MEETING OPTIMALLY THE ENVIRONMENTAL CHALLENGE: A METHODOLOGY FOR THE LEAD INDUSTRY

A thesis submitted for the partial fulfilment of the degree of
Doctor of Engineering in Environmental Technology

by

John Graham Stuart ROBERTSON

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Bringing the project to this stage has been a long and, at times, tortuous experience. It seems a bit of an old cliché to say, "I would never have got his far without the advice and moral support of my friends, colleagues and family", but it is true! Therefore, I feel it is only fitting, I should use this opportunity to thank them all. There are, however, a number of people who deserve a special mention.

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There are also a number of people at my sponsoring company, Britannia Refined Metals Ltd. (BRM), and at the head office of its holding company, MIM Ltd., who deserve a special mention. At BRM, I would like to extend a special thanks to John Wood for being my industrial supervisor during most of my time at the company. I would also like to thank Peter Bryant for taking over that role following John's retirement. In addition, I am greatly indebted to Frank Boyes, Peter Morris, Chris Hayden and Julia Swain for their help and support, during my time at BRM's Environmental Department. In particular, I would like to thank Frank for his role of sounding board to my ideas. Paul Framp (BRM's General Manager) also deserves a special mention, for his support of my project for the duration, and for helping facilitate my data collection trip to Australia. In this context, Rob MacNamara and Nick Lefebvre (Environment Manager and Environmental Supervisor for MIM Ltd. respectively) also deserve a special thanks for their support and funding of my data collection trip to Australia.

The greatest thanks of all, however, need to be extended to my long suffering and often neglected family, who during the last few years, have provided me with emotional and,

latterly, financial support. Without that, I would never have "survived to tell the tale". No words can really ever express how grateful I am to them.

AUTHOR'S DECLARATION

The author certifies that none of the work or documents submitted for the Engineering Doctorate (EngD), in either the EngD Portfolio of submissions or this thesis, have been submitted anywhere else for a degree or for any other type of academic qualification.

John Graham Stuart Robertson
18 December 2000

If many faultes in this book you fynde,
Yet think not the corrections blynde;
If Argos here hymselfe had beene
He should perchance not all have seene.

Richard Shacklock 1565

To my family
For patience, support and enduring encouragement

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ABSTRACT

Does the lead industry have a future, in the face of the developing environmental challenge? This thesis addresses this question and concludes, it should have for the foreseeable future, providing it adopts the changes detailed. These changes are posited within a framework, which consists of a strategy, approaches and tools. The changes are both technical and philosophical. They are technical, in the sense that the tools and approaches provide practical means whereby the environmental 'risks' may be identified, assessed and managed. They are philosophical, because they set out and identify the features of a new conceptual paradigm, whose basis is in the concept of the 'risk society'. The paradigm is significantly more holistic, multi-dimensional, inherently flexible, and is intended to be reflexive. Adoption of the elements of the framework, will facilitate a more effective establishment, and management of environmental 'risk' credentials, which will help encourage better environmental decision making. Hence, it will facilitate, the balancing of resource consumption and environmental impact costs, versus social and economic benefits, in an improved manner. The modelling approaches, and selected inventory and environmental impact assessment tools, enclosed within this thesis, have been designed to facilitate the development of, and to function within, the new paradigm. These have been developed for BRM and MIM case studies, and function at the site-specific and the cradle-to-gate scales. The former consider the company site of Britannia Refined Metals (BRM) Ltd., where refining to produce primary and secondary refined lead products takes place, whilst the latter consider the life-cycle of the refined primary lead products of MIM Ltd. The modelling approaches have also been designed so, that they may be re-aggregated into models able to operate at many different scales, as required. The framework, and its elements, are applicable for all industries facing similar challenges.

LIST OF FIGURES

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LIST OF PUBLICATIONS

Peer Reviewed Journal Papers:

1. Robertson J.G.S., Wood J.R., Fenn R., Ralph B. (1997) "Analysis of Lead-Acid Battery Life-Cycle Factors: Their Impact on Society and the Lead Industry" *Journal of Power Sources* 67; pp. 225 - 236.
2. Cowell S.J., Wehrmeyer W., Argust P.W., Robertson J.G.S. (1999) "Sustainability and the Primary Extraction Industries: Theories and Practice" *Resources Policy* 25; pp. 277-286.

Industry and Academic Conference Papers:

1. Robertson J.G.S., Wood J., Ralph B., Fenn R., "Developing Strategies and Means to Enable the Lead Industry to Meet the Emerging Environmental Challenge" Presentation to the 'Lead into the Future' Conference; 14 to 16 October 1996.
2. Robertson J.G.S., "Meeting the Environmental Challenge: Using Life-Cycle Assessment as a Management Tool in the Lead Industry" 12th International Lead Conference, Salzburg, Austria; 22 to 25 September 1997.
3. Robertson J.G.S. "A Framework for Structuring Environmental Decision Making" SETAC-Europe Conference 1999, Leipzig, Germany; 25 - 29 May 1999. [Abstract and presentation]
4. Robertson J.G.S., "MIM Primary Lead Production: Using a Life-Cycle Approach to Identify and Manage Environmental Effects" Presentation to Conference: 'Developments at Processing Plants in Great Britain and Northern Ireland', The Institution of Mining and Metallurgy (The Southern Counties Branch) with The Minerals Engineering Society, Imperial College, London, England; 18 Oct. 2000. [Abstract and presentation]

Other Relevant Publications:

1. Review of: Wenzel H., Hauschild M., Alting L. (1997) 'Environmental Assessment of Products' Volumes 1 and 2; Chapman & Hall; plus accompanying 'LCV-System PC tool (beta version). In: *Journal of Industrial Ecology*3:4; pp. 179 - 183.

ENGD PROJECT FINAL OUTPUTS:

EXECUTIVE SUMMARY

1. Introduction:

The overall aims of this Executive Summary are:

1. to set the Engineering Doctorate (EngD) Portfolio[1] and this thesis within the context of 'Environmental Technology'.
2. to direct the reader to evidence, in these documents, which demonstrates the innovations and contributions to knowledge in the field of 'Environmental Technology'.

The Brunel/Surrey EngD in Environmental Technology Course handbook for the academic year 2000 - 2001, states that Research Engineers need to:

- be able to plan and execute flexible, innovative research and development programs that respond to customer needs;
- form, work within, and where necessary, lead teams with multidisciplinary backgrounds;
- have expert knowledge in the field of environmental technology, and be able to apply techniques that balance social and economic benefit against resource utilisation and environmental impact;
- possess a working knowledge of project management and business methods; and
- have excellent communication skills.

Hence, this Executive Summary also demonstrates how these additional requirements have been met.

The complete Portfolio and the enclosed thesis together constitute the final overall research outputs.

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[1] The EngD Portfolio contains an extensive range of supporting documentation for this thesis plus evidence that the author has met fully the criteria set out above. It has not been enclosed with this thesis.

2. Setting the Final Outputs in the Context of Environmental Technology:

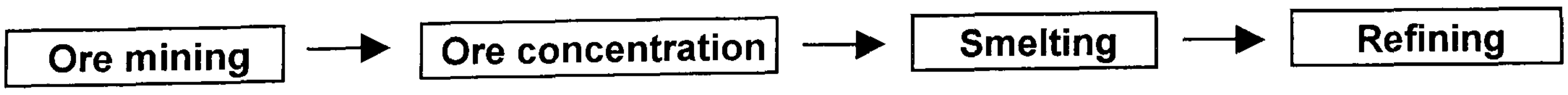
The Brunel/Surrey EngD Course handbook for the academic year 2000 - 2001, states the overall objective of the Brunel/Surrey EngD Program is create graduate Research Engineers "with the necessary background knowledge, skills and experiences to understand the relationship between the environment, technology and business and to apply this understanding to the development and promotion of corporate strategy". In full compliance with this aim, 'Environmental Technology' is defined as the application of techniques that balance social and economic benefits against resource utilisation and environmental impact. The overall objective of the Brunel/Surrey EngD Program has also been the guiding principal for the whole research project and underpins the thesis.

In the thesis, the lead industry is considered to consist of both primary and secondary operations, which in turn consist of both foreground and background processes. Foreground processes, are defined as those under the direct control of lead industry companies, whilst background processes are those non-lead industry processes, involved in:

- the production of ancillary materials and fuels used in lead industry processes (from their own respective origins as raw materials), and
- the disposal of wastes (from their point of production by lead industry processes, to their final disposal).

Lead industry primary operations, produce refined lead and lead alloys from ores in the ground, whilst secondary operations produce refined lead and lead alloys from secondary sources (principally scrap lead-acid batteries). For both primary and secondary operations, all transports of bulk intermediate materials have been treated as foreground processes.

The primary lead production sequence of foreground processes, consists of:



Whilst, the secondary lead production sequence of foreground processes consists of:

For both primary and secondary operations, the activities of the lead industry, are considered to end at the point where refined lead and lead alloys are produced. This is because, the further processing to produce the final manufactured products, is considered to be conducted by other industries. Hence, lead-acid battery production (the principal manufactured product from the refined lead alloy products) is conducted by the battery industry, not the lead industry.

The social benefits of the lead industry, are considered to be associated, principally, with the manufactured product end uses to which its products, by-products and co-products are put. Social benefits are also associated with the employment opportunities, plus associated community support activities (such as encouraging the formation and sustaining of habitations), associated with its foreground and background processing operations. Therefore, the social benefits of the lead industry are associated, in part, with downstream processes conducted by allied industries, such as the lead-acid battery industry, which are not under its direct control.

The economic benefits of the lead industry are both direct and indirect. The direct benefits are from its contribution to both the global economy, plus national and regional economies, arising from the sale of its products, co-products and by-products. With respect to indirect contributions, these are from:

- the sale of the final manufactured products, from the lead industry products, co-products, and by-products, and
- the economic contributions associated with consumer spending, by employees both directly within the lead industry, and indirectly in background plus downstream processes.

The resource consumptions and environmental impacts associated with the lead industry are considered to be both direct and indirect. The former, are associated with consumptions and emissions, respectively, from:

- foreground primary and secondary lead industry processes (including intermediate bulk material transports),
- background processes, for ancillary materials and fuels used in lead industry processes (from their origins as raw materials), and for the disposal of wastes (from production by lead industry processes, to final disposal).

The indirect resource consumptions and environmental impacts, are associated with:

- the production and use of the final manufactured products, from the lead industry products, co-products, and by-products, and
- the eventual recycling or disposal of these final manufactured products, at the end of their useful life.

A further distinction between actual and potential environmental impacts, has been made in the case studies, supplied in the thesis, where the former are either measured or predicted by fate and effect modelling (as described in section 5.1 of chapter 5), whilst the latter are calculated from inventory data (inflow and outflow data for processes and operations).

The industrial sponsor from the outset, has been Britannia Refined Metals Limited (BRM), of Northfleet, Kent, UK. However, additional financial and material support was provided by MIM Holdings Limited, of Brisbane, Queensland, Australia for the modelling of the cradle-to-gate life-cycle of its primary lead products, which is one of the major outputs from the research. BRM is a company operating under the umbrella of MIM, and produces MIM's entire refined lead output. In compliance with the Brunel/Surrey EngD Program overall objective, the aim throughout has been to conduct research whose outputs may make significant contributions to the development and promotion of corporate strategy. Since the principal project sponsor is a major player within the lead industry, the research has been structured to address its needs. However, the innovations and contributions to knowledge resulting from this research, identified in this thesis, are also applicable for industry in general. Whilst the research has been aimed at meeting the needs of BRM, MIM, and the lead industry, the identification of the nature of these needs, and of approaches to enable it to meet them most effectively, have been conducted by the RE independently of the sponsors, though in full consultation with them.

The balancing of social and economic benefits, against resource consumption and environmental impact is normative. The environmental challenge (which is reviewed in chapter 3 for industry in general and the lead industry in particular) is tending to shift the balance, so that similar social and economic benefits are being demanded, with lower resource consumptions and environmental impacts. The overall contribution of the thesis, is the setting out of a framework, containing an integrated strategy, plus specific approaches and tools designed to meet this challenge with optimal effectiveness. (These elements are set out in the thesis in section 4.5 of chapter 4.) The strategy is for the whole of the lead industry, whilst the approaches are company management based, and the tools are specific methods to be adopted within individual businesses. Therefore, all elements of the framework are posited within corporate strategy.

The modelling approaches, and specific models (the tools) developed in the case studies, provide assessments of the resource consumptions and environmental impacts, at various scales of interest, ranging from the individual process, to the whole cradle-to-gate for the MIM primary refined lead life-cycle (i.e. all stages from raw material extraction, through to the production of the final refined product). The assessments, identify explicitly, the relationship between the technologies and techniques in the processes and/or operations (which are themselves aggregates of different processes), and their respective resource consumptions and environmental impacts. The thesis also considers how the integrated strategy, approaches and tools, may be used in decision making, and argues for the need for effective decision making approaches (section 4.5 of chapter 4). It concludes, finally, (in chapter 11), that the key to ensuring a proper balance between the elements is 'reflexive dialogue', which it defines as a decision making context where all participants are required "to reflect critically upon, and to challenge, in a constructive manner, their own preconceptions throughout the process". However, it points out that currently this is still an aspiration. Without such dialogue, the framework will tend to foster 'better' decision making, rather than 'best' decision making.

In considering these questions, the thesis addresses, explicitly, therefore, the relations between technology, the environment, corporate strategy, and the balancing of social and economic benefits. Hence, it is concluded, the strategy, approaches and tools, developed in the thesis, are all elements of 'environmental technology.

3. Evidence of Innovations in the Field of 'Environmental Technology'

'Innovations' are defined as the introduction of something new. In the field of environmental technology, therefore, these are considered to be the introduction new techniques, which may be used in the balancing of social and economic benefits against resource utilisation and environmental impact. The major innovations, meeting these criteria, are provided by: the framework, modelling, results interpretation approach, environmental performance indicators and the BRM whole site modelling approach. These are reviewed in turn.

a) Framework:

The main elements of the framework are summarised in section 4.5 of chapter 4. It is an innovation, because it facilitates the more effective establishment, management and communication of environmental 'risk' credentials (in the manner defined in section 4.3). As a result, it helps encourage better environmental decision making, and thereby facilitates the balancing of costs and benefits in an improved manner. It also identifies the need for reflexive dialogue, if 'best' decision making is to occur.

b) Modelling:

Two case studies have been enclosed, the BRM and the MIM case study. Each is self-contained, and will, on termination of the project, be distributed to BRM and MIM as separate reports. Each case study contains two main models, as follows:

- *BRM case study:* 'BRM Whole Site' and 'BRM: Secondary Refined Lead Production Operations'
- *MIM case study:* 'MIM Pb LCA (allocated)' and 'MIM Pb LCA (unallocated)'

In the BRM case study, 'BRM: Secondary Refined Lead Production Operations' is a separate model in its own right, even though the 'BRM Whole Site' model has secondary operations modelling subsumed within it. This is because both primary and secondary operations take place at the BRM site. Versions of both models containing two different years of data have also been produced. In the MIM case study, however, the input data and model structures for both are similar. The only significant difference between them, is the use of allocation factors in the former.

For each of the models, data for all of the foreground inputs and outputs have been collected, as a result of site visits to the processing facilities. Every foreground system process, has been treated as a separate node, including the transports of ancillary materials, fuels and wastes (which cross between the foreground and the background systems). The raw data have been entered into Excel spreadsheets, developed specifically for each process modelled. For each non-transport process, a completely separate set of spreadsheets have been created. For transport processes, however, all of those associated with operations, at each specific site, have been calculated together in separate sets of spreadsheets. The complete set of sheets are in the Portfolio as Volume 3. This contains about 400 pages of raw input/output data, modelling information, calculations and normalised data, for 41 non-transport and 144 transport processes. The complete set of Excel files, from which the sheets in the Portfolio have been printed, will be supplied to the project sponsors, on final termination of the project.

