no clear policy that will ensure removal of water pipes these emissions will still continue.
- The most important emission to soil is related to the use of ammunition. Consumption levels will not increase, but the stocks of spent ammunition in soils will still grow due to the very long residence time, implying higher emissions in future.
- Another important contribution to the emission to soil is related to lead sheet. These stocks are also growing, and hence the emissions as well. As we discussed in Chapter 3.5, the data on the historical lead stock in buildings and the emission factor are very uncertain. Furthermore, the fate of the lead corrosion from lead sheets is not clear. In the Netherlands most of the emission is collected in the sewer and is treated in the sewage treatment. The effluent containing a small part of this lead is emitted to the surface water. The sewage sludge is partly dumped, partly incinerated, and partly used as fertiliser on agricultural soil. In other European countries this situation will certainly be different. We have assumed a direct emission to soil of 50 %, and an additional 25 % emission to soil via sewage sludge. It is clear that this lead sheet issue needs practical research into emission factors, the fate of the run-off, etc. A complication is that the use of lead sheet is mainly concentrated in the UK, the Netherlands, Belgium, and Ireland, so that

regional differences are very important. There could be a double counting with the indirect emissions mentioned under 'sludge', since the sludge mentioned there is likely to be mainly contaminated by run-off from lead sheet (cf. Annema et al, 1995).

- According to the model we used, the leakage of lead from landfill to soil is considerably lower than from, for example, lead shot. As indicated before, emission factors for lead in landfill are fraught with uncertainty. In the model calculations we used, metallic lead and lead in ceramics are seen as slightly or not mobile, leading to the situation that incineration residues from plastics have a main contribution. Yet, other assumptions could have indicated other priorities. If we had followed the assumptions advocated by some LCA experts, i.e. that metals will become available for a considerable part on a long time frame, much higher emissions could have been calculated.
- As for incineration, our emission for 2030 corresponds with an emission factor that is equal to or lower than the emission factor valid for incinerators complying with the EU's incineration directive. It could be that for 2000 we under-estimated the emissions from incineration to air, since not all waste incinerators in the EU were equipped with the most advanced flue gas cleaning in that year. Furthermore, we assumed that in 2000, a high recycling rate of 95 % for car batteries had already been reached, which might be an overestimation implying a much higher flow to incineration than assumed here. For 2030, emissions from incineration have become relatively important for two reasons. First, there is a shift from landfill to incineration in the EU. Second, for some product groups, particularly car batteries, more lead will enter the waste stage than in 2000, due to a steady incline in use (however, as indicated, the 95 % recycling in 2030 might in fact be a much better rate than achieved in practice in 2000, for which we also assumed 95 % recycling).

Table 3.31 Lead emission estimates in 2000 for the EU15 from product during production, use and waste disposal (kilotonnes lead per year). Uncertainties, particularly in the model calculations for emissions to soil, are high and can easily be several dozen percent or more (see earlier sections)

Initial emission :	Production	Use				Waste treatment	
	Air	Air	Drinking water	Sedi- ment ⁶	Soil	Air	Soil
Batteries	0.21					0.03	0.02
Rolled and extruded products							
- Lead sheet	0.04				0.89	0.01	0.01
- Water pipes ¹			0.28			p.m.	p.m.
- Other extruded products	0.01					0.00	0.00
Pigments	0.04					0.04	0.31
Ammunition ²	0.01				8.80		
Alloys	0.01					0.01	0.01
Cable sheathing ³	0.01				0.10		
Additives in petrol	0.01	0.15					
Miscellaneous	0.01			0.03		0.02	0.01
Fossil fuels		0.48					
Fertiliser					0.31		
Manure ⁴					0.66		
Sludge ⁵					0.53		
TOTAL	0.35	0.63	0.28	0.03	11.29	0.12	0.36

1 Based on Dutch emission factors, probably underestimated.

2 Based on the corrosion from a stock in the environment due to an annual loss of lead ammunition to the agricultural soil of 14 kilotonnes.

3 Based on the corrosion of lead in cable sheathing, assuming that 50% of the cables are left in the environment after disposal.

4 Internal cycle in agricultural sector: depends upon inflow phosphate fertiliser, sewage sludge (corrosion lead sheet, water pipes) and deposition from air.

5 Based on Dutch data, probably overestimated: indirect flow depends on corrosion lead sheet, water pipes and air deposition.

6 The emission to the sediment is based on the corrosion from a stock in the environment due to a yearly loss of fishing sinkers to the water of 3 kilotonnes.

Table 3.32 Lead emission estimates in 2030 for the EU15 from product during production, use and waste disposal (kilotonnes lead per year). Uncertainties, particularly in the model calculations for emissions to soil, are high and can easily be several dozen percent or more (see earlier sections)

Initial emission :	Production	Use				Waste treatment	
	Air	Air	Drinking water	Sedi- ment ⁶	Soil	Air	Soil
Batteries	0.25					0.06	0.04
Rolled and extruded products							
- Lead sheet	0.05				0.89	0.02	0.02
- Water pipes ¹			0.22			p.m.	p.m.
- Other extruded products	0.01					0.00	0.00
Pigments	0.02					0.05	0.65
Ammunition ²	0.01				10.11		
Alloys	0.01					0.02	0.02
Cable sheathing ³	0.01				0.07		
Additives in petrol	0.00	0.00					
Miscellaneous	0.01			0.03		0.04	0.03
Fossil fuels		0.48					
Fertiliser					0.31		
Indirect emissions							
Manure ⁴					0.66		
Sludge ⁵					0.53		
TOTAL	0.37	0.48	0.22	0.03	12.57	0.18	0.76

1 Based on Dutch emission factors, probably underestimated.

2 Based on the corrosion from a stock in the environment due to an annual loss of lead ammunition to the agricultural soil of 14 kilotonnes.

3 Based on the corrosion of lead in cable sheathing, assuming that 50% of the cables are left in the environment after disposal.

4 Internal cycle in agricultural sector: depends upon inflow phosphate fertiliser, sewage sludge (corrosion lead sheet, water pipes) and deposition from air.

5 Based on Dutch data, probably overestimated: indirect flow depends on corrosion lead sheet, water pipes and air deposition.

6 The emission to the sediment is based on the corrosion from a stock in the environment due to a yearly loss of fishing sinkers to the water of 3 kilotonnes.

4 Preliminary Risk Assessment

4.1 Introduction

This chapter gives the preliminary risk assessment for various applications of lead. We will make a distinction between the following situations:

- Direct or semi-direct exposure of workers to lead emissions during the production and use of the products discerned in Chapter 3;
- Exposure of the general public, which can take place via the following channels:
 1. Semi-direct exposure due to the fact that people live close to factories where lead is processed. This situation is in a way comparable to exposure of workers, though in general the exposed group is seen as more critical (and for whom more stringent standards will apply), whilst the exposure will probably be lower due to the larger distance to the source.
 2. Exposure via food, drinking water and environmental media (particularly air and soil/dust). This is in general a rather indirect exposure, with the exception of food canned in lead-soldered tin cans, and drinking water delivered via lead pipes.

In section 4.2 to 4.4, typical exposure data given in literature will be presented for each exposure situation. The exposure data will be compared with the acceptable values for exposure of workers and/or the general public presented in the concluding section of Chapter 2. This comparison allows for assessment of whether the exposure is above acceptable concentrations, providing the input for the preliminary human health risk assessment. We will use the most authoritative limit value found in our toxicological literature survey (see Chapter 2). Apart from daily intakes, where possible we will evaluate on the basis of Pb blood levels as well. This concerns:

- For exposure of *workers*, the main Occupational Exposure Limits (OELs) will be used. As indicated in Chapter 2, occupational health standards for lead range from 0.05 mg/m³ (in the US and Norway), to 0.1 mg/m³ (a.o. Germany, the Netherlands, Denmark and Japan) and 0.15 mg/m³ (a.o. France, Spain and the UK). Recently, the EU's Scientific Committee on Occupational Exposure Limits (SCOEL) also recommended a value in this range (i.e. 0.1 mg/m³), and suggested blood limit values of 30 µg/dl.
- For exposure of the *general public*, the most authoritative limit value is the Tolerable Weekly Intake of 25 µg/kg b.w. established by the JEFCA and adopted by the WHO in its guidelines for drinking water quality. This value equals an average Tolerable Daily Intake of 3.6 µg/kg b.w. As for blood limit values for young children, some bodies have suggested a value of 10 µg/dl.

The section on indirect exposure of the general public will contain data concerning concentrations in air, water and soil. These data can be compared with the (ecotoxicological) NOAECs presented in Chapter 2, allowing for an ecotoxicological risk assessment as well (see section 4.6). It goes without saying that this 6-month study has important limitations and can in no way be compared with the formal, multi-year risk assessments currently performed under the EU's existing substances programme. Hence, the main goal is to indicate on which points additional information is needed, rather than to give a final assessment of risks.

Having said this, it should also be stated that an evaluation purely based on classical risk assessment leaves out some important issues relevant for a non-degradable material like lead. Risk assessments executed in a traditional way tend to give no more and no less than a 'picture' of risks to human health and environment at a given moment in time. As we saw in Chapter 3, and as we will see further in this chapter, the system of substance flows related to lead is quite dynamic. Hence exposure situations in the past might not be valid for the present, and those encountered in the present might not be valid in the future. Where for many degradable substances one can easily assume that reducing overall emissions from present sources will ensure a steadily lower trend in exposure in the future, for a non-degradable material like lead the situation is much more complicated. For instance, exposure reduction measures in the past tended to be concentrated on sources that led to rather direct exposure to lead (emissions to air via leaded petrol, contamination of water due to the use of lead pipes and solder in the water transport system, and reducing the use of lead solder for food cans). As we will see in this chapter, this resulted in a relatively rapid decline in the (semi-direct) exposure of humans to lead. However, it is also clear that the *use* and hence the economic *stock* of lead is growing, as is the stock in landfills and certain residues from MSWIs (which might be re-used). This led to a discussion driven by the fear for slow 'enrichment' of the economical and ecological system with lead, and hence a possible higher exposure in future.

Indeed, several national authorities in the EU therefore think that a purely risk-based approach, as we described above, is not sufficient to deal with the long-term risks of use of lead. For instance, in Sweden a criterion was suggested that would prevent a significant enhancement of existing lead concentrations in soil. Denmark recently established a (controversial) lead ban in relation to the desire to prevent a slow accumulation of lead in the ecosystem. Though the Danish step in particular has not gained general acceptance (e.g. LDA, 2001), we feel that just looking at classical risks without evaluating the rationale for these alternative approaches would do no justice to the debate that is actually taking place at EU level. Indeed, this debate can be seen as representative of the current discussion as to whether substance policy should be based solely on a 'risk assessment' approach, or if it should be strengthened and supplemented by a more 'precautionary' approach. Box 4.1 gives an overview of the main positions in this debate. Reference is also made to the EU's position paper on the Precautionary Principle, and documents written for the EU's new chemicals policy. In brief, the EU acknowledges here that precaution is a sound principle in policy making, but also takes care not to make its use of this principle too absolutist. For instance, the EU states that precautionary measures should, among other things, be proportional to the chosen level of protection, non-discriminatory in their application, based on potential benefits and costs, etc. (EU, 2000; cf. EU, 2001). Despite this guidance issued by the EU, one can expect the practical interpretation of how to apply the precautionary principle in a specific case to raise quite some discussion.