The sheets have been designed, so that when raw data are entered into the 'Quantity per Year of Production' column in the non-transport process sheets, and into the 'Transport Details' columns in the transport data sheets, the value normalised per functional unit of output from that process is calculated automatically, and displayed in the 'Data Normalised per Functional Unit of Output' column, in the normalised set of data sheets. (The functional unit of output for each process, is indicated at the top of each of the normalised data sheets. Generally, it is 1000 kg of the main product of the process.) The normalised data sheets, also indicate the raw data value, the name they are called in the modelling (i.e.

'Article Name'), their linkage details in the modelling, and any other comments which may be necessary. In both sets of raw data sheets, where data have been available to do so, simple statistics describing the 'Variation of the Data per Period', have been indicated. Where applicable, information regarding the name and location of the suppliers of ancillary materials have also been noted on these sheets.

In some cases, the data in the raw data sheets have had to be calculated, before they could be entered into the cells. Where this is the case, it has been made obvious in the Excel sheets by the use of a red font for the data, and calculations are conducted in the appropriate referenced and linked 'Note'. However, red font data are not distinguishable from normal black font data, in the printed hard copies in the Portfolio. Nevertheless, the presence of such calculated data may still be identified by the referenced 'Note'.

The raw data sheets, for both types of processes, also contain columns entitled 'Characterisation of Data'. The criteria for the values in these columns, are provided in Document 18 (of volume 5 of the Portfolio). These columns enable the quality of all data entries to be tracked.

All of the innovations described above, ensure that updating the sheets with data in future iterations will be straightforward. This is because, only the 'Quantity per Year of Production' or 'Transport Details', plus the 'Characterisation of Data', and 'Variation of Data per Period' columns in the raw data sheets, will need to be altered. The sheets also facilitate the tracking, systematically, of data variability and quality, from one iteration to the next. This feature may then be used, during future iterations, to manage and control, progressively, the uncertainties associated with the modelling data.

Modelling has been conducted using the TEAM^{TM} software, for both the BRM and MIM case studies. The normalised data from each process, forms a separate foreground atom in the models. In chapter 11 of the thesis, it is explained that the TEAM^{TM} has a facility, which allows all transport and non-transport atoms to be exported as 'modules' which may then be considered in isolation, or aggregated into various other models, designed as appropriate. 'BRM Whole Site' model was thus formed from the re-aggregation of modules from the 'MIM Pb LCA' model, with additional modules created to model the secondary operations at the site.

All flows, in all foreground atoms, have been treated as variables. The ranges of these variables are expressed as coefficient of variation values, which have either been calculated from data or estimated. (Documents 5 and 9, of volume 5 of the Portfolio, indicate the complete set of values used for the BRM whole site model, and for the MIM Pb LCA allocated and unallocated models.) The variable values they may be altered using spreadsheet control panels which are external to the model. The TEAM^{TM} software also has an export facility to facilitate the export of these variables.

Further information on the Excel data sheets, and modelling using the TEAM^{TM} software, is provided in section 9.6.2.4 of chapter 9.

The overall structure of the 'BRM Whole Site' and 'MIM Pb LCA' models are indicated in Documents A1 and A2 (of the thesis Appendix). The databases created from the 'BRM Whole Site' model for the two modelled years (denoted as '(a)' and '(b)'), and from the 'MIM Pb LCA' allocated and unallocated models, will be supplied to the sponsors on completion. The printed hard copy of the inventory data, for the main operations, in these models forms volume 4 of the Portfolio. Tables 9.1 (chapter 9) and 10.11 (chapter 10) in the thesis indicate the operations, which have been depicted. These inventories are considered to constitute a significant reference source for supporting environmental decision making. They are, therefore, a significant innovation.

c) Results Interpretation Approach:

Due to the inherent uncertainties in complicated models, such as those provided in the BRM and MIM case studies, any interpretations of inventory data and effect assessment data must be made in conjunction with an assessment of the potential influence of such uncertainties on the findings. In the analyses of the graphs for the 'BRM Whole Site (a)', and 'MIM Pb LCA' allocated and unallocated models, a methodology has been applied which considers the influence of predicted potential variation ranges attributable to uncertainties in dominant contributors to the inventory or effect in question on any findings, which may be made. (The method of calculation is indicated in section 9.6.4.3 of chapter 9, and 10.4.4.3 of chapter 10.) Whilst the innovation makes results interpretation (in sections 9.6.4.5 of chapter 9 and 10.4.4.5 of chapter 10) more complicated, it is viewed as essential if environmental claims are to be made using such data.

d) Environmental Performance Indicators:

A simple methodology for producing environmental performance indicators, which allows performance to be compared from one modelled period to the next and plotted graphically has been devised. In the thesis, it has been used to compare the performance of several operations, for various inventory flows and effects of interest. Section 9.6.4.5.3 of thesis chapter 9, explains that simple comparisons, without having to conduct uncertainty analyses, are possible because the data used to construct the models, for the two years being compared, have been measured and calculated, or estimated and calculated, in a similar manner.

e) BRM Whole Site Modelling Approach:

This approach, developed for the BRM case study, is a hybrid between generalised approaches (such as used in the MIM Pb LCA) and site-dependent impact pathway assessment (IPA) approaches (defined in thesis section 5.3 of chapter 5). It has been adapted from the LCA method, and is considered to be most applicable for supporting decisions at the site-specific scale. The method of construction of the model is described in the review of the modelling approaches above.

4. Evidence of Contributions to Knowledge in the Field of Environmental Technology'

The major contributions are both philosophical and technical. The philosophical contributions are considered first.

a) Philosophical Contributions:

The thesis sets out the foundations for changes in the manner in which 'risk' is assessed and managed by the lead industry (and by other industries facing similar environmental challenges). In chapters 1 to 3, it is explained that whilst such changes will be needed by all industries, for the lead industry and these others, the challenges within the foreseeable future could threaten their entire existence. However, it is shown that the lead industry has a relatively strong position to defend, and provided appropriate strategies and means to accomplish these strategies are promulgated, it could have a 'healthy' future. It is also stressed that the onus for developing these strategies and means rests with the industry itself.

The changes set out in the thesis, are posited within a framework (see section 4.5 of chapter 4) which consists of a strategy, approaches and tools. These changes are both philosophical and technical. They are philosophical, because they set out, and identify, the features of a new conceptual paradigm, which has its basis within the concept of the 'risk society' (as defined by Beck (1992)), and which when fully developed will be:

- significantly more holistic than currently,
- multi-dimensional (i.e. operates effectively at different geographical and/or managerial scales),
- inherently flexible, and
- reflexive (i.e. developed iteratively through ongoing dialogue with all relevant stakeholders).

Since the environmental challenge is driven by the changes in the 'views of society' of the time, there is a need for a more effective bridge between these views and the attitudes and actions of organisations, such as the lead industry. Whilst it could be argued, the approaches and tools appear to be posited fundamentally within the structuralist paradigm (as explained in section 4.3.2 of chapter 4), the intention is that they will be developed into the future, through a process of ongoing 'reflexive dialogue'. In this process, they will become part of the new and developing paradigm.

The need for 'reflexive dialogue', where all participants are required to reflect critically upon, and challenge (in a constructive manner), their own pre-conceptions, is identified in the Overall Conclusions and Recommendations to the thesis (chapter 11), as being the key to achieving 'best' environmental decision making. It is also explains, that if environmental decisions are to continue to be viewed as legitimate in the developing 'risk society', then active moves must be made towards developing such dialogue. Hence, the framework (plus the strategy, approaches and tools), should be viewed as providing the foundation step for the evolution of the new paradigm within the industry and its other stakeholders. It is also viewed as providing a significant contribution to the evolution of the new paradigm in society at large.

b) Technical Contributions:

Whilst there have been many detailed reviews on the subject lead and human health and environmental effects, reviews attempting a comprehensive assessment of all significant emissions, from the processes involved for primary and secondary production operations (as identified in section 1 of this Summary), appear to be lacking. Therefore, the detailed reviews of available literature on the potential human health and environment effects attributable to the global lead industry, are considered to be a significant contribution.

The investigation of the underlying causes of the environmental challenge facing the lead industry, and the need to meet it through 'best' decision making (sections 4.1 to 4.4 of thesis chapter 4), makes a contribution to the discussion of what the challenge really is, and how all industry will have to respond.

The inventories provided in the Portfolio for the 'BRM Whole Site' (a) and (b) models, and the 'MIM Pb LCA' allocated and unallocated models, provide bench mark data, of operations which are considered to represent environmental best practice (or close to best practice at least). They also identify the key processes which give rise to particular inventory flows, and effects. Considerable efforts have also been made to try to provide a transparent audit trail from the raw data (and its likely quality), through to the inventory data. The RE is not aware of any other databases, covering the same processes, which provide data with such a detailed audit trail and, hence, such verifiability.

The interpretations of the inventory and effect graphs, provided in the thesis also provide important information, which were not available to the managers of these operations, and others, previously.

5. Attainment of Brunel/Surrey EngD Program Additional Requirements

Full attainment of these attributes, is indicated in Document F2 of the appendix to the thesis, which was submitted by the author for the final viva examination. It has not, however, been enclosed with this copy of the thesis, since it relates to the overall requirements of the EngD program rather than to the thesis itself.

AUTHOR'S NOTE

This thesis forms part of a Portfolio of documents submitted for the degree of Engineering Doctorate (EngD) in Environmental Technology. There are some references to documents in the Portfolio, which have not been enclosed with this copy of the thesis. However, the thesis is considered to be complete without the enclosure of such additional documentation.

1. Introduction

The lead industry faces an uncertain future. Working in an uncertain environment is not new to the industry. However, this uncertainty is more profound and threatening than before. For example, Robert J. Muth of Asarco Inc. USA in his paper to the 11th International Lead Conference 1993 noted, "recent attacks" on the lead industry had changed from being on the products and human exposures to "attacks directed at lead markets" [Muth (1993)]. He also suggested that whilst this was especially the case in the United States (US), it was occurring globally. Muth further contended, this was "a totally different kind of confrontation and that [it would] make past challenges seem trivial in comparison." He concluded from this that, in the US, the government had started to adopt the attitude that "lead is an indestructible toxic [substance] and should be left in the ground." As a result, he contended that practically all lead uses are susceptible to early elimination.

These statements raise questions of fundamental concern for the whole of the lead industry.

- Is Muth correct in this assessment?
- If it is the case what are the underlying causes?
- How should the lead industry respond?
- Does the lead industry have any future?

This thesis examines these questions in some detail. The argument is made, that he is correct in his contentions of there having been a shift in the nature of the attacks on the industry and that it could face the early elimination of its entire product range. The argument is also made, however, that, to a great extent, the industry holds within its grasp the keys to its own destiny.

The underlying theme of this thesis, is the lead industry needs to understand the changes which have occurred, the reasons for their occurrence and what future changes are considered likely. The thesis also explains that, despite the apparent threats, the industry has a relatively strong position to defend. Therefore, if appropriate strategies and means to accomplish these strategies are promulgated, it could have a 'healthy' future. However, a significant proportion of the onus for developing these strategies and means rests with the lead industry itself. Indeed, it is explained that the future of the industry rest to a significant extent on its ability, and others, to:

- assess and manage the human health and environmental effects associated with its products and activities,
- communicate its human health and environmental case effectively to all relevant 'stakeholders'[1], and
- influence the perception of the human health and environmental 'risks' associated with the industry and its products in the minds of these stakeholders, so that more 'balanced' (i.e. unbiased) attitudes are engendered.

In essence, therefore, this thesis is concerned with developing more effective human health and environmental 'risk' assessments and communication, and from this influencing the peoples' 'risk' perception of the industry. It is thus also concerned with ensuring fair and effective 'risk' management measures are adopted both within the industry and by those charged with regulating it.

However, for this to happen with optimal effectiveness, a conceptual change in the way the industry views itself, will be necessary, and new strategies, tools and assessments will have to be developed. The nature of this conceptual change, an integrated strategy, plus specific tools and assessments developed by the author, to provide the basis to meet this challenge, are detailed below.

The manner in which they provide this basis and their potential use for decision making are also assessed. In such decision making situations, decisions will have to be made where a variety of epistemological, methodological and technical uncertainties will always tend to exist and some of them cannot be eliminated. Therefore, approaches are needed which allow for those uncertainties, which cannot be reduced to negligible levels, to be taken into account effectively and the 'best' decisions still be arrived at. These questions are

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[1] A 'stakeholder' is defined here as someone with a legitimate interest in a decision. The 'someone' may be an organisation or an individual, and 'legitimate' is defined as someone who may be affected by the decision [Cowell et al. (1997)].

considered in some detail and approaches are developed in this document which take effective account of this problem.

2. An Industry Under Threat

Lead and its compounds are toxic. This is indisputable. However, this is only a problem if it is able to enter body tissues. Unfortunately, lead's usage leads to the potential for it to escape into the environment and thereby to pollute it. The usage of lead in human society dates back thousands of years. Therefore, with it went its inevitable dissemination into the environment. This in turn, resulted in its uptake into the tissues of living organisms. Indeed, there is evidence to suggest that all people living in the modern world have a biological burden of lead in their tissues and that it is at least partly anthropogenic in its origins. As knowledge of the toxic and bioaccumulative nature of lead has developed, there has been a concomitant development of concern. This concern has centred both on environmental and occupational exposures. In turn, it has led to a variety of initiatives and legislation aimed at reducing or trying to eliminate exposures.

In response to these developments, the industry has spent considerable resources over the last few decades aimed at reducing occupational and public exposures from emissions. This was brought about partly as a result of the more stringent regulation of the industry but also because of its desire to be environmentally responsible. Considerable resources have also been spent on improving the environmental performance of existing process technology and developing and installing both clean-up and cleaner technologies.

As knowledge of the significance of the environmental burden of lead contamination developed, it became apparent that the vast majority of the environmental and human exposure came from three main contaminative uses. These were:

- additives in petrol;
- additives in paint;
- lead pipes and lead based solders used in pipes to convey drinking water.

As a result these are all now in decline. This has left the industry with a range of products which are relatively non-dissipative in nature. However, concern over lead and the potential threats to the lead industry appear to be continuing unabated.

A substantial amount of research has been and continues to be conducted to identify the extent and routes of environmental contamination and its environmental and biological significance thereof. This work has taken place both at national and international levels. Thus, a number of countries have attempted to assess the extent of soil contamination nationally. In the United Kingdom for example the Environment Agency (the regulatory authority) now has the duty to assess the extent of environmental pollution (including lead contamination) both nationally and regionally and to carry out regular reviews of the extent of this contamination. On an international level, a number of agencies including the United Nations-Economic Commission for Europe (UN-ECE), the International Joint Commission (IJC) and the Organisation for European Co-operation and Development (OECD) have formed task groups which have studied the problem of the dissemination of heavy metals in the environment. [Rasmussen (1996)] In 1995, the UN-ECE task force reported that there is now enough evidence to conclude that long range transport and deposition of heavy metals (including lead) has occurred [Wilson (1995)]. As a result of these findings, it recommended for lead, mercury and cadmium that:

- emissions should be reduced;
- the principle of Best Available Technology and the precautionary principle should be applied;
- restrictions should be placed on certain uses;
- substitution should be used as a principle.

Whilst all are of concern, the last recommendation is the most profound since it threatens the future of the entire lead industry. Indeed, this recommendation is now showing signs of becoming a reality, since active steps have been made, recently, by the Danish government towards imposing a ban on lead containing products within Denmark [Ministry of the Environment and Energy (1998)].

Thus, all lead products face the threat of substitution. There are numerous examples of other potentially threatening moves. However, probably, the most potentially significant development has been the publication of a document entitled *A common Strategy for International Chemical Control Work in Sweden* [Boreiko (1995)]. This document was jointly produced by a collection of Swedish regulatory authorities consisting of the National Chemicals Inspectorate, the Swedish Environmental Protection Agency and the National Board of Occupational Safety and Health. It details Sweden's national position on the control of chemicals and outlines measures and/or vehicles that it hopes could be used to impose their national policy at the international level.