To generate information about potential long-term risks and potential problems related to accumulation, in principle a dynamic substance flow analysis (SFA) for lead should be executed that covers both the economic and the ecological system. This falls well beyond the mandate in the contract for this project. Yet, in section 4.6 we will give a semi-quantitative impression on the basis of some data available for some EU member states where SFAs for lead have been performed. This allows an assessment of whether or not there is a rationale behind fears of developments like a 'slow enrichment' of the environment with lead.

*Box 1: A description of frames in the toxicity controversy (from: Tukker, 1999)***The risk assessment frame**

The *first* frame broadly follows the classical risk assessment approach. It is therefore called here the *risk assessment* frame. In terms of *problem analysis*, this frame basically believes that the whole emission-effect chain can be analysed, and that meaningful assessments of final effects are possible. Information on emission volumes, fate-related properties and toxic properties of a substance is used to calculate a *prediction* of (likely) effects. In terms of *management solutions*, the approach is to reduce such effects (or better: effect scores) to a certain threshold. In sum, (predicted) effects on species and ecosystems are used as a starting point for managing toxic substances. This frame reflects:

- a high degree of confidence in the capability of mankind to acquire adequate knowledge about emissions of substances, their fate and effects;
- a high degree of confidence in technological emission reduction measures and in flawless, skilled behaviour of the people that manage these systems;
- a high degree of confidence that nature can cope with the consequences of errors made by man in assessing the effects of substances and managing emission abatement technologies.

The strict control frame

The *second* frame acknowledges that risk evaluations have limitations. In terms of *problem analysis* they are sensitive to the possibility that concentrations that are currently regarded as safe may be severely flawed. They also use information on emissions, fate and toxicity in their problem analysis, but in a different manner than the adherents of the risk assessment frame. In particular, by using fate information (derived, for example, from biodegradation tests and octanol-water partition coefficients), they discern three groups of substances, classified according to uncertainties in effect assessment. Substances that are not readily biodegradable and that are alien to nature are viewed as the most sensitive category. They have a long lifetime in the environment, so their emission leads to irreversible contamination. Errors in risk estimates can hardly be corrected by reducing emissions. Such corrections can much more easily be made for substances that are alien to nature but that are readily degradable. Finally, substances that are also naturally produced form the least sensitive category. In terms of *management solutions*, naturally occurring substances may be emitted up to a level that depends on the natural background. For degradable, non-natural substances, a risk assessment approach is still acceptable. But persistent substances should be kept out of the environment. Since this frame is based firmly on a belief in the technical and organisational feasibility of maintaining substances in closed loops, its adherents opt for a strict minimisation of emissions of such substances (rather than phasing them out). To sum up, adherents of this frame choose emissions from the production-consumption chain as a starting point for substance management. I therefore call it the *strict control* frame. It reflects:

- a moderate to high degree of confidence in the capability of mankind to gain adequate knowledge about emissions of substances, but low confidence in the ability to acquire knowledge about their fate and effects;
- a high degree of confidence in technological emission-reduction measures and in flawless, skilled behaviour by the people that manage these systems;
- a moderate degree of confidence that nature can cope with the consequences of errors by man in assessing the effects of substances and managing emission abatement technologies.

The precautionary frame

The *third* frame assumes that risk evaluation is too weak a basis for management of toxic substances. I shall call this the *precautionary* frame. This frame is particularly adopted by environmental pressure groups (e.g. Thornton, 2000). In terms of *problem analysis*, it tends to classify substances in the same way as the strict control frame. However, the precautionary frame is even more pessimistic about the feasibility of effect assessments. For example, the issue of endocrine substances convinced the CPC (1997) that toxicity assessments are so fallible that toxicity is not a useful criterion in substance policy. The analysis concentrates on identifying substances for which this lack of knowledge seems most important, making use of all possible information sources. Indications that important amounts of by-products are formed in a process, or the unexplainable occurrence of substances in nature that belong to the same group, for example, are seen as additional reasons for suspicion. So this frame, too, uses information on emissions and fate, for instance, but once again a differently from the other two frames. Rather than the quantitative effect calculations favoured by the risk assessment frame, the precautionary frame adopts a more qualitative, descriptive analytical framework, making quite holistic use of the available information. In terms of *management solutions*, this frame is pessimistic about the practical effectiveness of control measures. A preventive and precautionary approach is preferred, which implies a choice in favour of alternatives that are inherently safer (in terms of known toxicity problems and potential uncertainties). In particular, the emissions of persistent and bioaccumulative substances should be fully stopped. This goal should be realised by a phase-out, since all materials-handling implies that the material sooner or later ends up in waste, or will be directly released into the environment. This is a management scheme that is concerned with the *production processes* themselves. It reflects:

- a low confidence in the capability of mankind to gain adequate quantitative knowledge about emission, fate and effects of substances, and a preference for a rather holistic, qualitative evaluative approach;
- a low degree of confidence in technological emission-reduction measures, and the idea that we should expect that people will make mistakes;
- a low degree of confidence that nature can cope with the consequences of errors by man in assessing the effects of substances and managing emission abatement technologies;
- hence, a strong preference for inherently preventive and cautious options.

4.2 Occupational exposure

4.2.1 Batteries

Table 4.1 reviews the exposure data for various steps in the production process of batteries in the UK's HSE's National Exposure Database as 8-hour Time Weighted Averages (TWAs). Exposures are between 0.003 and 5.4 mg/m³, with an average of 0.1 mg/m³. The most critical process steps are pasting, burning, plate-cleaning and brushing (HSC, 1995). WHO (1995) gives similar values: an average value 0,3 mg/m³, and a maximum value of 2 mg/m³.

Table 4.1: Exposures in the battery manufacturing industry in mg/m³ (HSE, 1995)

Process	Geometric Mean	Minimum	Maximum	Sample Size
Overall	0.1	0.003	5.4	1,400
Casting	0.08	0	0.69	156
Pasting	0.12	0.004	2.2	286
Hand assembly	0.12	0.02	4	159
Assembly	0.06	0.01	2.8	306
Cast-on strap	0.11	0.01	5.4	178
Formation	0.11	0	0.6	77
Lead oxide mill	0.04	0.01	0.34	33
Plate-cleaning	0.12	0.02	0.86	84
Machine-stacking	0.09	0.01	2.3	109

4.2.2 Rolled and extruded products

This product group consists mainly of lead sheets used in building, and radiation-shielding materials. According to WHO (1995), grinding and turning processes lead to exposures in the order of magnitude of 0.1 mg/m³ (average) to 1 mg/mg (maximum).

4.2.3 Pigments and other compounds

The manufacture and use of lead-containing paint can be a critical application for workers. Exposure is highly dependent on control measures. WHO (1995) mentions values of 0.1 mg/m³ (average) to over 2.5 mg/m³ (maximum) for paint manufacture, and 0.25 mg/m³ (average) to some 3 mg/m³ (maximum) for spray painting. The National Exposure Database (NEDB) of the UK's HSE gives values for paint manufacture of 0.01 mg/m³ (mean) to 0.33 mg/m³ (maximum).

As for the use of stabilisers in plastics, exposure (WHO, 1995) is between 0.1 mg/m³ or less (mean) and 0.5 mg/m³ (maximum) during the manufacturing process.

For glass manufacture, the HSE's NEDB gives values of 0.18 mg/m³ (mean) to 0.95 mg/m³ (maximum). For workers in an electronic components plant that makes ceramic-coated capacitors and resistors using leaded glass for the ceramic coating, exposure levels up to 1.7 mg/m³ were at stake (ATSDR, 1999).

4.2.4 Alloys

As for the production of alloys, we have assumed that the data given by HSE (1995) for non-ferrous foundries are appropriate. The HSE's NEDB gives values of 0.08 mg/m^3 (mean) and 1.4 mg/m^3 (maximum).

A more critical application, however, is the use of alloys, particularly solder. Particularly under non-ventilated circumstances, exposure can be relatively high, e.g. up to 1 mg/m^3 in radiator repair shops (ATSDR, 1999).

4.2.5 Shot and ammunition

The main problem related to shot and ammunition is exposure to lead fumes in (closed) practice areas for shooting practice. Instructors in particular are a vulnerable group, since they spend more time at a shooting range than shooters. Exposure depends on the type of bullets used (non-jacket: pure lead, or jacketed: copper-coated) and can be up to $0.2\text{-}0.5 \text{ mg/m}^3$ in unfavourable circumstances (ASTDR, 1999).

4.2.6 Cable sheating

This product is produced in a similar way to rolled and extruded products. According to WHO (1995), grinding and turning processes lead to exposures in the order of magnitude of 0.1 mg/m^3 (average) to 1 mg/mg (maximum).

4.2.7 Lead additives in petrol

No exposure data have been found for the production of lead additives. However with the phase-out of leaded petrol, this application is less relevant for this study. For motor repair/servicing in general, the exposure to lead can be 0.05 mg/m^3 (mean) to 0.83 mg/m^3 (maximum).

4.2.8 Other products and processes

There are various other processes during which workers can be exposed. . These are shipbuilding, general construction and demolition, scrap metal and the refining/recycling industry. Particularly during recycling, exposures can be relatively high, up to 4 mg/m^3 as maximum (HSE, 1995). The geometric means for these processes tends to be in the range of 0.05 to 0.2 mg/m^3 at most (HSE, 1995).

4.2.9 Conclusions

Table 4.2 reviews the occupational exposure data given in earlier sections. The variability of the data, even within one process group, is striking. In this context, various authors have stated that *'The diversity and extent of the industrial applications of lead is such that it is impossible to make general statements about exposure levels. In many instances actual exposure levels have not been measured and often work is carried out in small enterprises which may not be subject to workplace controls or legislated requirements'* (WHO, 1995: 69).