The document outlines three basic regulatory policies, which if implemented internationally would have far reaching implications. These are:

I. The precautionary and substitution principle

This would require anyone handling or importing a chemical (i.e. any lead compound and lead itself) to judge whether the same results could be achieved at a lower risk with a different product or in a different way. It also requires that if they could be achieved, then they <u>must</u> substitute the product.

II. The duty of investigation principle

This would place the burden of proof on manufacturers of chemicals to establish the hazards of the chemical products themselves rather than on the regulatory authorities.

III. The polluter pays principle

This would require the manufacturers and importers of these chemicals to cover their own costs for compiling data concerning the impact of their chemicals on both health and the environment.

If any or all of these principles were implemented internationally, the cost overheads and uncertainty on the industry would increase dramatically.

All of this provides evidence to support the claim of Muth (1993) that attacks on the lead industry have shifted and now threaten the future of the whole market. It also gives strong support to his claim that the industry faces the threat of early substitution of its entire product range.

Additionally, it is predicted the lead industry will be threatened, increasingly, by societal demands for sustainable development. Ultimately if humans are to have any long term future, the technology of our society will need to be environmentally clean and societal development will have to be sustainable. Whilst it is likely to be some time before such objectives could be realised, the first governmental pressures on industry to develop strategies for sustainable development have begun to be exerted. Currently, the main source of this pressure is through Agenda 21 (which was one of the outcomes of the Rio Conference of June 1992). The obligations placed upon governments to develop action plans for industry were considerably whittled down from the original proposals, as a result of an effective lobbying effort by the International Chamber of Commerce [Mayer (1992)]. However, Agenda 21 still contains requirements, such as the international enforcement of the polluter pays principle, which could prove somewhat onerous to industry. Agenda 21 is likely to be merely the first step in this pressure to move industry towards sustainability. Hence the pressures on industry to develop strategies and means to achieve sustainability are expected to grow. The change to a sustainable society will require a world-wide shift in the conceptions of the roles of all members of society. Radical changes will have to occur across the whole of industry. In such a society, the production and usage of metals such as lead, might be extremely limited. Therefore, in the coming period the whole of the metals industry will be subject to fundamental changes and over the long term could be virtually eliminated.[2]

However despite these threats, in the short to medium term the prospects for the lead industry appear to be favourable in several respects. For example, lead-acid batteries are one of the major contenders for providing the power for the new generation of zero emission vehicles (ZEV), [see for example Hunt (1998)] , which may be based predominantly on fuel cell and lead-acid battery hybrid systems. Additionally, since the global usage of cars is predicted to show a sustained increase, the usage of lead for the

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[2] How sustainability is being interpreted both theoretically and in practice for the primary extraction industries has been examined in a co-authored paper in *Resources Policy*, entitled "Sustainability and the Primary Extraction Industries: Theories and Practice". MIM, one of the two companies studied in this thesis, also forms one of case studies in the paper. (The paper, plus its full citation details, are detailed in the 'List of Publications', provided at the front of this copy of the thesis)

production of automotive lead-acid starting lighting ignition (SLI) batteries is also expected to increase, at least until 2005 to 2010 [Pugh and Breeze (1995), Rich (1995)], and almost certainly beyond then. Hence it appears, the lead industry has (potentially at least) a strong position to defend. What it needs is appropriate strategies and tools to fight its case. The industry's challenge is to meet the increasing threats in a positive and proactive manner. It is an emergent field, those industries that are more effectively prepared are likely to fare better in the battles ahead.

The author believes, the lead industry needs to appreciate that these potential threats and opportunities are themselves part of a wider process which is already occurring within industrialised society. Indeed, there is a growing body of opinion, which has been arguing that industrialised society is undergoing a decisive transitional period. Therefore, the author considers:

- the pressures to move towards a more environmentally sustainable society,
- the rapid growth in the size and influence of environmental groups since the 1960s [Lowe and Goyder (1983)],
- the increasingly pervading view amongst lay people of mounting environmental damage and progressive poisoning by waste (despite the fact that human health and environmental standards have never been so high throughout the entire history of the human race) [Irwin (1995)],
- the increasing opposition of lay people to the siting and disposal of facilities such as oil platforms, waste incinerators and nuclear power repositories (a phenomenon termed 'technological gridlock' by some American academics) [Irwin (1995)], and
- the aforementioned threats to the lead industry, are all symptomatic of this wider process.

One of the most significant and influential theoretical contributions in this field is from Ulrich Beck. Beck (1992) has argued that industrial development is becoming increasingly characterised by two features. The first of these is that the axial principle of industrial society is shifting from the distribution of wealth and goods to a distribution of 'risks'.[3] He describes this new society as a 'risk society'. However, Beck contends that this society is

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[3] The term 'risk' in this context is defined by Beck (1992, p.21) as a "systematic way of dealing with hazards and insecurities induced and introduced by [the process of] modernisation itself".

still, and will continue to be, an industrial society. This is because, it is industry in conjunction with science that has been, and will continue to be, responsible for the creation of these risks. The second feature of this society, Beck argues, is that risk is becoming 'reflexive' i.e. that it is becoming its own theme. Thus, in the industrial society the 'logic' of wealth production has dominated the 'logic' of risk production. However in the risk society this relationship is reversed. Hence in the risk society, the distribution of risk becomes the dominant theme [Beck (1992)]. The author believes that the growth of the perceived threats to the lead industry provide strong evidence to support this claim. Thus, the perceived growth of the threats is thought to stem from an increasing domination of the perception of society, that the risks caused by the actions and products of industry need to be reduced and eventually eliminated.

In this thesis, 'risks' are referred to in their broadest sense. Thus, they include technical risks, which many physical scientists would argue can be evaluated from measurable environmental burdens and effects. However, they also include the plurality of different 'public' risk perceptions which are influenced by aspects such as the degree of trust and the credibility of the risk causers.[4] The author believes that since these 'factors' may be regarded as equivalent to risks they are also reflexive. Hence they are becoming their own theme. Therefore, for the development of an integrated strategy and tools that will be successful in enabling the lead industry to meet the emerging environmental challenge, it will need to be reflexive as well. Thus, the strategy will need to consider their impacts both upon society and the lead industry, which is itself both a product of and part of that society.

3. Optimally Meeting the Environmental Challenge

The previous section showed how the threats to the industry have broadened and, some such as the substitution principle, are now challenging its whole future. With the development of the substitution principle, the industry will need to be able to demonstrate

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[4] The 'public' are here defined as those exposed to the risk in question. The 'public' may be divided through factors such as age, social class, occupation, gender, ethnicity, and degree of support for a particular position. Different 'publics' may be separated from each other by both physical (e.g. geographical) and/or social (e.g. cultural) differences. Within the various publics that may also be a variety of divisions, differences and conflicts.

clearly the environmental burdens of all of its products, at a variety of different scales. These will range:

- from their whole life cycles i.e. from their cradle (as raw materials) through to their grave (as waste),
- to the individual production process step and
- to the individual environmental emission source scales.

The author contends that the traditional responses of the industry are unable to provide a proper and balanced consideration either to these factors or to the way in which they impact both on society in general and upon the lead industry itself. These shortcomings are expected to become increasingly apparent in the times ahead. Therefore new strategies and tools are urgently needed.

Unfortunately, it would never be possible to create a perfect system which had zero losses to the environment at all stages in the product's life cycle and on all occasions. Nevertheless, it will be possible systematically to design products, their modes of use, recycling and disposal in such a way that the potential human health and environmental effects are minimised and reduced to levels that are competitive and may even outperform potential rivals throughout their life cycles. Where appropriate, such design should also occur at the individual process and environmental emission source scales. Also, where emissions are unavoidable, for thermodynamic reasons, the life-cycle of the products with which they are associated, need to be designed in such a manner that when they are released, it is into environments where their fate is such that resultant human and biotic environmental exposures are minimal. To achieve these requirements, approaches which are significantly more integrated will be required.

The author contends that, in the immediate to medium term future, the biggest threats to the industry will be from attempts at substitution, from demands for lower emission limits and from views that lead is a dirty polluting substance. Thus an approach needs to be adopted which will ensure these threats are addressed. i.e.

I. To meet the threats of substitution, it needs to develop means to compare most effectively the environmental burdens associated with lead and lead products with

- so called 'environmentally friendlier' substitutes;
- services that are currently supplied by other means but which the industry believes its products should replace (e.g. replacing petrol driven vehicles with lead-acid powered ZEVs).

II. To meet the demands for lower emission limits, it needs to assess the accuracy and adequacy of the currently accepted techniques for measuring plant emissions, their dispersion in the environment and for identifying their human health and environmental effects.

III. It also needs to adopt more effective means to improve both the image of lead and the trust of the public in the industry.

To date, efforts have been made by lead industry to tackle at least some of these issues. However, if efforts are to be directed in the most effective manner, they need to be organised systematically and co-ordinated within an integrated strategy. Section 4 explains why this is the case and outlines the framework of such a strategy.

4. Developing Means to Promote the Case for Lead and to Argue Most Effectively Against its Substitution

Demand for primary lead is expected to remain relatively constant for at least the next ten years. However, estimates of the demand for secondary lead predict it will increase from about 52% of all lead (1995 figure) to between 59 to 63% by 2005. Pugh and Breeze (1995) have also estimated that consumption of lead for batteries will increase from 66 to 73% over the same period. However, dissipative uses are expected to continue to decline at a rapid rate and the other non-dissipative uses such as cable sheathing are expected to remain relatively constant over the same period. Thus, it is expected, the lead industry will grow over the next ten years due to an expansion of the secondary lead business but that this growth will be increasingly dominated by batteries. Since battery recycling rates are already between 80 to 90% [Wilson (1993)], this expansion will, presumably, have to be achieved by an increased recycling of lead from other sources. The industry is also currently pinning much of its future hopes on being able to provide a convincing argument that lead-acid batteries will be the best option, both practically and economically, for powering the new generation of ZEVs. Whilst it has been argued that it will not be a dramatic market driver for the next ten years, it is essential the industry starts to prepare its environmental case.

These trends and the aforementioned threats, indicate the future of the industry hangs upon its ability to argue effectively its case in a number of key potential battle areas. There is the real danger, if inadequate assessments are made and inadequate approaches are adopted, the lead industry could lose out in arguments over substitution when its position environmentally may be significantly stronger than it had realised. Thus, it is clear it needs to adopt the most effective means available to argue its case.

The urgent need for adequate assessments, which would enable it to argue an effective environmental case for lead being used to power the new generation of ZEVs, was put into stark perspective by Lave et al. (1995) who conducted a preliminary analysis of the lead emissions and lead wastes associated with the life cycle of lead-acid batteries (i.e. from raw material extraction, production, their use, end of life recycling and disposal). They contended, that the amounts of lead in wastes and emissions, generated per kilometre, would be 60 times greater with an electric vehicle powered using lead based batteries than would be emitted from a comparable vehicle operating on leaded gasoline. As Steele and Allen (1996) have noted, this assertion led to a heated debate, both within and outside of the industry. [See for example, Allen (1995), Stempel and Ovinshinsky (1995), Gellings and Peck (1995), Gaines and Wang (1995), Hwang (1995),Rubenstein and Austin (1995), Socolow (1995), Sperling (1995), Lave et al. (1995a).] It also revealed, however, that data to support or refute these arguments was “exceedingly sparse” [Steele and Allen (1996)].

It was in response to this problem, that the author started to develop the methodology outlined in this thesis. The paper written by the author, presented to the 5th European Lead-acid Battery Conference (5ELBC) in September /October 1996 and subsequently published in the Journal of Power Sources (Vol. 67 (1), pp. 225 - 236) provides some of the initial elements of this framework.[5] The presentation and the paper explain, *inter alia*, the need for life-cycle based environmental assessment approaches, such as life-cycle assessment (LCA), set within an integrated strategy. Also in response to the debate generated by the assertions of Lave et al. (1995), and contemporaneous with the author's writing of the presentation and paper, the International Lead Zinc Research Organization Inc. (ILZRO) entered into discussions with Ecobalance Inc. ILZRO (who are based in North Carolina, USA) are an industry funded research organisation for the lead and zinc industries, and Ecobalance (who are an affiliate of the Ecobilan group) market the TEAM™ software and DEAM™ database for conducting LCA. Their discussions led to the development of a proposal and the subsequent initiation of a project to conduct a global life-cycle inventory analysis of lead-acid batteries (ILZRO Program Number: LEH - 8) [Ecobalance (1996)].

The proposal for this project was developed without the active participation or knowledge of the author, who was only made aware of its existence a few days before his presentation to the 5ELBC. Unfortunately, it now appears be in abeyance due to cost and time overruns. However, the fact the lead industry have been prepared to countenance such undertakings, indicates not only its commitment to safeguarding its future, but also vindicates the author's assertion that life-cycle based approaches are essential if the industry is to mount a successful campaign to argue its environmental case. Indeed, the debate generated by Lave et al. (1995) continues, since Steele and Allen (1998) have now conducted an 'abridged LCA', comparing four emerging battery technologies for electric vehicle batteries and have concluded that the recyling of nickel-metal hydride batteries appear to be the most environmentally benign [Steele and Allen (1998)]. However, other researchers have argued that lead contamination of the environment would be unlikely to increase if lead-acid battery powered cars hit the roads in large numbers [Anonymous 1997)]. Such developments further underline the urgent need for effective assessment processes to be developed by the industry. Nevertheless, whilst these efforts can be valuable contributions, they are insufficient on their own to ensure the lead industry meets the environmental challenge in the most effective manner. Why the author considers this to be the case is addressed below.

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[5] The full citation details for this paper are detailed in the 'List of Publications', provided at the front of this copy of the thesis.

4.1. Mapping out the Underlying Dimensions of the 'Argument'

From the foregoing discussion, it is apparent that the stakes facing the industry are extremely high. Therefore, if it is to stand a fair chance at winning, with the deck of cards with which it has been served, there needs to be an examination of the major strengths and shortcomings of current strategies.

With respect to strengths, the author recognises the valuable efforts of those within, and allied to, the lead industry in:

- identifying the major human health and environmental effects associated with the lead industry and its products,
- identifying, developing and implementing new technologies and techniques for reducing these effects to ever lower levels,
- communicating the environmental and human benefits and risks of lead products and lead product production supply chains, and
- conducting research efforts to identify and to help place the industry in a strong position to defend current and develop potential future markets.

Of particular note are the efforts of the Advanced Lead Acid Battery Consortium (ALABC), which was formed in 1992 as a program of ILZRO, and represents a collaborative effort between the world's lead producers, battery manufacturers, component suppliers, and related industrial sectors. Its purpose has been to conduct a world wide program, aimed at making electric vehicles (EVs) a viable element in the vehicular market place [Moseley (1999)]. Specifically, research and development have been focused on improving the cycle-life span and specific energy storage capabilities of valve regulated lead-acid batteries (VLRAs), whilst still maintaining their high power density and major cost advantages over other battery systems. Efforts have also been made to improve, dramatically, the battery charging time [Cole (1996)]. As part of its program, the ALABC obtained funding from the European Community (under the Brite-EuRam program) for a US$4.4. million project, to bring together these various lead-acid battery developments into prototypes for EV applications and to conduct bench testing of them [Cooper (1996)]. Over the seven years (which the ALABC's project was scheduled to run), considerable success has been achieved in achieving these targets. Thus, Moseley and Cooper (1999) report, the design target for specific energy of 36 W h kg^{-1} has been achieved successfully and, whilst the cycle-life is still short, the deployment of suitable charging regimes "continues to be a fruitful area for extending the life of VLRA batteries".

In February 1996, the Environment Ministers of the Organisation for Economic Co-operation and Development (OECD) issued a declaration on Lead Risk Reduction, seeking to develop voluntarily (and to strengthen) national and co-operative efforts to reduce the risks from and exposure to lead [Wilson (1999)]. The lead industry has demonstrated its active commitment in meeting this goal, with its setting up and funding of the International Lead Management Centre (ILMC), which it has charged with organising and co-ordinating these efforts within the industry. Such pro-active efforts also demonstrate the ability of the global lead industry to organise itself co-operatively, where the need arises over specific issues.

These collaborations also show what can be achieved when the lead and other associated industries organise and commit themselves to concerted action. The problem, as the author sees it, is not the efforts themselves or the quality of the output, which as shown above can be outstanding. Rather, it lies in the fact that such efforts have tended to be 'atomised'. Thus, there has been a tendency for efforts to be fragmented and unbalanced in the sense that some partners within co-ordinated efforts end up being made to shoulder more responsibility than others. This is also not to say that there has been a lack of attempts (and success) at co-ordination between efforts. The ALABC and ILMC are testiment to the fact that such co-ordination exists. Rather, it is that they lack a cohesive conceptual 'glue', a new 'paradigm' in the Kuhnian sense, to hold them together. Kuhn (1970) explained that a 'paradigm' acts like a "preformed and relatively inflexible box" (p. 24), a conceptual 'box', that colours the way one views the world and becomes a deep seated tool that is applied to all problems.

A paradigm thus has a philosophical basis. Every person and every organisation carries with them one or more paradigms which 'colour' the way they view the world. Kuhn argued that paradigms, tend to be in competition between each other. Also, whilst a paradigm is capable of being modified through the processes of adjustment (being manipulated to enable it to encompass new data and ideas within its framework of understanding) and articulation (deepening of understanding within the framework provided by it), it has a conceptual core, which may prove incompatible with other competing paradigms.

The problems of competing paradigms are aptly illustrated by the fundamental divisions which cut through the field of risk research. Two competing paradigms exist; one is essentially structuralist (the dominant approach within industry), whilst the other is interpretivist (a social action centred way of thinking). Underlying them are differing and competing theories about what risk is and how it should be explained. As a result, it has led to the tendency for substantially differing interpretations when researchers are investigating similar problems. It was noted in the Introduction (chapter 1) that the key tasks facing the lead industry may be resolved around developing more effective human health and environmental risk assessment and communication approaches, and from this influencing peoples' risk perception of the industry. The question, therefore, is now raised of how can one achieve these aims, if a consensus is lacking and may not be possible to achieve, not just over the applied methodologies but also with regard to their fundamental conceptual underpinnings. This question is of crucial importance to this thesis, since one of its key aims is to set out a strategy which will enable the lead industry to meet the environmental challenge with optimal effectiveness. Since this strategy is viewed by the author as being the explicit delineation of the paradigm being proffered, the strategy is, in effect, a mapping out of the dimensions of this 'conceptual box' for the industry. If there is no consensus to the above, however, and if no consensus is possible, then how can one develop a strategy which is optimally effective, without fundamentally disadvantaging one or other of the different risk camps?

In the author's assessment, such potential deficiencies should not just be ignored on the basis that consensus may not be possible and that the interpretivists are a relatively small group anyway. Two reasons underline this stance. Firstly, such an approach would be blinkered and would not help to advance development in this area. Secondly, there is evidence to suggest that society, globally, is undergoing a fundamental shift in the way it views itself and the industrial base which supports it. Thus, it was noted in chapter 2, that Beck (1992) contends the way in which people tend to view the modern world has changed, and that we now live, increasingly, in what he describes as a 'risk society'. This society, though still industrial, now has risk as one of its most increasingly dominant themes. If his assessment is accepted, this 'risk society' is a paradigm in the Kuhnian sense, since it colours the entire way that people view the world. Crucially also, as explained below, some elements of the interpretivist paradigm appear to be becoming part of this new view. Note, that since risk appears to be becoming an increasingly dominant theme, the strategy, approaches and tools developed in this thesis, to meet the environmental challenge, are directed at developing significantly more effective means of:

- risk assessment and management within the industry and
- risk communication between it and its stakeholders.

If the change described by Beck is already occurring in society, then the lead industry must take this on board and it should be properly reflected in the responses it develops to counter the threats posed by the environmental challenge. Indeed, this is already happening to some extent. This is because, the lead industry is itself part of industrialised society. Hence, the organisation and the people who comprise it are undergoing these conceptual changes anyway. What it requires, however, is a fundamental strategy, tools and managerial approaches which will enable it to move from being in a more passive participatory role to being a far more active steerer of these changes. The differences between these structuralist and interpretivist paradigms cannot, therefore, just be ignored. This is because identifying what they are is a necessary prerequisite to adopting such a role, if it is to be optimally effective whilst at the same time not fundamentally disadvantaging of either one or other of the competing paradigms.

Tiemann (1987) has identified the salient differences between the structuralist and interpretivist paradigms. He argued that "approaches to risk management [and I would argue risk assessment and risk communication also] depend upon the society and times in which risk questions arise". He also argued that these changes led early on to a split between natural and naturalist scientists, which in turn led to the division of the social sciences into the 'structuralist' and 'interpretive' camps. The 'structuralists' modelled their ideas on the natural sciences whereas the 'interpretivists' modelled theirs on naturalistic explanations.

This division has had a profound affect on both the methodologies used as well as the kinds of conclusions they tend to draw from their data. These divisions are summarised in Table 4.1 below.

STRUCTURAL (Basis in the Natural Sciences)		INTERPRETIVE (Naturalistic Basis)	
1.	Experimental	1.	Naturalistic observation
2.	Deductive reasoning/proof	2.	Inductive reasoning/analogic reasoning
3.	Few variables with simple relationships	3.	Confounding variables in complex interactions
4.	Similar research elements	4.	Individual differences
5.	Determinism	5.	Stochastic processes
6.	Constant (no change)	6.	Variable change rates
7.	Continuous change	7.	Discontinuous change

[Modified from Tiemann (1987)]

Table 4.1: Summary of a Few of the Differences Between Structuralist and Interpretivist Methodologies

Watts (1983), whose views are based within an interpretivist paradigm, has decried what he considers to be a poverty of theory in hazard research. He argued that:

> "hazard theory has been framed by concepts and assumptions which carry a historically specific view of nature, society and man and hence by extension, of the relations between them. This colours the <u>entire</u> corpus of hazards research".

Thus, Watts (1983) has criticised of much of the conventional work which rests on building blocks inherited from biology and cybernetics (i.e. its basis is in the natural sciences and so it is essentially structuralist).

He further adds later, in the same paper, that:

> "this ecosystemic approach, broadly defined, blurs and obfuscates the character of inner actions and erodes the irreducibly social character of human life to atomised individuals organisms...".

These criticisms parallel the problems faced by the Royal Society when it invited , for the first time, psychologists, anthropologists, sociologists, economists and geographers to participate with physical scientists in the writing of its report on 'risk', entitled: *Risk: Analysis, Perception and Management* [Royal Society (1992)]. The result, was disagreement about the nature and meaning of 'risk'. To quote Adams (1995, p 9)

> "A contention...that the physical scientists found variously maddening or frustrating is that risk is culturally constructed. According to this perspective, both the adverse nature of particular events and their probability are inherently subjective".

Indeed, the whole contention that there is a distinction between perceived and actual 'risk'; a notion that has been accepted as gospel by most 'risk' assessors (who operate almost exclusively from a structuralist paradigm) was put into question by those social scientists who tended to opt for more interpretivist explanations.

The criticisms made by Watts (1983) of natural hazard research are from his interpretivist paradigm. From his point of view, his claim of their being a poverty of theory in natural hazard research because most human cultural ecological work (i.e. natural and technological hazard research work) has been performed by persons employing the structuralist paradigm. Therefore he is justified in making this claim, since the majority of work has been carried out by researchers employing a paradigm that he does not recognise as being valid!

Thus, the interpretivist and structuralist paradigms are two very different ways of interpreting the world. Whilst the dominant approach in the industrialised world is essentially structuralist, some more interpretivist elements in the way people view the world are discernible within the environmental challenge. In the shift towards a 'risk society' as defined by Beck (1992), the magnitude of 'risks' are no longer viewed by many as something which can be objectively measured but are seen as entities where the assessment is culturally determined. The fact that this is starting to become a pervading theme is, in the author's assessment, an indication that more interpretivist views of the world are starting to be exerted.

In industry, amongst regulators and within governments, structuralist approaches dominate. Therefore, strategies, approaches and tools which are developed to help the lead industry meet the environmental challenge must reflect this if they are to be effective. However, they must also be able to accommodate the more interpretivist elements which exist and which, if Beck's assessment is correct, will grow in importance. The author has aimed to achieve this by ensuring that the strategy itself, the managerial approaches and the tools that are designed to go with it and the way they are expected to be used in decision making are all flexible enough to accommodate the more interpretivist manifestations of the environmental challenge.

The intention is also that they will be reapplied regularly into the future and will be modified, as necessary, to meet changing needs. This also applies to the assessments, which serve as tools to aid decision making.

4.2. The Underlying Causes of the Environmental Challenge and Building an Effective Response

It was argued, in chapter 3, that the environmental challenge appears to be linked with a concomitant conceptual change in the way lay people view the environment and the role(s) of industry within it. Thus, to be optimally effective, the strategy, approaches and tools all need to be normatively based (i.e. to reflect and help to impose these norms). It is for this reason they need to be flexible, since the norms will evolve with time. It was also explained that this evolution is now characterised by two increasingly dominant phenomena, namely

- the management of 'risk' is becoming an increasingly contentious and dominant theme, in the manner described by Beck (1992), and
- the phenomenon of 'technological gridlock' has become an increasingly significant problem [Irwin (1995)].

Therefore, if the strategy, approaches and tools developed in this thesis are to be optimally effective, an understanding of the likely underlying causes of such phenomena is required.

In relation to the increasing contentiousness of risk management and the problem of 'technological gridlock', Slovic (1993) has argued that trust (or lack of it as is often the case) has played (and continues to play) a central role in their development and sustainment. Additionally, he contended that public fears and opposition to developments, such as nuclear waste disposal plans, result from a profound breakdown of such trust by lay people in the scientific, governmental and industrial managers of nuclear technology. He also attributed the limited effectiveness of risk communication efforts about nuclear waste in the USA, to a lack of trust and concluded that "trust is more fundamental to conflict resolution than is risk communication". Such assertions have, potentially at least, profound implications for the lead industry, as this may be at the root of some of the more apparently prejudiced decisions and actions, which have been and continue to be made against it.

The author considers that this growing realisation of the inadequacies of the current risk management approaches and the decline in trust by the lay public of those charged with managing such activities, stems from the changing social and technical contexts of society. This has manifested itself in two ways. Firstly, technical innovation is tending to become more complex. This has meant that the uncertainties, and hence the risks, associated with them have also grown. Secondly, lay people have also started to "realise that what accounts for fact is conditioned by political, organisational and peer pressures" [Otway 1987)]. Thus, they have also started to realise the limitations of science, although for many this process is perhaps only intuitive. Otway has argued that as a result of these changes, the authority and expertise of science, in the field of risk assessment and management, is fading. Hence it may be concluded, this is the likely reason why the process of risk management is becoming more contentious. These changes seem likely to continue.

Therefore, unless its approaches (and hence also its underlying paradigm) undergo fundamental change, the crisis will almost certainly worsen.

However, intellectual change is coexistent with societal change [Tiemann (1987)]. Therefore, as society changes so do ideas. Thus, it was societal change that caused the Royal Society to change their ideas, break with tradition and enlist the help of social scientists to research and write its 1992 report on risk. Radical changes in the way risk management is to be conducted in the US have also been proposed in the US National Research Council (1996) report entitled *Understanding Risk: Informing Decisions in a Democratic Society*. These too are a reflection of underlying societal changes that have been taking place.

Thus, building an effective response to the environmental challenge must involve an active engagement with the above problems, so that trust between the lay public in general (and stakeholders in particular) are developed. How this may be achieved using the framework developed by the author is discussed in section 4.4.3.

4.3. Effective Establishment, Management and Communication of Environmental Risk Credentials

From the preceding discussion it is apparent that significantly more effective approaches for risk assessment, risk management and risk communication are necessary both within the lead industry and by other stakeholders associated with it. Since this is of such fundamental and critical importance for providing the basis for the strategy, approaches and tools outlined in the thesis, it is necessary at this juncture to examine, in some detail, the strengths and shortcomings of current approaches from the above perspective. Information on the similarities and differences between UK and US approaches to risk management has also been provided in section 1.2 of Document 24, in volume 5 of the Portfolio. Its purpose is to show how the analysis below can be used to assess the leading approaches currently in existence.

4.3.1. What Is Meant By The Term 'Risk Management'?

Hood et al. (1992 p. 135-136) noted that 'risk management' is a term which has no single meaning. However, they identified two broad types of approaches to 'risk management'. Thus approaches such as those of quantitative 'risk' assessment and economic and government regulation address it in a *generic* way, whilst others contain *detailed investigations* of how particular 'risks' arise. In the latter context 'risk management' is normally broken up into distinct specialisms and disciplines. Thus, traditionally, academic 'risk management' research has been divided according to the type of 'risk' being managed. Hence it has been, and continues to be, divided into the 'natural', 'technological' and 'social' 'hazard' fields.[6] This has occurred despite the fact that such distinctions are not absolute. Indeed, it has been argued that all 'hazards' are either 'natural or at least 'quasi-natural' [Burton and Kates (1972)]. The detailed investigation approaches conventionally also distinguish between levels of *scale* and *frequency* in 'risk management'. Thus, research into disasters (which are high-magnitude, low-frequency events) has become a recognisable sub-field of the social sciences since the 1950's. Other recognisable sub-fields deal with incidents of lower magnitude and higher frequency such as road accidents and accidents in the home [Hood et al. (1992)].

Whilst 'risk management' is a wide ranging field with no single meaning, Hood et al. (1992) have noted that, at its most general level the process of 'risk management' can be understood in terms of the three basic elements of organisational control theory. i.e.

1. The setting of goals (implicitly and/or explicitly).
2. The gathering and interpretation of information.
3. Action to influence human behaviour, to physical structures or both.

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[6] The relation between 'risks' and 'hazards' and between the 'natural', 'technological' and social' 'hazard' fields is discussed in detail in the document by the author, *Hazard Research: Two Separate Streams of Concern. What Future Directions?* EngD. Module: Risk Perception, pp. 2-5; 1995. This was enclosed in volume 2 of the Portfolio developed by the author, but has not been enclosed with this copy of the thesis.

Each of these elements are interlocking. However, they are also problematic and disputed. Thus disputes arise as to:

- what are/should be the goals of the 'risk management' process;
- who is/should be involved in the setting of these goals;
- who decides/should decide what information is to be collected and how it should be interpreted;
- who decides/should decide what action is/should be taken to ensure that the 'risk' is effectively managed?

Such decisions inevitably involve a trade-off of harms and inconveniences against costs[7] since all human activities entail certain degrees of 'risk' [Horlick-Jones (1996)]. Thus the above disputes concern where the boundaries (epistemologically, physically, spatially and temporally) are drawn and who are involved in the assessment, decision making, implementation and monitoring processes. Where they are set will depend on the following:

- the nature of the 'risk' being managed;
- the relationship(s) of the person(s) and/or organisation(s) involved in managing the 'risk' to the 'risk' itself;
- the conceptual paradigm(s) within which the participants are operating.

All three are inter-linked, evolving and also subject to contention. However, an understanding of these aspects is a fundamental prerequisite to an appreciation of why 'risk management' is such a wide ranging field with no single meaning. Therefore they are considered in some detail below.

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[7] The 'costs' are not necessarily always monetary. For example, they may refer simply to reductions in amenity value.

4.3.1.1. The nature of the 'risk' being managed:

Robertson (1996) noted that problems exist, which have currently not been fully resolved in achieving a suitable definition that characterises the term 'risk'. He pointed out that these problems result from

> "epistemological arguments that cut across the whole of the 'risk' assessment field...[that]...have created profound and as yet unresolved disagreements between physical and social scientists as well as within the various areas studied by social scientists themselves" (p. 3).