In sum, airborne lead concentrations in the occupational setting vary considerably according to the type of industry and the level of industrial hygiene practiced at each plant. As indicated in Chapter 2, occupational health standards for lead range from 0.05 mg/m^3 (in the US and Norway), to 0.1 mg/m^3 (a.o. Germany, the Netherlands, Denmark and Japan) and 0.15 mg/m^3 (a.o. France, Spain and the UK). Comparison with table 4.2 shows that in practice, concentrations in workplace atmosphere in some individual cases might be above these standards. It should be noted that this does not necessarily

mean excessive exposure, as long as respiratory protection equipment is used. It is beyond the scope of this study to review all data with regard to blood level concentrations in relation to specific occupational exposure situations, though this might have been preferable, since ingestion of lead might not only depend on the concentration in air (cf. SCOEL, 2000). From the main investigations carried out between 1980 and 1995, the mean concentration of lead in blood in occupational exposure groups ranges from 20 to 30 µg/dl., which is in compliance with the limit values suggested by the SCOEL (2000). In any case, it is clear that in an occupational situation the (legal) OELs mentioned have to be met. The exposure data listed in table 4.2, particularly in view of the minimum values given, suggest that plants applying adequate worker protection and emission prevention should not exceed the OELs.

Table 4.2: Exposure to lead of workers in production processes (in mg/m³)

Industry	(Geometric) Mean	Minimum	Maximum	Sample Size	Reference
Batteries	0.3		2		WHO 1995
	0.1	0.003	5.4		HSE 1995
Rolled and extruded products					
Pigments and other comp.					
- Paint manufacture	0.01	0.01	0.33	34	HSE 1995
	0.1		2.5		WHO 1995
- Paint spray applications	0.25		3		WHO 1995
- Plastic production	0.1		0.25		WHO 1995
- Glass manufacture	0.18	0.02	0.95	27	WHO 1995
- Glass used for coating			1.7		ATSDR 1999
Alloys (non-f. foundries)					
- Non-ferrous foundries	0.08	0.01	1.4	41	HSE 1995
- Solder applications			1		ATSDR 1999
Shot and ammunition			0.2-0.5		ATSDR 1999
Cable sheeting	0.1				WHO 1995
Petrol (Motor repair/servicing)	0.05	0.01	0.83	41	HSE 1995
Other					
- Shipbuilding	0.12	0.05	0.17	13	HSE 1995
- General construction / demolition	0.19	0.03	1.5	10	HSE 1995
- Scrap metal	0.14	0.01	5.1	33	HSE 1995

For this study, this implies that production processes in which workers are exposed to lead must be monitored carefully. In cases where no adequate protection measures have been taken, lead exposure of workers can be critical. Strict enforcement of the applicable standards is necessary in order to avoid risks to the working population.

4.3 Exposure of the general public near production facilities

Exposure of the general public living near production facilities takes mainly place via the air. In well-managed situations, as is likely to be the case in most situations in Western Europe, no clear danger for humans or the environment seems to be at stake. Various analyses have been done on, for example, lead blood levels of the population living near lead industries (e.g. smelters), and no clearly elevated blood levels have been detected (cf. Farago et al, 1999).

Where emissions are largely uncontrolled – a situation unlikely to occur in the EU – the situation can be much worse. The WHO (1995) mentions an example of a lead smelter, with concentrations of lead in air of over $10 \mu\text{g}/\text{m}^3$ 50 meter from the smelter to $1.5 \mu\text{g}/\text{m}^3$ at 1 km (WHO, 1995). With an inhalation of 20 m^3 air per day for adults (weight 70 kg), this leads to an intake of some 0.4 to 3 $\mu\text{g}/\text{day}$ per kg b.w.; for children inhaling 5 m^3 air per day and a body weight of 10-15 kg the intake is slightly higher and can be above the TDI. The concentrations are in any case above the WHO air quality standard of 0.5-1 $\mu\text{g}/\text{m}^3$. In a report for the 'Black Triangle Study' executed by IIASA in the mid-1990s, for the Katowice district in Poland, Nowinska (1996) reported locally high annual lead concentrations in air ($1.6 \mu\text{g}/\text{m}^3$), high deposition rates, and resulting loads in soil that made grass and crops unsuitable for consumption by animals/humans.

This implies that under poorly controlled conditions, the population close to facilities where lead is used can be at risk. Though this conclusion has been based solely on data related to exposure via the air, if emission management is poor pathways like exposure via inhalation or ingestion of dust, and via crops grown on land close to the production process in question can also be critical. .

Hence, the conclusion here should be very similar to the conclusion drawn for worker exposure. Production processes where the the general public can be exposed to lead have to be monitored carefully. In cases where no adequate protection measures have been taken, lead exposure of the public living close to the factory can be critical. Strict enforcement of the applicable standards is necessary in order to avoid risks to the general population

4.4 Exposure of the general public via environmental media, water and food

4.4.1 Introduction

In principle, for the exposure of the general public via environmental media, water and food a full overview of links between emission, fate and exposure should be established to analyse what the main causes of an exposure are. As indicated in the introduction to this chapter, the confines of this study made such a comprehensive approach impossible. Therefore the general exposure via different pathways was assessed on the basis of different literature sources (section 4.4.2), compared with the available toxicological limit values, both in terms of intake (section 4.4.3) and possible influence on lead levels in blood (section 4.4.4). These two elements were used in the evaluation of human health risks (section 4.4.5). Section 4.6. discusses to what extent the dynamic development in the lead system might lead to changes in exposure.

4.4.2 Discussion per intake route

Various exposure data can be found in literature. The most important intake routes are via:

- Inhalation of air
- Ingestion of drinking water
- Ingestion of food
- Ingestion of soil and dust (particularly for children).

The exposures have varied considerably in the last decades, and depend greatly on specific situations. Lead exposure via inhalation used to be a very important exposure route, but since the virtual phasing out of leaded petrol in the nineties in the EU this route is now much less important. Drinking water can be an important exposure route, but only in those cases where leaded drinking water pipes are used, in combination with a relatively acidic water composition. Food surveys have shown that lead contamination of foodstuffs in general has declined over the years. Particularly in older food basket surveys the influence of lead soldered tin cans for storage of food is noticeable. Unfortunately, most of the review studies that we would take as a basis for this assessment concentrate on Western European countries. Specific data for the EU's accession countries (the Czech Republic, Poland and Slovenia) were scarce or absent. Our main data source was a series of reports written for IIASA's so-called Black Triangle²⁷ study (Anderberg et al, 1999). As already indicated in section 4.2., work of Nowinska (1996) suggests that due to historical (and probably still present) burdens of lead into the environment lead to standards normally used in the EU for soil and food quality being exceeded. However, no specific and recent analysis of the daily intake of lead for these countries was encountered.

Air

As for concentrations in air, table 4.3 gives data with different time frames for several locations. The table makes clear that in the 1980s and earlier there was a major difference between concentrations in rural and urban sites, and that particularly for urban sites about a factor 10 improvement of air quality has been reached as a result of decreased use of leaded petrol. In the Netherlands, levels in urban air are no longer much higher than levels in rural air.

Table 4.3: Concentrations of lead in air by location and time frame (in $\mu\text{g}/\text{m}^3$)

Approximate year Location	1980	1985	1990	1995	Ref.
USA, Urban	0.8 (1979)		0.1 (1988)		ATSDR 1999
USA, Urban			0.07 (1990)		WHO, 1995
Denmark, Urban				0.026 (1995)	Danish EPA, 1998
Denmark, Average				0.01 (1995)	Danish EPA, 1998
Netherlands, Urban		0.25 (1985)	0.11 (1989)	0.032 (1995)	Janus et al, 1999
Netherlands, Average		0.12 (1985)	0.08 (1989)	0.024 (1995)	Janus et al, 1999

Drinking water

As for concentrations in drinking water, the use of lead pipes and leaded solder for water pipes determines the concentration in tap water. In general, water leaving drinking water supplies contains just a few μg lead per litre. Such values have been reported for the US (WHO, 1995; ATSDR, 1999) and the Netherlands (Janus, 1999).

Tap water monitoring at the user gives the following ranges:

- 2.8 $\mu\text{g}/\text{l}$ for the Netherlands, 1993 (Janus et al 1999);
- 4 $\mu\text{g}/\text{l}$ for Denmark (Danish EPA, 1998; cf. CSTE, 2001);
- < 5 $\mu\text{g}/\text{l}$ during a test on 1 litre first-draw systems from lead-free systems in the US (ATSDR, 1999).

²⁷ The 'Black Triangle' is the area around the meeting point of the frontiers of Poland, the Czech republic, and former Eastern Germany

For leaded pipes and plumbing, however, concentrations can be much higher. In the Netherlands, 5 % of the population consumes water with a concentration of >50 µg/l and 1 % water with a concentration of > 100 µg/l. In the Danish water supply system, 'hot spots' occur with reported levels of 900 µg/l (Danish EPA, 1998; cf. CSTE, 2001), and for the US hot spots of 500 µg/l have been reported (ATSDR, 1999).

Food

For adults, food is the most important intake route. Data on intake vary quite a lot, mainly in relation to the time that a market basket survey or diet duplicate study was conducted. Table 4.4 gives some total intake data for different populations during different time frames, and related references. The table suggests a clear declining trend in lead intake via food. This might have to do with declining overall emissions to air, particularly from road traffic (which might have led to deposition on certain crops), declining levels in drinking water (which plays a role in food industry), a cease of the use of leaded materials for food cans and wine caps, etc.

Table 4.4: Daily intake via food by target group and approximate time frame (µg/day)

Approximate year Location/group	1980	1985	1990	1995	Cited in/by
<i>Adults</i>					
Canada	54 (1981)				ATSDR 1999:408
USA (1)	56.5 (1980-82)				ATSDR 1999:408
USA (2)	84 (1980)	28-40 (1984)	2.2-8.5 (1990)		ATSDR 1999:410
UK	71-110 (??)*				WHO, 1995
Belgium	96-282 (??)*				WHO, 1995
Japan		31 (??)*			WHO, 1995
Netherlands	107 (1976-78)	34 (1984- 1985)	24 (1988-89)	18 (1994)	Janus et al, 1999:56
Denmark		42 (1983-87)	27 (1988-92)	18 (1993- 97)	D. EPA, 1998; CSTE, 2000
<i>Children (age)</i>					
USA (6-11 months)	34 (1980)	16.7 (1984)	3.8 (1990)		ATSDR 1999:410
USA (2 yrs., male)	45 (1980)	23 (1984)	4.3 (1990)		ATSDR 1999:410
Netherlands (1-4 yrs., fem.)			9 (1988-89)		Janus et al, 1999:56
Denmark (1 yr.)			8 (1988-92)**		Danish EPA, Tab. 4.2

Note 1): For not all data sources it is clear if they included water as a source

Note *: Unclear for what year the survey was carried out. Year of survey estimated as 3-5 years before the literature reference was published.

Note **: Mean value. The 95 percentile is at 12 µg/day

As for the contributing foodstuffs, detailed breakdowns have been mainly found for the older surveys. They are given in table 4.5. These data have to be used with a lot of care since the trends indicated above may well imply that disproportional reductions of lead contents have been realised over the years.