Robertson (1995) pointed out that these disagreements between the physical and social scientists have become centred on the contention of the physicals scientists that there is a distinction between perceived and actual 'risk'. Thus the 1983 Royal Society Study Group Report entitled *Risk Assessment*, which detailed the views of physical scientists on the matter, contended that even though

> "the uncertainties involved in risk assessment cover a wide range of types; some are quantifiable by established techniques...." (p. 38).

Hence the physical scientists (practically all of whom were inculcated with a structuralist paradigm), who made up the study group and compiled the report, held to the view that is possible to objectively assess and quantify 'risks', even though they appreciated that in some cases obtaining appropriate data would be a difficult task. Their work represented and "exemplified the international orthodoxy of the time on the subject of risk, and became a major work of reference" [Adams (1995) p. 7]. However, as was noted previously, when the Royal Society invited social scientists to participate with physical scientists in compiling the 1992 report on risk, it resulted in the aforementioned unresolved disagreement about its nature and meaning, whose root cause appears to have been the fact that there were fundamental conceptual and methodological (i.e. paradigm) differences between the various participants. The study group working for the 1983 report had not faced this problem since it contained only physical scientists. The dominance of the structuralist paradigm in the field of 'risk' assessment had stemmed from the fact that the structuralist approach, being reductionist and deterministic, appeared to enable 'risk' assessors to objectively identify and quantify 'risk'. This domination also existed and continues amongst social scientists. Within the technological hazard field this domination was almost total and within the social and natural hazard fields the 'structuralist' paradigm also dominated. However, as was noted earlier, a significant minority within the latter fields, such as Watts (1983) supported the interpretivist paradigm and did not recognise the validity of the approach of the structuralists in their field.

However, this does not mean that the structuralists approach, the orthodox approach in industry, is necessarily wrong. Rather, it is simply that the structuralists and interpretivist approaches are competing paradigms and so neither accept the validity of each others position.

Tiemann (1987) p. 11) pointed out that "approaches to technological risk management depend on the society and times in which risk questions arise". I would argue that this statement may be applied to all forms of 'risk' management for the following reasons.

- As was stated earlier, it has been contended by Burton and Kates (1972) that all hazards are either natural or quasi-natural. Hence they do not accept the concept of technological hazards.
- Some researchers [for example Watts (1983)] view social hazards as maladapted hazard responses.

Thus the distinction between technological, natural and social hazards is blurred and indeed not recognised by some researchers. Hence, the inclusion of social scientists in the study group that produced the 1992 report on 'risk' reflects the changes in 'society' that had led to a "widening of [the] legitimate communications about risk" [Tiemann (1987 p. 13)].

In the context of this discussion, the term 'society' is considered to refer to the totality of social relationships around organised groups of human beings. Therefore, the incorporation of social scientists in the second of the two aforementioned study groups may be interpreted as being a result of the changes in society that had occurred during the intervening period. Also these societal changes were, in turn, a reflection of changes that had occurred in the totality of the relationships of the individuals in society during the period. The dynamic continues. Its likely influence on 'risk' management and its implications for the lead industry discussed further in section 4.3.3.

It is these, as yet unresolved, disagreements that have been responsible for the fact that it has not been possible to achieve a suitable single definition to characterise the nature of the term 'risk'. Nevertheless, for the purposes of this discussion it is necessary to provide a working conceptual framework that identifies its nature and which is acceptable to all participant within the 'risk debate. To achieve this, it is necessary to view 'risk' as being a polythetic entity. Therefore currently it can only be characterised by a definition that is also polythetic. "A polythetic definition is one which is composed of a chain of items that shares features with their neighbours at either side but which lacks any essential individual feature" [Rayner and Cantor (1987 p. 5)]. Such a polythetic definition would be able to encompass both social concerns about factors such as perceptions of trust in those managing the risk and equity of risk sharing at one end of the chain and engineering type concerns such as probability measures at the other.

However, the use of a polythetic definition is merely an artifice to provide a framework for the discussion. It does not overcome the fact that the field is divided by methodological, epistemological and other conceptual differences. It has already been explained that the distinction between the natural, technological and social hazard fields is artificial. Therefore, I would argue that either a merger or the development of a new paradigm, which is able to encompass elements of both is imperative if risk management is to become fully effective. However, as has been pointed out, the former option would almost certainly run into the sort of problems the Royal Society faced for its 1992 report on risk, due to the presence of the competing paradigms in the field. The alternative, the development of a new paradigm is, therefore, in the author's opinion, the only viable option for a way forward from the impasse.

4.3.1.2. The relationship(s) of the person(s) and/or organisation(s) involved in managing the risk to the risk itself:

Hood et al. (1992) pointed out that the participants in risk management do not stop with the activities of private individuals and private institutions (such as corporations, associations and insurers) but also include a public risk management dimension. The relationship of each of the participants in risk management, to the risk itself, will be dependant upon the scheme(s), if any, within which they are involved. Also public risk management activities overlap and interact with those of private individuals and private institutions. Since, the point(s) where the distinction between private and public risk management should be drawn is subject to debate (Hood et al. 1992 p. 139), the question is discussed in some detail in section 4.3.3.

Hood et al. (1992) identified nine types of players in public risk management which are differentiated both at the territorial level and with respect to the type of public institution involved (Table 4.2). In public risk management regimes, the nine basic types of players could, in principle, be combined in over 500 different ways. However, the manner in which risk management functions are allocated, in practice, varies from one type of situation to another as well as from one country to another. This is because the process of allocation involves local, national and international politics and multiple levels of organisation that range from the international or supranational, through national government level, to the subnational level [Hood et al. (1992)].

Territorial level	**Institutional type**		
	‘core executive’ bodies	‘independent public bodies’	‘private or independent bodies’
Supranational	(1) example: EU[a] Commission	(2) example: EU[a] Court of Justice	(3) example: Greenpeace
National	(4) example: National Parliaments and Ministerial departments	(5) example: national courts and independent regional/local statutory bodies	(6) example: national associations of insurers
Subnational	(7) example: state or local governments	(8) example: independent regional/local statutory bodies	(9) example: local firms & activists

Notes:

[a] European Union

[Source: Hood et al. (1992)]

Table 4.2: Nine types of institutional player in public risk management, with examples

Each of the participants involved in any public risk management regime occupy positions that result in relationships specific to the regime. However, the general nature of their relations are denoted by the type of schemes employed by the private individuals, private institutions and public institutions themselves. Public risk management schemes in the US and UK have tended to follow this pattern. This pattern also exists currently with the lead industry, as illustrated in Figure 4.1.

The diagram focuses on the UK based Britannia Refined Metals Limited (BRM) who are a wholly owned subsidiary company operating under the umbrella of MIM Holdings Limited. BRM are a major producer of refined primary and secondary lead and lead alloys and primary silver, whilst MIM have global interests in lead and various other non-ferrous metals. Both are also the subject of case studies in chapters 9 and 10. (More detailed descriptions of their activities are provided in these chapters.) Note, that the structure is hierarchical, in the sense that lower scales (i.e. levels) of risk management (the smaller rings) are subsumed successively into the higher scales (the larger rings). For example, BRM operates to 'MIM Safe' standards, which, though implemented at the BRM scale, are designed to bring control of workforce occupational risks, throughout the MIM group of companies, under a uniform set of standards. The background shading for the 'lead industry sector' indicates that from the global scale downwards, the risk management hierarchy in the example shown interacts with similar hierachies throughout the industry.

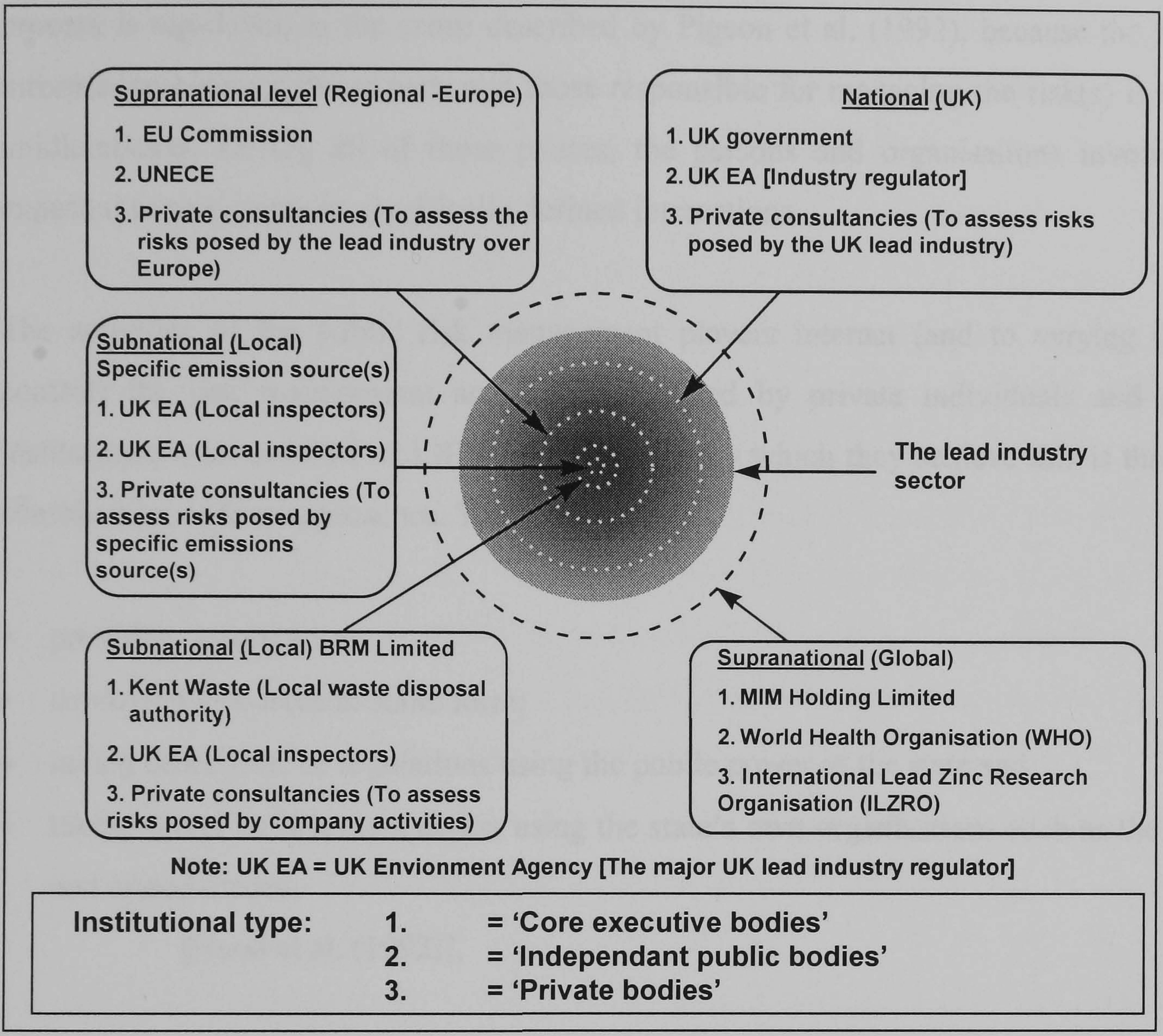

Figure 4.1: The Major Institutional Players in the Public Risk Management of the Activities of MIM Holdings Limited and Britannia Refined Metals Limited (BRM)

Historically, public risk management schemes have tended to be technocratic, twin phased and top-down in their approach The twin phased approach to risk management separates risk assessment from risk. The two phases also have tended to be separated both temporally (with the assessment phase being carried out first) and spatially (the two phases are often conducted out by different people and sometimes at different locations). It is technocratic because the assessment step is generally considered to be a predominantly scientific process consisting of risk identification, estimation and evaluation and is normally carried out by groups who are considered to be experts. The process is concerned primarily with the establishment of probabilities upon which decisions may be made. Management activities i.e. those involved with planning, resourcing, controlling and monitoring to meet legal, political and administrative requirements are restricted to the second phase. The process is top-down, in the sense described by Pigeon et al. (1992), because the flow of information between the experts and those responsible for managing the risk(s) is usually unidirectional. During all of these phases, the persons and organisations involved are expected to participate in specifically defined interactions.

The activities of the public risk management players interact (and to varying degrees control) the risk management activities conducted by private individuals and private institutions, such as MIM and BRM. The manner by which they achieve this is through a combination of four approaches. These are:

- providing information;
- deploying resources in some form;
- laying down laws or regulations using the public power of the state and
- taking direct enforcement action using the state's own organisations such as the police and armed forces

[Hood et al. (1992)].

Within the industrialised states, in some fields (notably occupational health, safety and hygiene and environmental protection) public risk management schemes are well developed and enforced. Therefore, in these fields, the risk management activities of private institutions and individuals are largely controlled by the public risk management players. The control is achieved by the selective utilisation of the four approaches available to each of the players. This allows for a variety of means to be adopted to identify and manage individual risks. Thus at one extreme, public risk management players may achieve their objectives by persuading private individuals and institutions identify and manage the risks themselves. At the other extreme, they may adopt a more directly coercive role. This may be achieved for example through the specifying of codes of practice, approved technologies and/or techniques and backing this up with legislation, regulations and enforcement actions.

This latter approach is used by the UK Environment Agency through its adoption of Integrated Pollution Control (IPC). Its aim is "to prevent where practicable, or minimise, the pollution of air, land and water by potentially polluting industrial processes so as to achieve a high level of protection for the environment as a whole" [Murley (1996)]. Under this scheme, operators of such processes have to apply for a permit (known as an authorisation) from the enforcing authority (the Environment Agency). The authorisations include emission limit values for certain substances. To obtain authorisation, IPC controlled processes are also expected to conform to what the Agency considers to be the best available techniques (BAT), although local environmental conditions, geographical location and other technical considerations are taken into account. To ensure compliance with the authorisation conditions, process operators are expected to provide regular monitoring data for the Agency and to inform them of any significant releases and changes to the authorised processes. The Agency can and does commission some of its own monitoring of these processes. However, it also relies on the monitoring data of the process operators and expects them to carry out self policing. IPC authorisation is one of the more directly coercive schemes of the public risk management schemes of the Agency. However, it is just one of many schemes from just one of the players that interact with the every day functioning of private institutions and private individuals.

The resources available to the public risk management players are also directed at those being managed with an intentionality which varies from, at one extreme of being entirely 'active' or 'corrective' to the other extreme of being entirely 'passive' or 'preventative'. 'Active' (and 'corrective) approaches are focused on modifying the sources of risk. Examples would be hurricane seeding and slope drainage schemes to limit land slides. In contrast, 'passive' (and 'preventative') approaches are focused on the effects of the risk, examples are financial compensation and evacuation [Hood et al. (1992)]. The actual approaches adopted by the risk management players are a balance between these two extremes of attitude.

Thus, the enforcing authorities have at their disposal a variety of tools, ranging from persuasion to coercion,

> "to steer the behaviour of [the] management [of private institutions], where they compete with commercial and other pressures on corporate policy and practice" [Horlick-Jones (1996) p. 146].

There has been an ongoing debate over where the distinction between public and private risk management should be drawn. The author contends, one of the underlying reasons for this is that public risk management has to compete with other pressures, such as public safety policy, and so "emerges as a political compromise between calls for strong regulation and arguments that over-regulation threatens wealth creation and suppresses the market's self-regulatory 'hidden hand'" [Horlick-Jones (1996) p.146)]. Therefore, the debate focuses on where the compromises should be made. Some also claim that public risk management, through its coercive tools of regulation and legislation, is inherently susceptible to 'rent seeking' behaviour. This consists of the capture of the state's legislative and regulatory powers by those who are able to organise effectively at low cost. The result, is that they are able to transfer resources of the state to themselves, at the expense of groups who do not have the competitive advantage of such low-cost organisation [Hood et al. (1992)]. Thus, the crux of the argument is that public risk management approaches are susceptible to distortion with some risks being dealt with preferentially i.e. at the expense of others.

There is also an ongoing policy debate, at the territorial level, over the extent to which the public risk management players should be insulated from the 'core executive structures'. Thus, they may lie anywhere along the spectrum ranging from being an integral part of the 'executive' to being fully autonomous i.e. 'independent' of it. (Table 4.2.) The current tendency is for these players to be 'independent public bodies' [Hood et al. (1992)].