Table 4.5: Food contributing to dietary lead levels in %, base year not known (WHO, 1995)

Country	Canada	Finland	Netherlands	UK
Food				
Vegetables	17	9	17	
Meat/fish/poultry	17			
Beverages	15	20		14
Cereals and products	15	24	25	15
Fruits and juices	10	22	9	
Milk and products		17		9
Wines and spirits			12	
Potatoes				10
Canned vegetables				8
Other	26	8	37	44
Total	100	100	100	100

Soil and dust

For small children, intake of dust can be an important uptake route. Often, intakes of 200 mg dust per day are taken into account (WHO, 1995; CSTEE, 2001; Danish EPA, 1998; Scoullou, 2001), where the Danish EPA took as a worst-case assumption 3 g/day for children showing pica behaviour, an unnatural consumption of soil/dust. The actual intake further depends on the concentration of lead in the ingested soil and dust. Typical background values in soil are between 10 and 30 mg/kg d.w. (ATSDR, 1999). The Danish EPA (2001) mentions a background value of 11 mg/kg d.w. for Danish agricultural soil. Soil adjacent to roadways and industrial facilities can contain much higher levels, several 100 mg/kg being no exception. Various authors do not hesitate to include worst-case concentrations in their calculations of 1000 mg/kg d.w. (Scoullou et al, 2001). As for dust, the situation can be even worse. Particularly in houses that have not been renovated, and where in former times lead-containing paint has been used, lead levels in dust can be relatively high²⁸. A study by Thornton et al (1994) showed that the concentrations in house dust in 10 % of the homes surveyed in a UK study had an excess of 2,000 mg/kg lead in house dust. These concentrations tend to increase with the age of the property. The study covered 100 households in different geographical locations. Concentrations in large cities tended to be higher, and a clear hot spot was formed by a region with a long history of lead mining and smelting. A study by Wang et al (1997) on dust concentrations in houses in Birmingham, UK, showed concentrations of about 300 mg/kg in the mid-eighties and 100 mg/kg in the mid-nineties. However, the maximum values were up to a factor 8 or more higher.

4.4.3 *Calculation of total exposure*

Based on the data given above, a total exposure will now be calculated for two types of groups: adults and children. In order to calculate the total intake, the weight of adults has been set at 70 kg, their daily respiratory volume at 20 m³, and their daily drinking water need at 2 litres. For children, the weight has been set at 10 kg, the daily respiratory volume at 10 m³, and the daily drinking water need at 1 litre (cf. Scoullou et al, 2001; Danish EPA, 1998; EC, 1993). To calculate the daily intakes we have used the following data. Where reasonable and defensible, as far as possible we tried to use the

²⁸ However, renovation could have a negative effect on short term. ATSDR (1999:416) reported that in newly renovated houses blood levels in children were elevated, suggesting that renovation of homes previously painted with leaded paint may increase at least temporarily a child's exposure to lead dust.

same data as the Danish EPA in its 1998 study, since it is this report that gave the main push to the current debate on lead in the EU. The data used are :

- Concentration of lead in air: $0.01 \mu\text{g}/\text{m}^3$ (also used by the Danish EPA in its calculations). This is the lowest value given in table 4.2 and will hence not lead to an overestimation of exposure.
- Concentration of lead in drinking water: $2.6 \mu\text{g}/\text{m}^3$ (value used by the Danish EPA not clear). This is the value in sanitised housing in the Netherlands, and likely to be a realistic value for exposure in the future.
- Daily intake via food: $27 \mu\text{g}/\text{day}$ for adults and $9 \mu\text{g}/\text{day}$ for children (used by the Danish EPA in its calculation). This value is much lower than many of the older market basket and diet duplicate studies, but given the data for the Netherlands seems to be appropriate for the early nineties. The Danish data for 1995 suggest a further 33 % decline in five years, and the US data in table 4.2 suggests that these intakes might eventually even be halved.
- Daily intake via soil and dust: $0.2 \text{ g}/\text{day}$ at a concentration of $100 \text{ mg}/\text{kg}$ for children only.

Making use of these assumptions, in table 4.6 the daily intake is calculated. It has to be noted that particularly for children the calculated intake can change dramatically if one uses somewhat different assumptions with regard to concentrations of lead in soil or dust, and daily intakes of soil/dust. Furthermore, it has to be stressed that effects are largely related to whether blood levels will be enhanced, which is a function of many other parameters, like bioavailability. For this issue we refer to the next section.

Table 4.6: Contribution of intake routes to daily intakes of lead

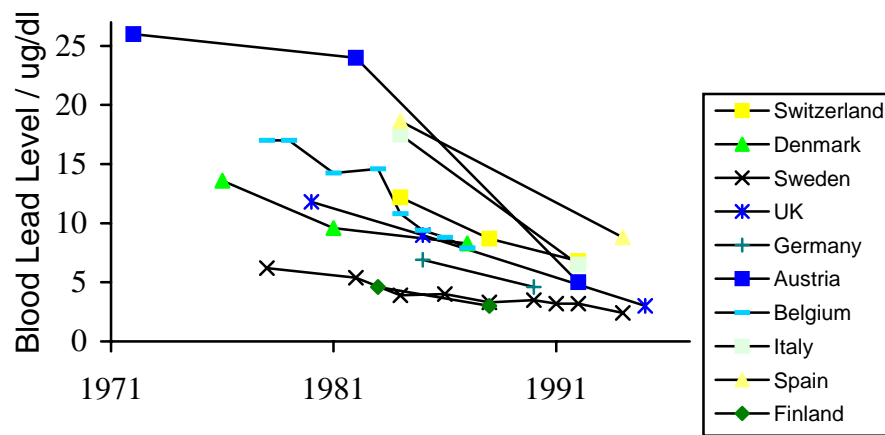
	Concentration	Unit	Intake volume	Unit	Total (μg)	$\mu\text{g}/\text{kg}$ b.w.	% of TDI ($3.6 \mu\text{g}/\text{kg}$ b.w./day)
Adults (70 kg)							
Air	0.01	$\mu\text{g}/\text{m}^3$	20	m^3	0.2	0.003	0.1%
Drinking water	2.6	$\mu\text{g}/\text{l}$	2	l	5.2	0.080	2.2%
Food					27	0.386	10.7%
Soil/dust			0				
Total						0.469	13.0%
Children (10 kg)							
Air	0.01	$\mu\text{g}/\text{m}^3$	10	m^3	0.1	0.01	0.3%
Drinking water	2.6	$\mu\text{g}/\text{l}$	1	l	2.6	0.26	7.8%
Food					8	0.8	22.2%
<i>Subtotal</i>						<i>1.09</i>	<i>30.3%</i>
Soil/dust	100	mg/kg	0.2	g	20	2	55.6%
Total						3.07	85.8%

4.4.4 The relation between exposure and blood levels

As already indicated in Chapter 2, an important issue with lead is not only the daily intake, but also the influence on levels in blood. The point is that lead exposure in the form of different compounds/contexts via different intake routes can lead to rather different absorption rates in the human body. At the same time, in most cases the critical effects due to exposure to lead seem to be related to the levels of lead found in blood. A further complication is that most of the lead absorbed is taken up in the bone, where it is in dynamic equilibrium with the blood. Hence, lead absorption from the past might still influence present blood levels.

In general, lead blood levels have declined over the last twenty years in the Western world from some 20 $\mu\text{g}/\text{dl}$ to 5 $\mu\text{g}/\text{dl}$. (SCOEL, 2000). Figure 4.2 gives an impression of this finding²⁹. It is likely that these declines are mainly related to the cessation of the use of lead in petrol, the reduction of the use of lead in solder for food cans, and diminishing use of lead pipes for tap water transport. As a general rule, one can state that lead blood levels in the EU population tend to be well below the 10 $\mu\text{g}/\text{dl}$. (SCOEL, 2000). For children in the US, a similar observation has been made; geometric mean blood levels in the early nineties were lowered to some 3 $\mu\text{g}/\text{dl}$., with only 4.4 % exceeding the 10 $\mu\text{g}/\text{dl}$ (ATSDR, 1999).

Figure 4.1: Decline in levels of lead in blood in various Western countries



As can be already deduced from Chapter 2, it is not easy to give predictions of blood levels on the basis of exposure/intake data. The uncertainties in studies that try to link these two elements appear to be important. The ATSDR (1999:416) states that 'lead levels of indoor dust and outdoor soil were found to be strongly predictive of PbB levels in over 200 urban and suburban infants followed from birth to 2 years of age'. A study by Berglund et al (as cited in Thornton, 2001:102) however, suggests that lead in soil and dust had little effect on blood concentrations. A study by Lanphear et al, cited by both ATSDR (1999: 417) and CSTE (2000:21), suggests that an increase of lead concentrations in soil to 400 mg/kg leads in 11.6% of children to blood levels of over 10 $\mu\text{g}/\text{dl}$. As indicated in Chapter 2.2.2, the actual absorption of lead from soil might depend on whether it is ingested with a meal or after a period of fasting. It is obvious that the scope of our study is too limited to solve this discussion about the relation between lead levels in soil and blood levels in young children.

4.4.5 Risk assessment

As for the assessment of risks to human health, two approaches can be taken. First, one can compare the values given in table 4.4 with the TDI as accepted by the WHO of 3.6 $\mu\text{g}/\text{kg}$ b.w./day. Second, one can compare the (expected) blood levels related to these intakes with a value of 10 $\mu\text{g}/\text{dl}$. As indicated in Chapter 2, for children some evidence

²⁹ The figure has been made available by the Lead Development Association, and included in this report after a cross-check with most of the original literature (Ducoffre et al., 1989; Begerow et al., 1994; Stromberg et al., 1995; Ponka et al, 1993; Moller and Kristensen, 1992; Nielsen et al., 1998; Bono et al., 1995; Rodamilans et al., 1996). Since the studies might have a somewhat different basis the results between countries are not always comparable, and the figure should therefore mainly be seen as an illustration of a trend of falling lead blood levels in the Western population.

of lesser growth and a small decrease in IQ might be involved at higher levels, a reason for some authoritative bodies to see this as an intervention level³⁰. It must be noted though, that using such a low blood lead value is not without dispute. This leads to the following conclusions.

Conclusion for adults

From the assessment it follows that adults in general will have an intake well below the TDI (a factor of about 10). This conclusion corresponds with the opinion of the CSTEE on the Danish Notification on lead (CSTEE, 2000), which concluded that the daily mean intake is 7 % of the TDI and the 95 % percentile 11% of the TDI. Blood levels in adults in the Western world tend to be well below 10 µg/dl., and hence the critical concentrations mentioned in Chapter 2 of 30 µg/dl. Only in the case of very high pollution of drinking water (e.g. levels of 100 µg/l or more) might the TDI be exceeded. Furthermore, under certain local and specific circumstances (a diet of foodstuffs from relatively contaminated soil, exposure due to living near lead using processes, etc.) risk limits might be exceeded.