Private individuals and institutions may also find it necessary to identify, assess and manage risk for which there is no guidance, regulation or control by any of the public risk management players. In such circumstances, the approach may be informal and ad hoc. For example, risk reduction practices may be passed on by word of mouth between private individuals, private institutions and between private individuals and private institutions and vice versa. However, some private institutions may choose to integrate these additional risk management practices more formally and systematically within their management structures.

4.3.1.3. The conceptual paradigm(s) within which the participants are operating:

Kuhn (1970) noted that the term paradigm may be used in two senses.[8] Thus

> "on the one hand it stands for the entire constellation of beliefs, values, techniques, and so on shared by the members of a given community" (p. 175).

Whilst

> "on the other, it [also] denotes one sort of element in that constellation".

In the former sense, a paradigm is what

> "the members of a community share and, conversely, a ....community consists of [people] who share a paradigm" [Kuhn (1970) p. 176)].

Hence, communities and paradigms are interdependent.

The latter sense, is concerned with the idea that a paradigm is an exemplar i.e. it is an example that can act as an model, which enables those who share the paradigm to see situations as like each other. Thus, people who share a paradigm in the first sense share paradigms in the second sense as well.

It was noted in section 4.3.1.2 that, on an organisational level, the participants in risk management consist of the public risk management players, the private institutions and private individuals. Members of any organisation will contain attitudes, beliefs, etc. (i.e.

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[8] Even though Kuhn (1970) used the term 'paradigm' with reference to the 'scientific' community, the concept may be applied to all communities. Thus, the lead industry and the companies within it are considered to be communities in the Kuhnian sense.

concepts) as well as affiliations and technologies that tie them physically and emotionally to the organisation (i.e. institution). Therefore, "institutions confer identity" on individuals [Douglas (1987) pp. 55-67] since they make them part of its community. (Hence, institutions also act as communities.) This process of institutional identification is necessary for individuals so that they become part of it. If it did not happen, they would not be consenting their legitimation to the organisation. They would also be rejecting its community and would, as a result, be treated as outsiders. Thus, as a result of the process, people within organisations become inculcated with some of their paradigms. This is undoubtedly the case within all industries, including the lead industry.

However, members of organisations are also individuals. Therefore they will also have other beliefs, attitudes etc. (i.e. paradigms) which will cause them to be members of other communities at the same time. Thus, both communities and paradigms may be viewed, not only as interdependent, but also as multidimensional entities that overlap one another.

As a result of the process of legitimation, public institutions tend to behave in a conservative manner since they are servants of the wishes of the state. To behave otherwise would be to reject the legitimacy of their organisation since they would be rejecting their role as state servants; which is to carry out what they perceive to be the wishes of society. This contention is supported by Douglas ((1987) pp. 9-19) in her claim that "institutions cannot have minds of their own". Thus their paradigm, and hence their rationality, is bounded within their organisations.

When constructing the framework outlined in section 4.5, this is borne in mind. Therefore, the structure of the strategy, approaches and tools have been designed to ensure that those in the industry, who are likely to become actively involved in using them, perceive them to be valid, acceptable and useful. To achieve this, it has been necessary to design them around the paradigms extant within the industry. This has been possible, since the author has had a number years of experience of studying and working within the industry. Therefore, he has been able to develop at least a partial insight into the ways people in the industry tend to think and respond to the environmental challenge. The axiom has been, that the strategy, approaches and tools should be designed around the needs and rationalities of the lead industry, rather than try to coerce the organisation to fit the model.

Max Weber argued that there are two main bases for organisation in modern society. These are rational-legal and charismatic authority. He argued that rational-legal authority, is the basis of 'bureaucratic' organisation; where 'bureaucracy' is defined as a hierarchical structure of office-holders in which decisions are based on impersonal rules. As a result, "bureaucracies tend to be order-oriented organisations which are concerned with stability and predictability i.e. they are essentially risk averse". On the other hand, "charismatic authority is based on the personal authority or charisma of a leader". Such organisations are associated with considerable fluidity, spontaneity, creativity and thrive on and foster risk [Alaszewski (1996) p. 482)].

Thus, of the nine types of institutional player shown in Table 4.2, the 'core executive bodies' and the 'independent public bodies'[9] are bureaucratic organisations since they are all servants of their respective masters. Hence, they are all conservative i.e. risk averse in their outlook. In public risk management, only with the 'private or independent bodies' do their members, in some cases, consider it acceptable to allow charismatic authority. An example of such an organisations is Greenpeace, where charismatic authority, and hence risk taking, has been a key ingredient in its success. Private institutions (such as corporations, associations an insurers), are also essentially bureaucratic in nature. Therefore, their approach to risk management is also largely risk averse. However, as has been explained, in the commercial environment risk management has to compete with the commercial and other pressures on corporate policy and practice. Therefore, within their approach and to varying degrees depending on the nature of the business, there are elements of risk taking. However, even in such circumstances, businesses try to minimise the risks that are posed. Therefore, they are still behaving as risk averse bureaucratic organisations.

It has been explained that individuals working within organisations will tend to support the attitudes of the organisations to which they belong because of their need to legitimate them. However, private individuals are not so constrained. To explain how individuals respond to various kinds of risk in society Wildavsky, Dake, Douglas, Thompson and Rayner developed the 'cultural theory of risk'. This argues that risks are culturally biased phenomena whose perception is "highly influenced by socially embedded values and

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[9] Despite the fact they are nominally 'independent' they are still public servants and, therefore, are still expected to serve what they perceive to be the wishes of society.

beliefs" and that "choices of risk taking and avoidance are made selectively on the basis of what 'way of life' or 'world view' an individual adheres to". From this position, cultural theory hypothesises that "people fear various things and perceive different kinds of danger depending on their 'cultural bias'" [Boholm (1996) p. 527)]. This bias, they argue, results in four types of 'world view'. These are the 'hierarchist', the 'individualist', the 'fatalist' and the 'egalitarianist'. (Some also add the 'isolationist'.) The 'cultural biases and the corresponding 'world views' of nature, plus examples, are shown in Table 4.3. The shaded cells on the diagonal of the matrix indicate where people adhere to their cultural bias. The other boxes show other possible combinations.

The theory makes it possible to identify the servant of bureaucratic organisations as containing mainly heirarchists. However the key feature from the perspective of risk management, is that it "provides a typology for organising responses to uncertainty" , with each of the cultural theory types being "likely to hold a distinctive view of how, or whether a risk might be managed better" [Adams (1995) p. 208)].

It needs to be noted however, that cultural theory has been subject to some criticism. For example, it is contended by Boholm (1996) that an individual's 'way of life' is not located in his or her mind but is instead within social contexts. Also, it has been argued that the empirical results of tests do not match the claims that have been made for the theory. From an interpretivist standpoint, the theory could also, of course, be criticised for being too reductionist and deterministic.

Whether or not one accepts the validity of the 'cultural theory' typology, for identifying the cultural biases of people and organisations and how they might perceive and respond to risks, it does provide a useful insight into the diversity of possible world views. It also indicates how such views can fundamentally influence the way risks are perceived, assessed, communicated and managed. Such diversity of viewpoints would also need to be catered for in any strategy, which is to be optimally effective in helping the lead industry meet the challenges ahead.

Cultural bias[a]	**Egalitarianist**	**Fatalist**	**Hierarchist**	**Individualists**
Ephemeral	communard	Hell's Angel	bankrupt Lloyds "name"	bankrupt venture capitalist
Capricious	Ned Ludd, Lottery winner	Somalian refugee	BBC weather forecaster	gambler & lady luck,
Perverse/ tolerant	Friend of Earth government adviser	rebel without a cause (James Dean)	ambitious civil servant,	venture capitalists in bear market
Benign	Utopian socialist on kibbutz	Calvinist, religious martyr	contented civil servant	venture capitalist,

Notes:
[a] Views of nature i.e. 'world views'

[Source: Modified from Adams (1995)]

Table 4.3: Cultural theory: an elaboration, with examples

In summary, it may be argued that the conceptual paradigms within which the participants (individuals and organisations) are operating, are framed as much by attitudes and affiliations as by knowledge and understanding of the risks. Indeed, since paradigms fundamentally colour one's perceptions and views, they also fundamentally influence one's whole approach to the management of risks. It was noted in section 4.1 and 4.3.1.1, that the dominant paradigm within public and private institutions is structuralist. It has also been demonstrated that most of these institutions are also bureaucratic. In such institutions, the emphasis is on formal rigid structures since they are concerned with stability, predictability and avoiding risks [Alaszewski (1996)]. Bureaucratic organisations also accommodate high specialisms of function (both horizontally and vertically) with a hierarchical managerial command [Dunsire (1990)]. As a result, the dominant approach to risk management has become technocratic, twin phased and top-down since it supports and reinforces the bureaucratic structures. The bureaucratic approach is also structuralist in nature. Therefore, it is both reductionist and deterministic.

Individuals, on the other hand, are not constrained to behave in the manner of organisations. If cultural theory can be believed, then in the absence of organisational direction, their personal approach to the question of risk management will depend on their 'world view'.

4.3.1.4. Organisational approaches to risk management:

In section 1, it was pointed out that risk management is a wide ranging field and that the term has no single meaning. However, sections 4.3.1.1, 4.3.1.2 and 4.3.1.3 demonstrated respectively, that the types of risk management approaches which are adopted depend on how the risk is characterised, the relationships of the participants to the risks and their conceptual paradigms. Considerable contention surrounds all three elements. Therefore, it is not surprising that it has not been possible to reach a consensus on the meaning of the term.

Indeed, Hood et al. (1992) identified seven disputed areas in risk management (Table 4.4) and noted that these debates "reflect competing world views" (i.e. competing paradigms).[10] These areas, which they termed 'doctrinal disputes', were identified on the basis of the differences in how organisations respond to risk. However, not all of the 'world views' are evenly matched and not all of them should be (or have been) given equal weight. Indeed, much of the running in risk management policy has been dominated by one area; that of the quantificationists. The author believes that this domination of the debate by the quantificationists is one of the main underlying reasons for the claim by Hood et al. (1992) that too much of the institutionally-oriented work has been heavily descriptive or driven by consultancy concerns and too much of it is still essentially mono-cultural in its approach and implicitly embodying a single optimum approach to institutional design. The quantificationists have been able to dominate the field through their effective arguing of the plea that there is no real alternative to quantitative risk assessment (QRA) as the primary tool of resource allocation in corporate and public risk management. However, this contention has been heavily criticised by some who argue that QRA's calculative

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[10] The extent to which the seven doctrinal debates are apparent in the US and UK approaches to risk management, has been discussed further by the author in Volume 5 of the Portfolio (as Document 24). However, this document has not been enclosed with this copy of the thesis.

techniques are not simply neutral decision aids, as claimed by its proponents, but actually "define the way problems are perceived" [Hood et al. (1992) p. 160)]. Which ever side one chooses to adopt in this argument, will of course depend on the extent of one's structuralist or interpretivist affiliations.

Thus, Hood et al. (1992) have suggested that the risk management research map is like the population map of Australia, because almost everything is around the edges and almost nothing in the central conceptual area. Also, the edges themselves are more like an 'archipelago' of scattered specialisms, isolated from one another than like a single land mass.

Hopefully, this account will have left the reader with the view that the analogy of Hood et al. is a fair impression of the problems facing the field and the problems involved in trying to develop a framework for the lead industry, which is optimally effective. Indeed, the question is raised, therefore, of whether or not under such circumstances, it is possible to develop such an approach. In the authors view, it is. However, to achieve this aim, there needs to be a new paradigm which has to be able to bring together these elements within a conceptual framework mutually acceptable to all. This is a tall order, since to date, overall consensus has not been possible.

Doctrine	**Justificatory argument**	**Counter doctrine**	**Justificatory argument**
anticipationism	apply causal knowledge of system failure to ex ante actions for better risk management	**resilienism**	complex system failures not predictable in advance and anticipationism make things worse
absolutionism	a 'no-fault' approach to blame avoids distortion of information & helps learning	**blamism**	targeted blame gives strong incentives for taking care on the part of key decision-makers
quantificationism	quantification promotes understanding and rationality, exposes special pleading	**qualitativism**	proper weight needs to be given to inherently unquantifiable factors in risk management
designism	apply the accumulated knowledge available for institutional design	**design agnosticism**	there is no secure base or real market for institutional design
complementarism	safety and other goals go hand in hand under good management	**trade-offism**	safety must be explicitly traded off against other goals
narrow participationism	discussion is most effective when confined to expert participants	**broad participationism**	broader discussion better tests assumptions and avoids errors
outcome specificationism	the regulatory process should concentrate on specifying structures or products	**process specificationism**	the regulatory process should concentrate on specifying institutional processes

[Source: Hood et al. (1992)]

Table 4.4: Summary of the seven doctrinal disputes (i.e. debates) in public risk management policy

4.3.2. How Will Risk Management Need To Develop If It Is To Become Fully Effective?

The previous sections have indicated there appears to be no single meaning to the term 'risk management' and that underlying conceptual, epistemological and methodological differences are responsible for this current state of affairs. They have also shown how risk management, particularly in the socio-technical hazard field has been dominated by a technocratic, twin phased, top-down approach which has placed heavy emphasis on QRA.

However, as noted in section 4.2. despite this technocratic domination and its apparent strengths in controlling the risks of many technological systems, the shortcomings of conventional risk management techniques have also become increasingly apparent. This is evidenced by

- the growing catalogue of spectacular failures in risk management, for example Chernobyl, Challenger, Bhopal, Piper Alpha, The Herald of Free Enterprise, BSE etc.,
- the aforementioned and increasing problem of "technological gridlock".

These factors support the argument by Beck (1992) that the management of risk is not only becoming increasingly contentious but also an increasingly dominant theme. Thus, Beck has concluded it is the modern science and technology itself which have created the 'risk society' This is, he argues, because

> "in advanced modernity the social production of wealth is systematically accompanied by the social production of risks" (p.19) .

The result, he contends, is that the creation of wealth and its equitable distribution, which have been the primary concerns of the 'industrial' society, have been overtaken by the quest for safety. He contends that this quest for safety has taken primacy due to the scale and invisibility of modern 'techno-scientifically' produced risk. Thus, he contends that what was the 'industrialised society' has now reached a stage of advanced modernity and become what he terms the 'risk society'. He also contends that this process of modernisation (i.e. societal development) is becoming 'reflexive'. Thus, "it is becoming its own theme" (p.19) .

Thus a process of 'reflexive modernism' is taking place. 'Reflexive modernism' is a type of society which has a sensitivity to all aspects of modern life. It is characterised particularly by the ongoing risk problems created by it but also by reappraisal and re-evaluation as a result of constantly emerging new knowledge and information, and the resulting changing values and priorities [Healey and Handmer (1996)]. Also, the developing realisation by lay people that science and technology is limited and inevitably produces extensive and unintended harms as well as benefits exemplifies the reflexive nature of this modernity. As a result of these contentions, he maintains that for society to continue to evolve, the modernisation process must itself become 'reflexive'. Thus, Beck's theory not only contests scientific authority and the technocratic approach, but also argues that new forms of democratic interactions (i.e. 'reflexivity') are necessary.

The author believes that Beck's theory has the potential to form the basis of the much needed new risk paradigm. It is for this reason that the assertion was made in section 4.2 that this is an example of an idea which help to take a leading role in steering the process of societal change. However if it is to fulfil this role, the paradigm he has proposed will need to inculcated by others and articulated. Thus, in the terminology of Kuhn (1970), a 'risk society' community that shares the 'risk society' paradigm will need to develop. There is some evidence that this is already occurring.

Thus, for example Healey and Handmer (1996) have joined the 'risk society' community and have started to articulate Beck's paradigm through the development of the concept of 'risk dialogue'. They contend that 'risk dialogue' is a 'reflexive' learning process in the sense put forward by Beck (1992) because it is

> "one in which participants are required to critically reflect upon the preconceptions, or what have been termed boundary premises, that they bring to the process" Healey and Handmer (1996) p. 544)].