Conclusion for children

For young children, however, the situation is much more difficult to assess clearly. The main reason is that the main intake for young children is ingestion of dust/soil, and the assumptions made for the concentration of lead in this dust and the amount ingested determine the outcome. Under the assumptions made by the Danish EPA, with an intake of 0.2 g/day and a lead content in soil of 100 mg/kg (a value which is seen in the Netherlands as just above the 'reference value' for good soil quality) the TDI is virtually reached. From table 4.4 it is obvious that using just slightly worse assumptions (not to mention the contamination levels of 1000 mg/kg as suggested by Scoullou et al, 2001) the TDI for young children will be clearly exceeded. Of course, the daily intake via food might be somewhat overestimated in view of the decline that has been noted for adults in 1995 in Denmark, and the lower levels of exposure in the USA. Table 4.4 makes clear, however, that such assumptions are of minor importance in the assessment. The intake via dust/soil dominates³¹.

We feel that it is quite difficult to come to a clear-cut, final conclusion as to whether young children might be at risk or not. First, the calculations and assumptions made for ingestion of soil/dust by children are fraught with uncertainty, and there might be major differences in behaviour between different populations. Second, blood levels of lead in children seem to be in most cases below 10 µg/dl. Hence, an important question is whether exceeding the TDI due to ingestion of soil will truly lead to elevated blood levels. But precisely on this point there is quite some debate about the exact relation, since absorption in the body of lead in soil seems less than the absorption of lead from other sources (ATSDR, 1999; compare also: CSTEE, 2000). The literature cited in section 4.4.2 and 4.4.4 suggest that at least lead in dust (e.g. related to old lead containing paint) might cause problems. Third, (potential) exceeding of the TDI most probably occurs during a limited period of a few years, whereas the TDI in principle is

³⁰ For adults, one could argue that on the basis of chapter 2 one should take a blood level of 30 (female) or 40 (male) µg Pb/dl as a reference. Since even the 10 µg/dl is hardly reached for adults this is mainly of theoretical value.

³¹ It has to be noted that the situation in other EU member states (or accession countries) than Denmark and the Netherlands might be less favourable as assumed in table 4.4. In quite some EU member states average lead contamination of drinking water, and the ingestion of lead via food might still be higher than assumed here. Even without taking ingestion of dust into account there is just a factor 3 between calculated intake and the TDI, which is not an excessive gap.

meant to give protection over a lifetime. In sum, this makes it quite difficult to make a final statement. Exceeding the TDI might occur regularly, but whether this leads to elevated blood levels is not clear.

Given the above, we tend to draw the following conclusion. For adults, in principle there seems at present *no need for further information and/or testing and no need for risk reduction measures beyond those already being applied*. In countries where key measures like diminishing the use of leaded petrol, banning lead drinking water pipes, and banning lead from materials in contact with foodstuffs have not yet been taken, *there is a need for limiting the risks taking into account those risk reduction measures which are already being applied*.

For children, *there is a need for further information and/or testing*. The questions that arise are: a) the relation between ingestion of different types and quantities of soil/dust and blood levels; b) the quantities soil/dust likely to be ingested by hand-mouth behaviour and the concentration of lead therein; and c) which blood level value (10 µg/dl or lower/higher) should be taken as a reference. In countries where key measures like diminishing the use of leaded petrol, banning lead drinking water pipes, and banning lead from materials in contact with foodstuffs have not yet been taken, *there is a need for limiting the risks taking into account those risk reduction measures which are already being applied*.

4.5 Environmental risks

As for environmental risks, the concentrations of lead in the environment will be compared with the no-effect concentrations given in Chapter 2.

As indicated in the former sections, typical concentrations in soil range from 10-30 mg/kg d.w. in relatively clean areas to a few dozen mg/kg d.w. In the Netherlands, the reference value of 85 mg/kg lead is met in most cases (Janus et al, 1999). According to the Danish EPA (1998) concentrations in urban soil in Denmark range between 30 and 500 mg/kg. From table 2.5 in this report it appears that the lowest critical values for chronic exposure for terrestrial organisms are in the range of 50-60 mg/kg d.w., a value suggested by the CSTE (2000) as a PNEC. This implies that there may be harmful effects associated with the soils that contain clearly higher amounts of lead than the normal background value (e.g. higher than the reference value of 85 mg/kg use in the Netherlands). It is often suggested that the highly contaminated areas have been caused by emissions from some time ago (e.g. in relation to car traffic, industries emitting lead dust, etc.). This is probably true in quite some cases, but a few questions have to be posed in this respect. Table 3.32 and 3.33 show that corrosion of lead plating might be an important emission to soil. As we will show in the next section, if this emission is evenly distributed over all soil the concentration increases due to this source alone might be limited. It might also be that most of this emission is to water rather than to soil. Nevertheless, it is likely that these emissions take place mainly in urban areas, and in that case it cannot be entirely excluded that on a local scale this source might still make relevant contributions to concentrations in soil.

As for water, concentrations tend to be not much higher than a few µg per litre at most (Janus, 1999). For the Rhine and Meuse, concentrations have declined from some 20-30 µg/l in the mid-seventies to below 10 µg/l in the mid-eighties. This is below the Dutch Maximum Tolerable Risk Level (MTR) of 11 µg/l, which is in line with the lowest

NOECs for fresh and salt water organisms given in Chapter 2. Hence, it seems that for water, apart from hot spots and specific local emission situations, environmental risk levels are not exceeded. This conclusion corresponds with the suggestion of CSTEE that 'current levels do not indicate heavy burden of lead for the aquatic environment.

4.6 Deliberation on long-term threats

4.6.1 *Introduction*

As indicated in the introduction to this chapter, in the case of lead a strictly risk-based approach might overlook some potential concerns. In principle a dynamic substance flow analysis has to be carried out to give an answer to questions such as: will soil be enriched with lead in the future? Are there feedback loops that countervail the current declining trend of lead concentrations in food? Are there outlets from the economic system that currently give no high emissions, but might do so in future? This study could not be set up as such a dynamic SFA, but there is some literature available that might give some insight into these problems. The approach we used was as follows:

1. A general figure giving the relation between the different emission sources and final exposure situations was drawn.
2. On the basis of literature sources, a rough and sketchy idea was generated about the quantitative relations between sources and exposure .
3. Making use of the information generated under 2) and 3), a qualitative evaluation was performed to consider whether, under the emission- and use pattern that can be expected in the next decades (see Chapter 3), there is a possibility that via accumulation of lead in certain media exposure limits might be exceeded in the long-term future.

4.6.2 *Lead flows in economy and environment*

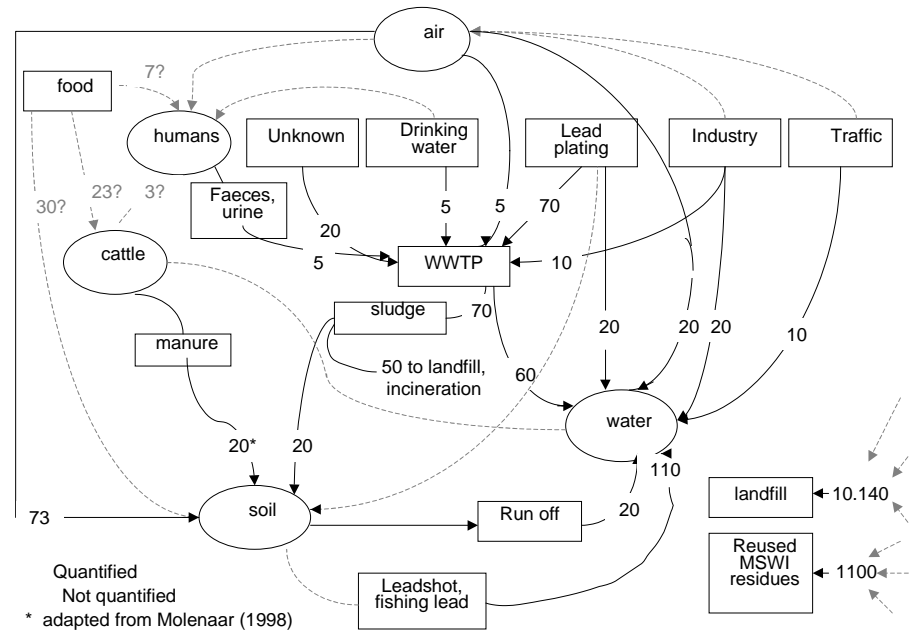
Figure 4.2 gives an overview of lead flows in the economy and the environment in the Netherlands. This is one of the few countries where an SFA has been produced covering both economic and ecological flows and stocks (Annema et al, 1995). This report does not give all flows, but with some additional literature sources rough estimates for the whole system could be made.

Obviously, these data have to be used with care. The lead system in the Netherlands on some points might differ considerably from other countries, or the EU as a whole. Furthermore, the analysis was performed for 1990 and certain flows have probably changed greatly³².

Figure 4.2 combines data from figures 5.1, 5.2 and 5.3 in the Annema (1995) report. Several relations could not be established , such as the emissions from secondary building materials and landfills to soil, and the contamination from foodstuff via soil.

³² For instance, in the Netherlands quite some lead was landfilled with jarosite waste, a by-product from zinc production. With the closure of the plant in question this flow has stopped. With the increase of incineration of waste in the Netherlands, the flow of lead to landfill has further decreased, but the flow of lead which is contaminant in secondary building material has increased, since MSWI slags are re-used in the Netherlands.

Figure 4.2: Substance flows of lead in the Netherlands (in tpa)



The figure shows that various sources contribute to flows to waste water treatment plants (WWTP). These are human faeces (5 tpa), traffic (5 tpa) drinking water (5 tpa), deposition from air (5 tpa), industrial sources (10 tpa) and particularly corrosion from lead plates (70 tpa). The output is partly to water (60 tpa) and mainly to sewage sludge. According to this data, corrosion from leaded building materials is the main source. The more recent report of Janus et al (1999) comes to the same conclusion. The lower influx via air and drinking water in future has relatively little influence on the total inflow. This suggests that also sewage sludge will continue to be contaminated with lead in future, and that no major concentration decline can be expected.

The inflow to water is mainly due to influx from WWTPs, and hence indirectly from corrosion of lead sheet. Lead sheet also contributes directly with 20 tpa to water, as do industrial sources and deposition from air. Traffic contributes another 10 tpa. In the mass balance exercise of Annema et al (1995) emissions related to the use of lead shot and fishing lead are mainly counted as emissions to water. The phasing out of lead for fishing might have made an important improvement in this area.