However, whilst Healey and Handmer's ideas are still at the research stage, an approach using a form of 'risk dialogue' has been proposed for adoption by the US regulatory agencies in the aforementioned US National Research Council (1996) report on risk management. The approach it proposes, represents a significant departure from that of the National Research Council's 'Red Book' of 1983 [Löfstedt (1996)]. Thus, it argues for greater public and scientific participation in the risk assessment process. It also argues that both the public and scientists should help define the problem and then both add inputs throughout the process. Additionally, it redefines the process of risk characterisation. Hence the conventional definition, that it is just a translation or summary at the end of the completed risk assessment is rejected. It is redefined as involving complex value-laden judgements and needing the effective dialogue between technical experts and interested citizens [Löfstedt (1996)]. As a result of these changes, if it were implemented, the process of risk management would no longer be a linear and twin phased but would be much more consultative in nature.

This report is particularly welcome for two reasons. Firstly, it indicates that the National Research Council, who are an influential authority in US risk management, have accepted the need for new means of risk characterisation and risk assessment. Secondly, with its proposal to use 'risk dialogue' it has the potential to develop and articulate a new paradigm. Whether or not it is able to fulfil this role will be shown with time. The report may also be seen as significant in the sense that, in widening the dialogue and scope for participation throughout the process, the approach is starting to show some degree of 'reflexivity' in the sense proposed by Beck(1992).

Thus in conclusion, Beck's (1992) theory of the 'risk society' points the way to a more effective management of risk. According to his theory, it will achieve this through the process of 'reflexive modernisation' which will continually challenge assumptions and change values. The author believes that only by moving towards 'reflexive modernism' can the distortions in the risk management field be overcome. Indeed without it, technological development including the moves towards sustainability would stagnate. However, as has already been explained society is already changing. The moves by the US National Research Council to 'democratise' the risk characterisation process are a symptom of this process. Hence if its proposals are implemented they would be an important step in the path leading towards the development of 'reflexive modernism'.

It is noted in the Introduction (chapter 1) that the lead industry holds, to a great extent, the keys to its own destiny. In the authors opinion, one of the key steps to taking an effective steering role, to guide it to this destiny, needs to be the adoption, articulation and adjustment of such an emerging 'risk society'/'reflexive modernism' paradigm. The influence that Beck's (1992) ideas have had in facilitating development of this emerging paradigm, indicates how ideas can sometimes take a leading role in steering the process of societal change. Similarly, the lead industry has the capability, if armed with an appropriate strategy, approaches and tools, to influence significantly societal change and thereby the development of the environmental challenge and how, ultimately, it is affected by it. The author considers that for this to happen most effectively, the strategy, approaches and tools need to be posited within such a paradigm. The lead industry, therefore, now has the opportunity to take a leading role in its development. The strategy, approaches and tools, developed by the author are intended, therefore, to help foster this development.

Due to its inherently reflexive nature, such a paradigm should also provide an opportunity for building approaches, which are able to overcome the aforementioned problems of lack of agreement between the currently competing structuralist and interpretivist paradigms and between the doctrinal disputes identified by Hood et al. (1992) in Table 4.4. Crucially also, it would achieve this not by trying to produce a merger, which it has been noted seems unlikely to succeed, but would resolve the differences by encouraging them to adopt this new more reflexive way of thinking. Under these new conditions, the strategies, approaches and tools, used in risk management, would be developed through an iterative process of reflexive dialogue between the various stakeholders (including the industry itself). Depending on the nature of the problem (i.e. 'argument'), strategies, approaches and tools, which in the current terminology may be described as being either more or less structuralist or interpretivist, would be adopted as required. However, the objections of the structuralists and the interpretivists would be overcome because they would be applied within this new paradigm. Thus, the approaches and tools developed in the thesis appear to be fundamentally structuralist. Indeed, when viewed from the structuralist paradigm they are. However, in time, the intention is that they will be iteratively developed through a process of reflexive dialogue with the various stakeholders and so, rather than being structuralist, will become part of this new paradigm.

Thus, in the field of risk management, a fundamental change in ideas is being argued for. However, to achieve this, people and organisations will need to see the necessity for the change themselves. Therefore, this thesis also serves as a vehicle for laying out the foundations for that change within lead industry. This is discussed further both in section 4.5, where the framework of the integrated strategy, approaches and tools is developed by the author, and in the Overall Conclusions and Recommendations section (chapter 11).

4.4. Meeting the Challenge Through 'Best' Decision Making

4.4.1. Introduction

Thus far, the philosophical and sociological underpinnings of the strategy, managerial approaches and specific tools developed in the thesis have been examined. The final element of this framework to be addressed, is how information gained from the adoption of such approaches can enable the industry to meet the challenge, through the encouragement of 'best' environmental decision making.

Meeting this challenge will require decisions to be made and acted upon not only by industry itself but also by other active decision participants, such as industry regulators and governments. Decisions currently are made and will continue to take place at different scales and to serve different purposes such as planning and operational control. Figure 4.1 illustrates some of the major scales of significance for the MIM and BRM, but the organised hierarchy for decision making, which it illustrates, is typical for the lead industry. However, whatever the scale and whatever the purpose of the decision making, the underlying premise is that decision makers want to make the 'best' decisions possible.

For the purposes of this thesis, the 'best' decision would be one which is:

- empirically verifiable,
- its underlying reasoning transparent,
- accepted by all parties, and
- it should reflect the values, preferences and needs of all of the parties affected by and/or involved in the decision (i.e. the stakeholders).

[In a retrospective sense, it should also not be proved to have been the wrong decision when examined at a later date.]

Unfortunately, however, the lead industry has inherited a poor public image which, as has been explained, may be related to the problems arising from its long history of use which concomitently lead to the widespread dissipation of lead and lead compounds into the environment. Statements such as that by Muth (1993), when referring to the attitude of the US government (see chapter 1), suggest that this poor image may have given rise to a subconsciously based lack of trust, in the industry's ability to manage its affairs, by some people in important decision making positions.

Such attitudes could lead to decision makers outside of the industry being unfairly biased against the industry. Even within the industry it could influence the way decision makers behave by making them more defensive and introspective. Nevertheless, the assumption is made that decision makers try to make the best decision possible, so they do not do this on purpose. However, it can (and the author contends does) occur due to the fact that it has become a subconscious but significant factor in their paradigm of the industry (i.e. it is part of their framing assumptions).

As such bias is thought to exist, it needs to be countered. Thus there is a need for the indutstry to make explicit an absolute committment and to take systematic action (at all scales within the life-cycle of its products) to:

- identify its human health and environmental credentials,
- identify its strengths and weaknesses against these criteria,
- pursue a thoroughly comprehensive and on-going program of self-assessment to monitor its environmental performance, the quality (i.e. relative uncertainty underlying) its data and to set out and monitor targets for improvement,
- develop empirically verifiable and transparent approaches for carrying out these functions in an on-going manner, and
- make available for public scrutiny all relevant perfomance data, so that interested citizens may be able to develop their own informed choices about the industry.

Such tasks are already being undertaken by the industry to varying degrees, through for example:

- the publication and distribution of company annual reports containing environmental performance data and
- the submission of various monitoring data onto publicly available databases by some companies (such as happens in the UK under the regulatory regime of Integrated Pollution Control (IPC)).

However, if the managerial approaches and tools developed in this thesis are applied across the whole of the lead industry (which is the authors hope), these tasks would be conducted on a significantly more systematic basis. Whilst such approaches and tools, if properly implemented, should ensure empirical verifiability and transparency of their underlying reasoning and so encourage better decision making, on their own, they would still be unable to guarantee that the 'best' decisions will occur. This is because, in current decision making contexts, both within the industry and between it and other parties (such as regulators), only some of the stakeholders are permitted to participate. The process also tends to follow the traditional top-down technocratic patterns discussed in section 4.3.1.2.

The only way to ensure the 'best' decisions are made, would be to open up the decision making process to all stakeholders and allow them to take a real and active part in defining the argument and adding inputs throughout the process. Thus, a 'reflexive' type of decision process of the manner discussed in section 4.3.3. would be needed. In such a process, the stakeholders would participate in actively defining the decision making problem and could add inputs and outputs throughout the process. It would, therefore, be a far more interactive process. It would also provide a forum for challenging preconceptions (i.e. the framing assumptions that could lead to subconscious bias) and for dealing with the uncertainties underlying decisions in a consensual manner.

However, the author appreciates that the reality of the conditions in which the lead industry operates makes such types of decision making unfeasible, currently. For example, whilst regulators of the industry in different countries need to be responsive to the needs of the business concerned, it is they and not the business who tend to take the leading role in dictating the nature of the decision making relations. Also, for many decisions linked to the operational control of plant production and of plant processes, such broader participatory processes would probably prove to be impractical and may not be perceived to be in the interests of the organisations concerned. In addition, to adopt such decision making would be a major conceptual leap (i.e. a paradigm shift in the sense discussed earlier) which many people are not yet ready to undertake.

Therefore, approaches are necessary which, though not able to solve the problem, are able to treat at least some of its major symptoms. The author has identified the following critical areas where they are necessary. Thus, approaches are needed:

- to ensure the uncertainties underlying decisions are taken into account effectively and
- to identify the framing assumptions of the decision participants.

Unfortunately, it is not possible to cure completely these symptoms using current decision making structures. However, as section 4.3.2 explains, it is possible to adopt approaches that alleviate them to a considerable extent.

4.4.2. Managing Questions of Uncertainty and Framing

4.4.2.1. The nature of the uncertainty and its influence upon decision making

Funtowicz and Ravetz (1990) have argued that uncertainty is composed of three interrelated dimensions, namely:

- technical uncertainty
- methodological uncertainty, and
- epistemological uncertainty.

Technical uncertainty corresponds with inexactness and is expressed as the variation between measurements, the variation on individual measurements during time and calculation errors. Methodological uncertainty corresponds with bias from the choice of the model and the chosen system boundaries, whilst epistemological uncertainty corresponds with ignorance (i.e. lack of knowledge on system behaviour) [Lindfors et al. (1995)]. Therefore, epistemological uncertainty is also inherent in the way the problem is conceived. The relations between these different types of uncertainties are illustrated in Table 4.5 by means of an example: the determination of the LD_{50} for a substance.

<table>
<tr><th>Example</th><th>Type of uncertainty</th><th>Comment</th></tr>
<tr><td>LD_{50} = 100 mg/kg bd w. ± 10 %</td><td>technical</td><td>10 % uncertainty in measurement (± coefficient of variation). Uncertainty in measurements are normally or log normally distributed.</td></tr>
<tr><td>LD_{50} = 100 mg/kg bd w.; 80-150 mg/kg bdw.</td><td>methodological</td><td>Bias in experimental design leading to the confidence interval being asymmetric. Methodological uncertainties often have a non-continuous distribution.</td></tr>
<tr><td>LD_{50} = 100 mg/kg bd w.; safety factor 10</td><td>epistemological</td><td>The factor applied when extrapolating to other species for which experiments have not been made i.e. the mean is multiplied by 1/10 because of lack of knowledge on how the substance reacts on other more sensitive species.</td></tr>
<tr><td colspan="3">The connection between the different uncertainties:

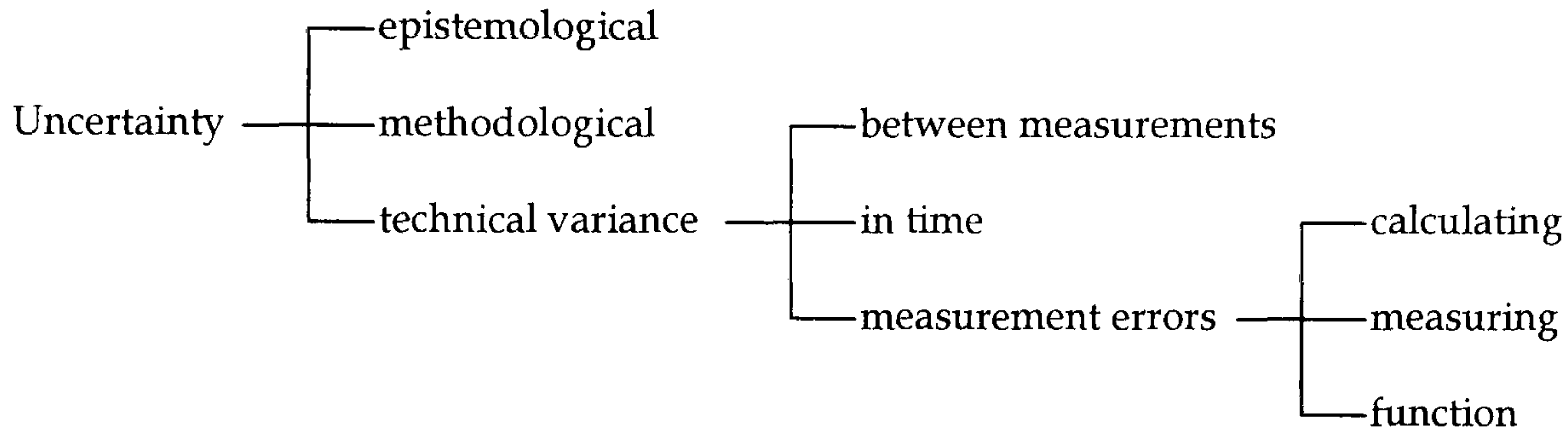

</td></tr>
</table>

Table 4.5: Relationship and connection between the different types of uncertainty (illustrated with an example from the determination of the LD_{50} for a substance) [Lindfors et al. (1995)]

Through the use of laboratory type controlled experiment, the scientific method tries to keep such uncertainties to as small a level as possible (so that within the defined parameters it should be negligible). Furthermore, variation between results from different laboratories may be reduced by using a common strain of test organism. For many problems, however, it is not possible to reduce uncertainties down to the level of laboratory experiment. To deal with these more intractable problems, consultants are often brought in to resolve them through the use of their professional experience and 'expertise'. However, for large scale problems, such as those dealing with the environment (where they are often global) these uncertainties are likely to be high and hence significant. Because of their scale, the stakes for decisions using such data also tend to be high. The author considers, therefore, that many of the decisions which are likely to result using from the assessments developed in this thesis represent examples of such decision contexts.

Despite these problems, decisions still have to be made. The question is, how can the best decisions be made in the face of such uncertainty? Figure 4.2 illustrates graphically these problems of increasing uncertainty and coping with it in decision making.