As for emissions to soil, the situation is not very clear from the Annema (1995) study. Some 20 tpa is emitted to soil due to the use of sludge as fertiliser. However, no data are given for the emission via the use of manure, the use of compost or deposition, to name a few. According to Janus et al (1999), in 1995 the deposition was about 73 tpa. According to our own analysis for the EU, the input with manure is almost equal to that via sewage sludge, leading to a postulated 20 tpa via manure. Molenaar (1998) has made an analysis of a specific trial farm. He arrived at an inflow of 38.7 g/ha/yr via manure, sewage sludge and fertiliser and 33.1 g/ha/yr via deposition, totalling 71.8 g/ha/yr. The outflow was 9.1 g/ha/yr via crops, 14.8 via leaching (as run-off and via

ground water), leading to a net enrichment of 47.9 g/ha/yr. Assuming a homogeneous enrichment of the top 25 cm of soil this would imply a net enrichment of some 0.02 mg/kg lead in soil, compared to a natural background of 10 to 30 mg/kg some 0.2-0.06 % per year. The Danish EPA (1998) calculated, making use of some different assumptions (much higher deposition from air and much lower influx via manure/fertiliser), similar accumulation figures of 32-45 g/ha/yr. From our own assessment of the total emission to soil in Europe, another, rough figure can be derived. Table 3.31 suggests an emission of at least 12 ktpa to soil in Europe, excluding atmospheric deposition. With Europe's surface area of some 10 million km², assuming this is accumulated in the top layer of 25 cm this leads to a concentration increase of some 0.048 mg/kg/yr, or some 0.5-0.2 % of a background of 10-30 mg/kg soil³³. This implies *on average* a doubling of the lead concentration in European soil in 200 to 500 years. However, for grassland accumulation mainly takes place in the upper 5 cm of the ground (Janus et al, 1999). Hence, in theory the top layer of grassland might increase its lead content by 10 mg/kg every 40 years (implying a doubling of concentration of 'clean' soil of 10 mg/kg in the same period).

Another rather striking point is that the accumulation in landfills and re-used MSWI-residues is quite high, probably 1-2 orders of magnitude higher than the emissions to soil. No data are available for the emission that this causes to water and/or soil.

It is not clear what the relation is between soil, crops and intake by humans and cattle. Our own attempt in table 3.29 gives some clues as to what the relative mass flows could be at EU level, but these might be rather rough³⁴. If we apply these relative values, assuming 20 tpa is introduced with manure the food system would end up with an influx of 23 tpa to cattle, of which 3 with meat and milk to humans, 7 tpa with other crops, and some 30 tpa extraction with crops from soil. Moolenaar (1998) calculated for a single farm that some 10 % of the lead input to the farm land (via all pathways) ended up in the crops, which is considerably lower than given in this mass flow calculation.

It has to be stressed that all the above values are highly tentative, particularly with regard to the food circle.

4.6.3 *Interpretation and conclusions*

What can we learn from section 4.6.2 about potential future risks? For this, one has to recognise that a number of exposure pathways will most probably be effectively blocked in future. Emissions to air, and hence deposition from air, will diminish quite significantly in relation to figure 4.1. Furthermore, several direct exposure routes relevant for humans will be eliminated (via drinking water transported through leaded pipes and food cans). Yet, as can be deduced from table 3.31 and 3.32, after the main success has been reached by reducing the use of leaded petrol to zero, the overall emission pattern will be remarkably stable. There are a number of reasons for this :

³³ Ten million km² and 0,25 m depth gives 25 10+13 kg soil. In mg, 12.000 tpa equals 1,2 10+13 mg. Dividing the two values gives the estimate above.

³⁴ For instance, table 3.29 suggests an output from agricultural soil of 226 ton lead/yr to food crops. Assuming that these are all consumed by humans in the EU, with 400 million people, 365 days/yr, and 60 kg b.w. a potential intake of several dozen µg/kg b.w. could be calculated. This is much higher than the 1 µg/kg b.w. or less calculated in table 4.4. on the basis of market basket surveys and duplicate diet studies. It could be that much of the food crops consist of non-edible parts, that refining processes in the food industry diminish lead content of food, but also that the 226 tpa is an over-estimation. At the same time, Annema et al. (1995) mention a value of 80 tpa in meat and crops for the Netherlands only.

1. The exposure routes directly relevant for humans for which measures have been taken (water, food cans etc.) do not contribute significantly to the overall mass flows in figure 4.1.
2. The main emission sources (after leaded petrol) are all related to “durable” applications of lead. Both for lead plating (in the economic system) and for lead shot and fishing lead (in the ecological system) stocks of metallic lead have been formed as a result of the use of these applications over the decades. The current emissions, mainly to soil and water, are the result of slow corrosion of lead and will continue for a considerable time to come even if the application of these uses is stopped immediately.

How can we ensure that the food generation system, as shown on the left in figure 4.1, will not be enriched with lead? Measures like minimising the use of fertilisers likely to be contaminated with lead, like sewage sludge and compost, might be advisable. Furthermore, table 3.30 and 3.31 suggest very strongly that lead shot is a very important contributor to emissions to soil, and given its accumulative character an irreversible one. As indicated before, however, the accumulation in soil is relatively slow, leading to a doubling of concentrations in a time frame of a hundred to several hundred years, depending on the type of soil³⁵. It is mainly a matter of policy, rather than science, to decide whether this should be stopped from a precautionary perspective or can continue since (current) TDIs seem only to be breached in relatively highly contaminated areas.

A number of other issues deserve attention, however. First, corrosion from lead plating is the main contributor to lead pollution of sewage sludge. Lead contamination is one of the reasons why sewage sludge in quite some EU member states is not applied as fertiliser anymore, but has to be treated. A preventive policy to clean sewage sludge inevitably has to address the emissions from the use of lead as plating material in the building sector.

Second, the flows in the waste stage need discussion. It is again a matter of policy rather than science to decide whether a relatively high accumulation of lead in landfills is desirable. One could state that modern controlled landfills presently have such low emissions (compare section 3.13) that it is unlikely that they pose risks; others argue that one should strive for prevention of landfilling of hazardous materials since it can never be guaranteed that the present quality of landfills will be maintained in the very long term. There is more concern about the residual flows from waste incineration. These are voluminous flows like MSWI fly ash and slags, and several EU member states prefer to re-use these materials in road building and other building products. This might lead to a slow enrichment of the building sector with lead, and the long-term consequences in terms of emissions to water and soil cannot be evaluated within the confines of this study, but are an issue in the sustainable management of lead flow. Again, it is a matter of policy rather than science to decide whether one desires relatively lead-free residues (implying close to 100 % recycling, or diminishing lead use in products for which this is not achievable), or that whether accepts them along with the related long-term uncertainties.

³⁵ In grass land accumulation finds place up to a lower depth (5 cm) as in other land (25 cm); compare Janus et al., 1999.

5 Overall summary and conclusions

In this study, an assessment has been performed of the toxicological risks of lead to human health and environment. To this end, various comparisons of exposures and applicable toxicological limit values have been performed.

For the *environment*, apart from hot spots concentrations in water and soil seem to pose no ecotoxicological problem. A question mark is formed by emissions from lead plating; they might be concentrated in urban areas so that a local increase in the concentration of lead in soil cannot be ruled out..

For *workers*, exposures in various lead using and producing industries have been compared with the OELs, which vary from 0.05 mg/m³ (in the US and Norway) to 0.1 mg/m³ (e.g. Germany, the Netherlands, Denmark and Japan) and 0.15 mg/m³ (e.g. France, Spain and the UK). This comparison showed that in practice these standards might be breached in several individual cases. However, particularly those cases where average and minimum data are available suggest that these breaches are mainly related to plants that have not taken adequate worker protection measures. In any case, neither this comparison nor the literature cited give the impression that lead exposure of workers is a structural problem that cannot be dealt with adequately with a traditional, stringent health and safety approach. For this study, this implies that production processes where workers may be exposed to lead have to be monitored carefully. In cases where no adequate protection measures have been taken, lead exposure of workers can be critical. Strict enforcement of the applicable standards is necessary in order to avoid risks to the working population.

For the *general public*, exposures have been compared with the most authoritative limit value, i.e. the TDI of 3.6 µg/day/kg b.w. Furthermore, a rather stringent standard for lead in blood levels of 10 µg/dl has been used in the evaluation. As for public living close to lead using and producing facilities, it became clear that under poorly controlled conditions they can be at risk. Though this conclusion has been based solely on data related to exposure via the air, pathways like exposure via inhalation or ingestion of dust and via crops grown on land close to the production process in question can also be critical if emission management is poor. Hence, for this group the conclusion here should be very similar to the conclusion drawn for worker exposure. Production processes where the general public may be exposed to lead have to be monitored carefully. In cases where no adequate protection measures have been taken, lead exposure of the public living close to the factory can be critical. Strict enforcement of the applicable standards is necessary in order to avoid risks to the general population living close to lead using and producing facilities.

Concerning the more indirect exposure of the general public via food, air, and water, the result of the analysis is shown in the table below.

Table 5.1: Contribution of intake routes to daily intakes of lead

	Concentration	Unit	Intake volume	Unit	Total (μg)	$\mu\text{g}/\text{kg}$ b.w.	% of TDI (3.6 $\mu\text{g}/\text{kg}$ b.w./day)
Adults (70 kg)							
Air	0.01	$\mu\text{g}/\text{m}^3$	20	m^3	0.2	0.003	0.1%
Drinking water	2.6	$\mu\text{g}/\text{l}$	2	l	5.2	0.080	2.2%
Food					27	0.386	10.7%
Soil/dust			0				
Total						0.469	13.0%
Children (10 kg)							
Air	0.01	$\mu\text{g}/\text{m}^3$	10	m^3	0.1	0.01	0.3%
Drinking water	2.6	$\mu\text{g}/\text{l}$	1	l	2.6	0.26	7.8%
Food					8	0.8	22.2%
<i>Subtotal</i>						<i>1.09</i>	<i>30.3%</i>
Soil/dust	100	mg/kg	0.2	g	20	2	55.6%
Total						3.07	85.8%

From the assessment it follows that adults in general will have an intake well below the TDI (a factor of about 10). Blood levels in general are below 10 $\mu\text{g}/\text{dl}$. Only in the case of very high pollution of drinking water (e.g. levels of 100 $\mu\text{g}/\text{l}$ or more) might the TDI be exceeded. Furthermore, under certain local and specific circumstances (a diet of foodstuffs from relatively contaminated soil, exposure due to living near lead using processes, etc.) risk limits might be exceeded. Hence for adults, in principle there seems at present *no need for further information and/or testing and no need for risk reduction measures beyond those already being applied*. In countries where key measures like diminishing the use of leaded petrol, banning lead drinking water pipes, and banning lead from materials in contact with foodstuffs have not yet been taken, *there is a need for limiting the risks taking into account those risk reduction measures which are already being applied*.

For young children, however, the situation is much more difficult to assess clearly. The main reason is that the main intake for young children is ingestion of dust/soil, and the assumptions made for the concentration of lead in this dust and the amount ingested determine the outcome. From the table it is obvious that using just slightly worse than the moderate assumptions applied the TDI for young children will be clearly exceeded. The intake via dust/soil dominates and assumptions about intake via food are less relevant. We feel that it is quite difficult to come to a clear-cut, final conclusion as to whether young children might be at risk or not. First, the calculations and assumptions made for ingestion of soil/dust by children are fraught with uncertainty, and there might be major differences in behaviour between different populations. Second, blood levels of lead in children seem in most cases to be below 10 $\mu\text{g}/\text{dl}$. Hence, an important question is whether exceeding the TDI due to ingestion of soil will genuinely lead to elevated blood levels. But it is precisely on this point that there is quite some debate about the exact situation, since absorption in the body of lead in soil seems less than the absorption of lead from other sources (ATSDR, 1999; compare also: CSTEE, 2000). Third, (potential) exceeding of the TDI most probably occurs during a limited period of a few years, whereas the TDI in principle is meant to give protection over a lifetime. In sum, this makes it quite difficult to make a final statement. Exceeding the TDI might occur regularly, but whether this leads to elevated blood levels is not clear. We

conclude that for children, *there is a need for further information and/or testing*. The questions that arise are : a) the relation between ingestion of different types and quantities of soil/dust and blood levels; b) the quantities likely to be ingested by hand-mouth behaviour; and c) which blood level value (10 µg/dl or lower/higher) should be taken as a reference. In countries where key measures like reducing the use of leaded petrol, banning lead drinking water pipes and banning lead from materials in contact with foodstuffs have not yet been taken, *there is a need for limiting the risks taking into account those risk reduction measures which are already being applied*.

It has to be stressed, however, that this purely risk-based approach leaves several questions about sustainable lead management unanswered. It gives a ‘picture’ at a given moment in time. As we showed throughout the report, the system of substance flows related to lead is quite dynamic. Hence exposure situations in the past might not be valid for the present, and those encountered in the present might not be valid in the future. Where for many degradable substances one can easily assume that reducing overall emissions from present sources will ensure a steadily lower trend in exposure in the future, for a non-degradable material like lead the situation is much more complicated. For instance, exposure reduction measures in the past tended to be concentrated on sources that lead to rather direct exposure to lead (emissions to air via leaded petrol, contamination of water due to the use of lead pipes and solder in the water transport system, and reducing the use of lead solder for food cans). This resulted in a relatively rapid decline in the (semi-direct) human exposure to lead. However, it is also clear that the *use* and hence the economic *stock* of lead is growing, as is the stock in landfills and certain residues from MSWIs (which might be re-used). This led to a discussion driven by the fear for slow ‘enrichment’ of the economic and ecological system with lead, and hence a possible higher exposure in future.

In principle a dynamic substance flow analysis has to be carried out to answer questions such as: will soil be enriched with lead in the future to levels which cannot be accepted? Are there feed-back loops that countervail the current declining trend of lead concentrations in food? Are there outlets from the economic system that currently give no high emissions, but might do so in future? This study could not be set up as such a dynamic SFA, but via a literature analysis some impression about these trends and relations between lead flows could be given. Some of the main features to come out of this analysis were:

1. After the main success has been reached by reducing the use of leaded petrol to zero, the overall emission pattern will be remarkably stable in the next decades. There are a number of reasons for this:
 - The exposure routes directly relevant for humans for which measures have been taken (water, food cans etc.) do not contribute significantly to the overall mass flows in the economy and the environment.
 - The main emission sources (after leaded petrol) are all related to ‘durable’ applications of lead. Both for lead plating (in the economic system) and for lead shot and fishing lead (in the ecological system) stocks of metallic lead have been formed as a result of the use of these applications over the decades. The current emissions, mainly to soil and water, are the result of slow corrosion of lead and will continue for a considerable time to come even if the application of these uses is stopped immediately.

2. Should the suggested analysis of feedback loops lead to a need or desire to diminish the lead inflow into the food generation system, this can be achieved as follows. Measures like minimising the use of fertilisers likely to be contaminated with lead, like sewage sludge and compost, might be advisable. Lead shot seems a very important contributor to the emissions to soil, and given its accumulative character an irreversible one.
3. It has been calculated that the average accumulation of lead is related to various uncertainties. For the EU on average, doubling of background concentrations in the top soil layer of 10-30 mg/kg might take place in a time frame of several hundred years. However, since for grassland the mixing zone is much smaller (the top 5 cm), here a 10 m/kg concentration might double in 40 years. In this calculation deposition from air has already been ignored. Ammunition and corrosion of lead sheet are probably the main contributors. It is mainly a matter of policy rather than of science to decide whether these emissions should be stopped from precautionary perspective or can continue since (current) TDIs seem only to be breached in relatively highly contaminated areas. In any case, research to obtain an insight into the actual emissions caused by lead shot and lead sheet are recommended, in view of the rather large uncertainties in emission factors.
4. Corrosion from lead plating makes the main contribution to lead pollution of sewage sludge. Lead contamination is one of the reasons why sewage sludge is no longer used as fertiliser in quite a number of EU member states, but has to be treated or landfilled. Should a preventive policy to clean sewage sludge be seen as desirable, the emissions from the use of lead as plating material in the building sector should inevitably be addressed. This source is also the main cause of emissions to water.
5. The flows in the waste stage need attention. It is again a matter of policy rather than science to decide whether a relatively high accumulation of lead in landfills is desirable. The mass flows are 1-2 orders of magnitude higher than emissions to soil. One could state that modern controlled landfills presently have such low emissions (compare section 3.13) that it is unlikely that they pose risks; others argue that one should strive for prevention of landfilling of hazardous materials since it can never be guaranteed that the present quality of landfills will be maintained in the very long term. Of more concern are the residual flows from waste incineration. These are voluminous flows like MSWI fly ash and slags, and several EU member states prefer to re-use these materials in road building and other building products. This might lead to a slow enrichment of the building sector with lead, and the long-term consequences in terms of emissions to water and soil cannot be evaluated within the confines of this study, but are an issue in the sustainable management of lead flow. Again, it is a matter of policy rather than science to decide whether one desires relatively lead-free final waste residues (implying close to 100 % recycling, or diminishing lead use in products for which this is not achievable), or one accepts them, along with the related long-term uncertainties.

6 Literature

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Annex 1: Detailed calculations of mass flows

Annex 1.1: Batteries

Annex 1.2: Pigments and other compounds

Annex 1.3: Alloys

Annex 1.1: Batteries

Batteries (ktonnes Pb)	production			consumption			waste treatment remains			emission		stock	emission
	emission	inflow Pb	outflow Pb	emission	in environ.	recycling 90-95%	landfill	incineration		on landfill	from landfill	from landfill	
								8%	2%				
Total batteries								7%	3%			0.008% corrosion	
1965													
1970	p.m.	446	0	0	0	0	0	0	0	p.m.	0	0.000	
1975	p.m.	476	410	0	0	390	16	4	p.m.	103	0.002		
1980	p.m.	573	474	0	0	453	18	4	p.m.	212	0.004		
1985	p.m.	694	568	0	0	541	21	5	p.m.	345	0.006		
1990	p.m.	800	682	0	0	650	26	6	p.m.	505	0.009		
1995	p.m.	869	793	0	0	755	30	7	p.m.	692	0.013		
2000	0.21	919	863	0	0	823	32	8	0.03	894	0.016		
2005	0.21	947	915	0	0	873	33	9	0.03	1108	0.020		
2010	0.22	975	945	0	0	901	34	10	0.04	1328	0.024		
2015	0.22	1005	973	0	0	928	34	11	0.04	1555	0.029		
2020	0.23	1035	1002	0	0	956	34	13	0.05	1789	0.033		
2025	0.24	1066	1032	0	0	984	34	14	0.05	2029	0.037		
2030	0.25	1098	1063	0	0	1014	35	15	0.06	2277	0.042		
(ktonnes Pb) SLI		consumption			waste treatment remains			incineration		stock	leakage		
		inflow of Pb	outflow Pb	emission	in environ.	recycling	landfill			on landfill	from landfill		
					0	95		4	1 [2000]		0.23 %		
1965								3.5	1.5 [2030]		0.008 % corrosion		
1970		353											
1975		377	353	0	0	336	14	4		88	0.002		
1980		453	377	0	0	359	15	4		183	0.003		
1985		540	453	0	0	430	18	5		296	0.005		
1990		621	540	0	0	513	22	5		431	0.008		
1995		674	621	0	0	590	25	6		586	0.011		
2000		713	674	0	0	640	27	7		754	0.014		
2005		734	713	0	0	677	28	8		933	0.017		
2010		756	734	0	0	698	28	9		1116	0.021		
2015		779	756	0	0	719	28	9		1305	0.024		
2020		802	779	0	0	740	29	10		1500	0.028		
2025		827	802	0	0	762	29	11		1701	0.031		
2030		851	827	0	0	785	29	12		1907	0.035		
(ktonnes Pb) Motive power batteries		consumption			waste treatment remains			incineration		stock	leakage		
		inflow of Pb	outflow Pb	emission	in environ.	recycling	landfill			on landfill	from landfill		
					0	95		4	1 [2000]		0.23 %		
1965								3.5	1.5 [2030]		0.008 % corrosion		
1970		57											
1975		60	57	0	0	54	2	1		14	0.000		
1980		77	60	0	0	57	2	1		29	0.001		
1985		96	77	0	0	73	3	1		48	0.001		
1990		111	96	0	0	91	4	1		72	0.001		
1995		120	111	0	0	105	4	1		100	0.002		
2000		127	120	0	0	114	5	1		130	0.002		
2005		131	127	0	0	121	5	1		162	0.003		
2010		135	131	0	0	124	5	2		195	0.004		
2015		139	135	0	0	128	5	2		228	0.004		
2020		143	139	0	0	132	5	2		263	0.005		
2025		147	143	0	0	136	5	2		299	0.005		
2030		152	147	0	0	140	5	2		336	0.006		
(ktonnes Pb) Industrial batteries		consumption			waste treatment remains			incineration		stock	leakage		
		inflow of Pb	outflow Pb	emission	in environ.	recycling	landfill			on landfill	from landfill		
					0	100		0	0		0.23 %		
1965											0.008 % corrosion		
1970		36											
1975		38											
1980		43	36	0	0	36	0	0		0	0.000		
1985		55	38	0	0	38	0	0		0	0.000		
1990		62	43	0	0	43	0	0		0	0.000		
1995		68	55	0	0	55	0	0		0	0.000		
2000		72	62	0	0	62	0	0		0	0.000		

Annex 1.2: Pigments, stabilisers, and similar compounds

Total pigments (ktonnes Pb)	production/consumption			waste treatment remains			emission		stock	leakage	stock	leakage		
	emission	inflow Pb	outflow Pb	emission	in environ.	recycling	landfill	incineration	incineration	on landfill	on landfill	in environ.	from stock	in environ.
1965														
1970		183	2	0	0	0	1	0		8	0.00	0	0	0
1975		178	2	0	0	0	1	0		16	0.01	0	0	0
1980		221	2	0	0	0	2	0		28	0.01	0	0	0
1985		202	104	0	0	0	83	21		545	0.09	0	0	0
1990		210	148	0	0	0	119	30		1288	0.16	0	0	0
1995		211	167	0	0	0	134	33		2123	0.24	0	0	0
2000	0.04	201	201	0	16	8	142	35	0,04	3010	0.31	80	0	0
2005	0.04	191	195	0	16	8	134	37	0,03	3865	0.37	160	0	0
2010	0.03	134	194	0	16	50	98	30	0,03	4503	0.43	240	0	0
2015	0.03	124	188	0	16	48	93	31	0,04	5125	0.48	320	0	0
2020	0.03	114	176	0	16	43	86	31	0,04	5711	0.53	400	0	0
2025	0.02	104	167	0	17	38	81	32	0,04	6274	0.59	482	0	0
2030	0.02	94	170	0	23	37	77	33	0,05	6827	0.65	597	0	0
Glass: pigments and other compounds														
(ktonnes Pb)	consumption			waste treatment remains			incineration		stock	leakage				
<i>cathode ray tube</i>	Inflow Pb	outflow Pb	emission	in environ.	recycling	landfill	incineration	incineration	on landfill	from landfill	0 %			
				0	0	80	20							
					50	70	30							
1965														
1970		37		0	0	0	0	0		0	0.00			
1975		41		0	0	0	0	0		0	0.00			
1980		58		0	0	0	0	0		0	0.00			
1985		72	37	0	0	0	29	7		183	0.00			
1990		84	41	0	0	0	32	8		386	0.00			
1995		84	58	0	0	0	46	12		676	0.00			
2000		80	72	0	0	0	58	14		1038	0.00			
2005		70	84	0	0	0	66	18		1458	0.00			
2010		60	84	0	0	42	32	10		1669	0.00			
2015		50	80	0	0	40	30	10		1869	0.00			
2020		40	70	0	0	35	26	9		2045	0.00			
2025		30	60	0	0	30	22	9		2195	0.00			
2030		20	50	0	0	25	18	8		2321	0.00			
Glass: other pigments/comp.														
(ktonnes Pb)	consumption			waste treatment remains			incineration		stock	leakage				
<i>(crystal, light bulbs, speciality glass/optical)</i>	Inflow Pb	outflow Pb	emission	in environ.	recycling	landfill	incineration	incineration	on landfill	from landfill	0 %			
				0	0	80	20 [2000]							
						70	30 [2030]							
1965														
1970		48		0	0	0	0	0		0	0.00			
1975		45		0	0	0	0	0		0	0.00			
1980		53		0	0	0	0	0		0	0.00			
1985		45		0	0	0	0	0		0	0.00			
1990		46	48	0	0	0	39	10		241	0.00			
1995		46	45	0	0	0	36	9		467	0.00			
2000		44	53	0	0	0	43	11		734	0.00			
2005		44	45	0	0	0	35	10		957	0.00			
2010		44	46	0	0	0	35	11		1186	0.00			
2015		44	46	0	0	0	34	11		1415	0.00			
2020		44	44	0	0	0	32	12		1633	0.00			
2025		44	44	0	0	0	31	12		1852	0.00			
2030		44	44	0	0	0	31	13		2070	0.00			
stabilizer in plastics (PVC)														
(ktonnes Pb)	consumption			waste treatment remains			incineration		stock	leakage	stock	leakage		
<i>(profiles, pipes, cables)</i>	Inflow Pb	outflow Pb	emission	in environ.	recycling	landfill	incineration	incineration	on landfill	from landfill	in environ.	from stock	in environ.	0 %
				50	25	20	5 [2000]				0,046			
						17,5	7,5 [2030]							
1960		32												
1965		32												
1970		32		0	0	0	0	0		0	0.00	0	0.00	
1975		33		0	0	0	0	0		0	0.00	0	0.00	
1980		46		0	0	0	0	0		0	0.00	0	0.00	
1985		41		0	0	0	0	0		0	0.00	0	0.00	
1990		46		0	0	0	0	0		0	0.00	0	0.00	
1995		46		0	0	0	0	0		0	0.00	0	0.00	
2000		44	32	0	16	8	6	2		40	0.02	80	0.00	
2005		44	32	0	16	8	6	2		80	0.04	160	0.00	

Annex 1.2 (continued)

(ktonnes Pb)	consumption			waste treatment remains				incineration	stock on landfill	leakage from landfill	stock in environ	leakage from stock in environ
	inflow Pb	outflow Pb	emission	in environ.	recycling	landfill	20 [2000]					
stabilizer in plastics (PVC) foils and films				0	0	0	80	20 [2000]		0,046		0
1960							70	30 [2030]				
1965	2	2										
1970	2	2	0	0	0	1	0		8	0,00	0	0,00
1975	2	2	0	0	0	1	0		16	0,01	0	0,00
1980	2	2	0	0	0	2	0		28	0,01	0	0,00
1985	2	2	0	0	0	2	0		38	0,02	0	0,00
1990	2	2	0	0	0	2	0		50	0,02	0	0,00
1995	2	2	0	0	0	2	0		62	0,03	0	0,00
2000	2	2	0	0	0	2	0		73	0,03	0	0,00
2005	2	2	0	0	0	2	1		85	0,04	0	0,00
2010	0	0	0	0	0	0	0		85	0,04	0	0,00
2015	0	0	0	0	0	0	0		85	0,04	0	0,00
2020	0	0	0	0	0	0	0		85	0,04	0	0,00
2025	0	0	0	0	0	0	0		85	0,04	0	0,00
2030	0	0	0	0	0	0	0		85	0,04	0	0,00

(ktonnes Pb)	consumption			waste treatment remains				incineration	stock on landfill	leakage from landfill	stock in environ	leakage from stock in environ
	inflow Pb	outflow Pb	emission	in environ.	recycling	landfill	20 [2000]					
other pigments compounds (glazes, paints, ceramics)				0	0	0	80	20 [2000]		0,23		
1965							70	30 [2030]		0,1 % wear		
1970	65		0	0	0	0	0	0	0	0,000	0	0,000
1975	57		0	0	0	0	0	0	0	0,000	0	0,000
1980	61		0	0	0	0	0	0	0	0,000	0	0,000
1985	41	65	0	0	0	52	13		325	0,075		
1990	32	57	0	0	0	46	11		612	0,141		
1995	32	61	0	0	0	49	12		919	0,211		
2000	30	41	0	0	0	33	8		1125	0,259		
2005	30	32	0	0	0	25	7		1284	0,295		
2010	30	32	0	0	0	24	7		1444	0,332		
2015	30	30	0	0	0	23	8		1596	0,367		
2020	30	30	0	0	0	22	8		1748	0,402		
2025	30	30	0	0	0	22	9		1900	0,437		
2030	30	30	0	0	0	21	9		2052	0,472		

Annex 1.3: Alloys

Alloys (ktonnes Pb)	production			consumption			waste treatment				emission from landfill 0.23 % 0.008 % corrosion
	emission	inflow Pb	outflow Pb	emission	in environ.	recycling	landfill	incineration	stock on landfill		
Total alloys					0	0	80	20			
1965				0	0	0	0	0	0	0	0.000
1970		82	0	0	0	0	0	0	0	0	0.000
1975		65	0	0	0	0	0	0	0	0	0.000
1980		56	0	0	0	0	0	0	0	0	0.000
1985		46	82	0	0	63	15	4	94	0.002	
1990		45	65	0	0	49	13	3	174	0.003	
1995		40	56	0	0	40	13	3	254	0.005	
2000	0,01	32	46	0	0	29	13	3	338	0.006	
2005	0,01	32	45	0	0	24	16	4	441	0.008	
2010	0,01	32	40	0	0	22	14	4	535	0.010	
2015	0,01	32	32	0	0	17	11	4	609	0.011	
2020	0,01	32	32	0	0	17	11	4	683	0.013	
2025	0,01	32	32	0	0	17	11	4	758	0.014	
2030	0,01	32	32	0	0	17	10	4	832	0.015	
(ktonnes Pb)											
solder in electronics		inflow Pb	outflow Pb	emission	remains in environ.	recycling	landfill	incineration	stock on landfill	leakage from landfill	0.23 % 0.008 % corrosion
1965				0	0	0	70	30			
1970		14		0	0	0	0	0	0	0	0.000
1975		12		0	0	0	0	0	0	0	0.000
1980		12		0	0	0	0	0	0	0	0.000
1985		14	14	0	0	0	11	3	68	0.001	
1990		18	12	0	0	0	9	2	126	0.002	
1995		16	12	0	0	0	10	2	189	0.003	
2000		13	14	0	0	0	11	3	257	0.005	
2005		13	18	0	0	0	14	4	345	0.006	
2010		13	16	0	0	0	12	4	426	0.008	
2015		13	13	0	0	0	10	3	490	0.009	
2020		13	13	0	0	0	9	3	554	0.010	
2025		13	13	0	0	0	9	4	618	0.011	
2030		13	13	0	0	0	9	4	682	0.013	
(ktonnes Pb)											
other solders (cars, plumbing, food cans)		consumption inflow Pb	outflow Pb	emission	waste treatment remains in environ.	recycling	landfill	incineration	stock on landfill	leakage from landfill	0.23 % 0.008 % corrosion
1965				0	0	100	0	0			
1970		42		0	0	0	0	0	0	0	0.000
1975		32		0	0	0	0	0	0	0	0.000
1980		26		0	0	0	0	0	0	0	0.000
1985		16	42	0	0	42	0	0	0	0	0.000
1990		12	32	0	0	32	0	0	0	0	0.000
1995		11	26	0	0	26	0	0	0	0	0.000
2000		9	16	0	0	16	0	0	0	0	0.000
2005		9	12	0	0	12	0	0	0	0	0.000
2010		9	11	0	0	11	0	0	0	0	0.000
2015		9	9	0	0	9	0	0	0	0	0.000
2020		9	9	0	0	9	0	0	0	0	0.000
2025		9	9	0	0	9	0	0	0	0	0.000
2030		9	9	0	0	9	0	0	0	0	0.000
(ktonnes Pb)											
other alloys (brass, bronze, bearings, bushings, terne plate)		consumption inflow Pb	outflow Pb	emission	waste treatment remains in environ.	recycling	landfill	incineration	stock on landfill	leakage from landfill	0.23 % 0.008 % corrosion
1965				0	0	80	16	4			
1970		26		0	0	0	0	0	0	0	0.000
1975		22		0	0	0	0	0	0	0	0.000
1980		18		0	0	0	0	0	0	0	0.000
1985		16	26	0	0	21	4	1	26	0.000	
1990		14	22	0	0	18	4	1	48	0.001	
1995		13	18	0	0	14	3	1	66	0.001	
2000		10	16	0	0	13	3	1	82	0.002	
2005		10	14	0	0	11	2	1	96	0.002	
2010		10	13	0	0	10	2	1	109	0.002	
2015		10	10	0	0	8	2	1	119	0.002	