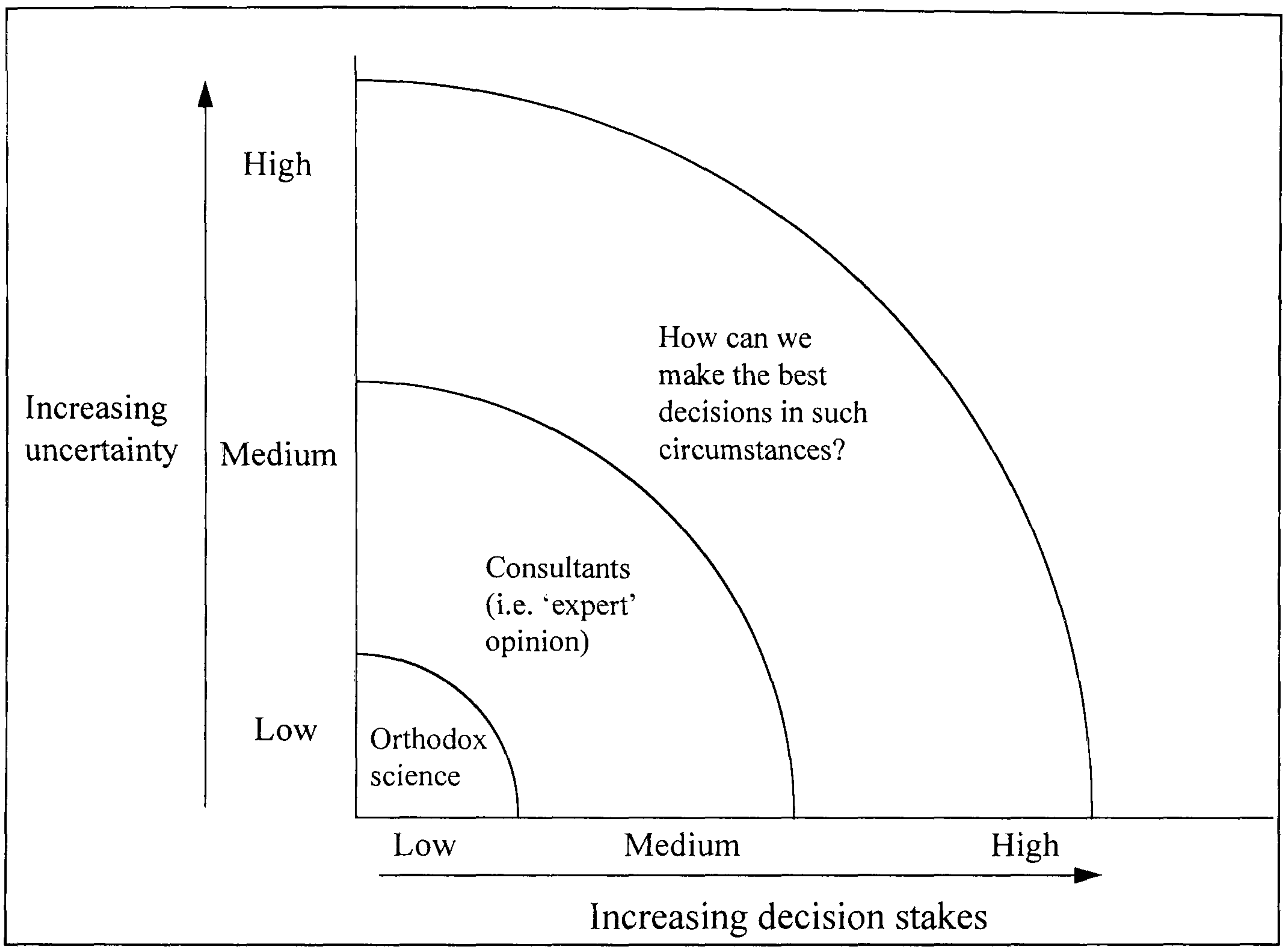


Figure 4.2: How can the best decisions be made when the stakes and the uncertainty are high? [Modified from Ravetz (1992), Ravetz (1997)]

Figure 4.2 suggests that once the uncertainty regarding decisions reaches a certain threshold, expert opinion tends to be called upon to make judgement. However, it also indicates that where both the stakes and uncertainty are high, expert opinion is not enough on its own. There are two main reasons for this. Firstly, one's ethics and views on society are implicit within all opinion, including that of 'experts', even though they may (and perhaps should) try to be unbiased in their assessments. Thus, whenever 'experts' are called upon to create sense out of uncertainty, they also imparting their own ethics and views. These will then be passed on to those relying upon their judgement. Since, they tend to be implicit rather than explicit, their mode of influence upon decisions will not be transparent. Secondly, even where experts operate from similar ethical standpoints and hold similar views on society, different experts may arrive at different opinions when asked to address similar questions, because they may frame their decision making rules and the boundaries of what should and should not be considered differently.

Despite these limitations, consultation with expert opinion, as an aid to decision making in situations of high uncertainty, should not be viewed as either wrong or as a weakness. The point, however, is that an expert judgement merely represents one opinion and it carries with it a specific set of ethical beliefs, views on society, boundaries and decision making rules (i.e. it places its own individual frame on the decision). Other opinions may be equally valid and should not be discounted simply because they are based upon different standpoints.

The danger inherent in the reliance upon expert opinion, however, is that it could be interpreted dogmatically. Thus, for example, once expert opinion has been translated by decision makers into emission limits, these could then become 'magic numbers' where they become an end in themselves (i.e. they dictate the decision rather than serve as a guide). This is thought likely to be a particular danger where reliance is placed on the opinions of only a limited selection of experts. This is because decision makers, aiming to make the best decision, will want to base their judgements on some guide (reference point). In the absence of additional input, they may be tempted to rely entirely on familiar approaches regardless of whether they are most the appropriate for the specific context.

Experts appear to create clarity from confusion. They achieve this through appearing to provide a systematic reduction of uncertainty. Indeed, the technical and some of the methodological uncertainty may be made explicit and hence be addressed by their efforts. Epistemological uncertainty, however, is inherent in the way they have conceived the problem. Because it corresponds to ignorance of the system being studied, it remains implicit and therefore still hidden. As a result, the uncertainty underlying decisions will tend to be systematically underestimated, should decision makers place reliance upon expert opinion alone.

Hence, if expert opinion alone is not enough to guide decision makers to arriving at the 'best' environmental decision in situations of high uncertainty:

- what role does it/should it have and
- what other factors could/should be brought in to improve the decision making?

The remainder of this section is devoted to considering these questions and outlining the principles and approaches which would need to be included in the decision-making process if these limitations are to be minimised in the most effective manner. In this context, specific reference is given to the manner in which the assessments provided in the case studies (chapters 9 and 10) should be used within decision making both by the industry itself, as well as by other participating parties.

4.4.2.2. Effective management of the uncertainty

In section 4.4.2.1, it was explained that for 'best' environmental practice to be achieved, the approaches adopted (and the decisions which follow from them) would need to be empirically verifiable, their underlying reasoning be transparent and accepted by all affected parties and they should reflect the values, preferences and needs of all of the parties affected by and/or involved in the decision. (Note, that such requirements have been developed in full in the 21st Report of the Royal Commission on Environmental Pollution [RCEP (1998)].

Hence, for a particular decision, it could not be viewed as the better environmental practice if the option favoured failed to meet one or more of the above criteria. When such a circumstance arises, however, it still needs to be resolved in a manner which leads to the best decision being made, given the constraints imposed by current knowledge and by other elements which the parties to the decision consider relevant. In some cases, identifying which is the better may be achieved by applying some of the approaches developed in the case studies of this thesis. However, in many cases, the relative advantages and disadvantages of different decision options are debatable, regardless of the approaches used, and thus are not so easily resolved. Such disagreements may result not just from disagreements over data interpretation but also from the fact that different parties to the decision may have framed the problem in different ways. Such framing assumptions have their basis in the ethical standpoints and the views on society which are held by those supporting the different positions in the decision making arena. Also, both experts and non-experts will be influenced by such standpoints.

According to the above criteria, if one is to make the best decisions, the influence of these standpoints must be fully reflected in the decisions which are eventually made. Because of this, the author is of the opinion that acceptance or rejection of the judgements of experts ought to be conditional upon acceptance of their underlying standpoints. Unfortunately, however, this is currently not possible, since such standpoints are generally implicit rather than explicit. Hence, there is the need for the development of methodologies which would make them explicit.

One practical means of limiting this problem would be to canvas a wider range of opinion. The regulators in some countries already show some commitment to this. For example, the UK Environment Agency are expected to "develop a close and responsive relationship its various affected parties". (Environment Act (1995) Section S4) [HMSO (1995)]). In addition, the regulators and other decision makers should be prepared to consider representations from all legitimate sources and should avoid clinging to emission limits as 'magic numbers' but try instead to use them as a guide to decision framing. However, these strategies are not necessarily enough on their own, since they do not challenge the underlying framing assumptions and, hence, do not make them explicit.

As noted earlier, the author considers that the decisions under discussion here are concerned with risk management. Where those making the decisions have a commitment to being responsive to the needs of other involved parties, it helps if this is achieved through consent thereby making it more likely to be effective. However, as noted previously, if the underlying framing assumptions of all of the parties are to be made fully explicit, a more 'reflexive' form of dialogue between all of the parties will be needed similar to that proposed by the US National Research Council (1996), as this would require them to critically reflect on their preconceptions [Healey and Hnadmer (1996)] (i.e. their framing assumptions). It may be noted that the 21st Report of the Royal Commission on Environmental Pollution (1998) also recommends essentially this approach [RCEP (1998)].

4.5. A Framework for the Lead Industry

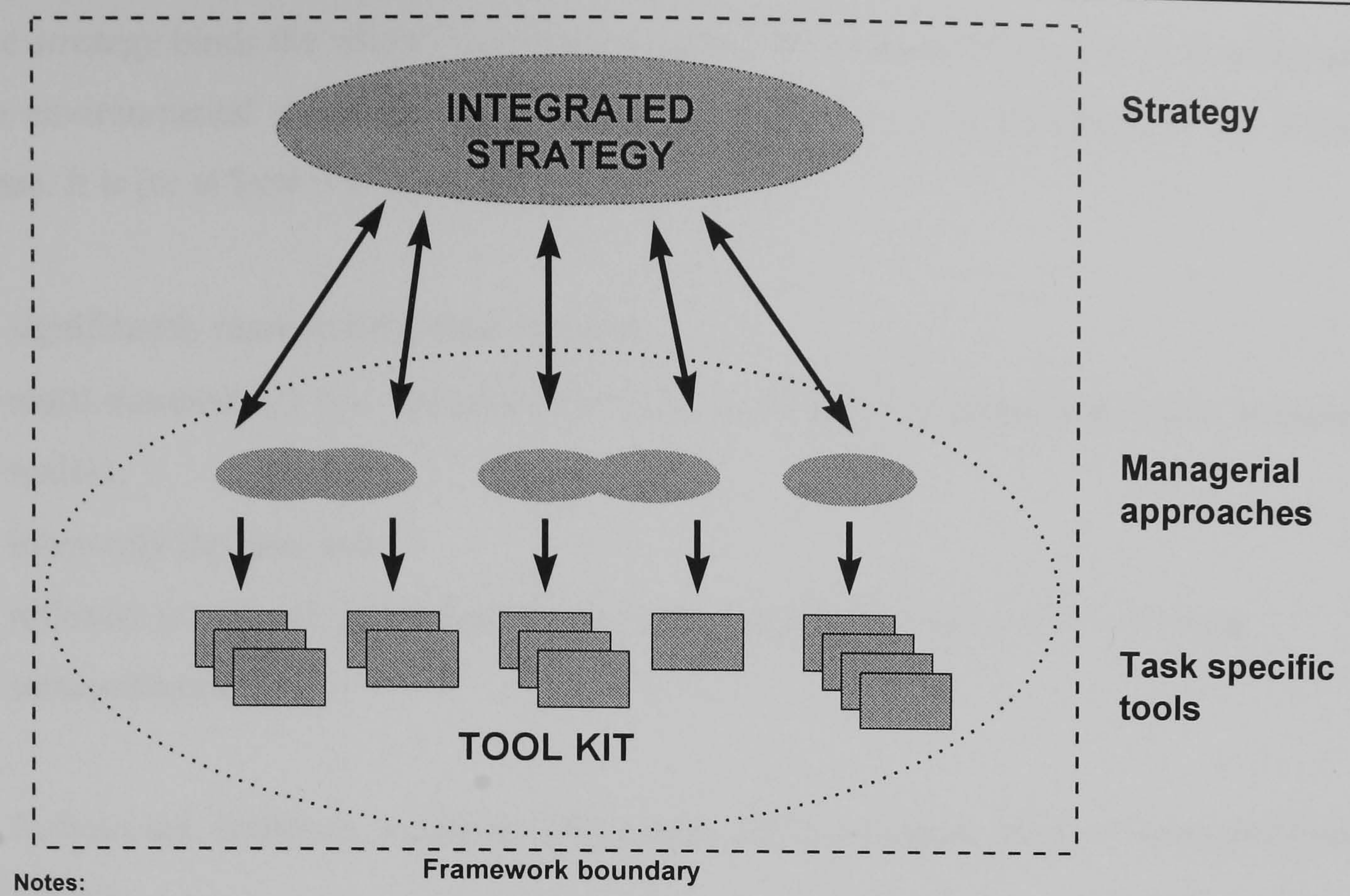
This section is, in effect, a crystallisation of the ideas developed in sections 3 and 4.

The framework comprises three interacting elements, namely:

- a strategy,
- various managerial approaches and
- various tools.

Its purpose is help the lead industry to meet the developing environmental challenge in a manner which is optimally effective. Whilst each of the individual elements may be used in isolation, they are intended to be used simultaneously. Also, whilst they already provide a complete package, the intention is that they will continue to be developed iteratively through a process of reflexive dialogue between the lead industry and all relevant stakeholders, where all elements will be subject to critical, fundamental and systematic re-evaluation and modification.

The principle interactions between the elements are indicated in Figure 4.3. Note, however, that whilst the elements have been separated for clarity, they do not represent a linear sequence, since iterative development will be expected to occur simultaneously with all of the elements.



Notes:

1. Managerial approaches may overlap to verying extents with one or more other approaches.
2. Managerial approaches may give rise to one or more task specific tools.
3. The strategy and objectives evolve through iterative execution, whilst tools may be modified or replaced.

Figure 4.3: Principle interactions of the strategy, approaches and tools within the framework

The strategy, approaches and tools are intended to form the basis for a new way of thinking about the environmental challenge within the lead industry. Thus, they are intended to serve as a starting point to a new paradigm (as outlined in section 4.3.3). Many of the elements should already familiar within the industry. This is because the approaches and tools, which have been adopted, have been adapted wherever possible from those used in the industry currently. Whilst they have been developed with specific reference to case studies (chapters 9 and 10), they are also generic in nature. Hence, they are intended to be applicable for the rest of the lead industry with little modification. They are also be applicable throughout the non-ferrous sector and for other sectors where similar types of data are collected.

The *strategy* binds the whole framework together. In essence, it is a way of thinking about the environmental challenge i.e. it is the 'conceptual box', a paradigm in the Kuhnian sense. It is (or at least it is intended to be):

- significantly more holistic than currently,
- multi-dimensional (i.e. operates effectively at different geographical and/or managerial scales),
- inherently flexible, and
- reflexive (i.e. developed iteratively through ongoing dialogue with all relevant stakeholders).

Its features are, therefore, a specific delineation of the paradigm which is being proffered in this thesis.

The *approaches* serve to translate the theory developed in the strategy into practical and proactive action at all of the scales where environmental questions need to be addressed. Therefore, the number and type of approaches that are developed is expected to vary according to the perceived needs of the time. (Since perceived needs also vary from scale to scale and from company to company, it is for this reason that the approaches provided in the case studies have been specifically tailored to their respective organisations.) In their current form, the approaches are based principally on ongoing managerial practices and are intended to provide regularly updated data sources for the various tools.

The *tools* are specific methods which have been used to produce specific assessments. The number and types of these tools is also expected to vary according to needs.

All three elements are intended to be inherently flexible. Flexibility between the various approaches and tools is achieved by basing the tools on modular data bases maintained and developed by the company managerial process. (The manner in which this is achieved is discussed in the case studies (chapters 9 and 10)).

There is a fourth element to the framework. This is the *decision making approach* to ensure that decisions made regarding the industry are on the basis of best practice. This is necessary to ensure that the industry does not suffer from biased decision making due to implicit prejudices in the framing assumptions of the decision makers. The intention, is that the approach will operate in the form of a 'reflexive dialogue' between the industry and all other stakeholders. In such circumstances, problem structuring will be conducted by the active participants and the strategy, approaches and tools would all be subject to systematic, fundamental and iterative review. Thus, the strategy would become an expression of the collective paradigm of these participants and the managerial approaches and the tools, expressions of this paradigm in active and practical terms.

Currently, however, this is little more than an aspiration. Therefore, to try to compensate, approaches have been proffered to minimise the limitations this causes to the effectiveness of the framework.

4.6. Purposes of the case studies

Case studies, based on two leading and related companies, within the lead industry have been conducted. The two companies are: MIM Holdings Limited and Britannia Refined Metals (BRM) Limited. The overall reasons for enclosing the case studies within this thesis are twofold.

Firstly, they demonstrate the capabilities of the strategy, approaches and tools (at the scales concerned and for the process chains studied) in:

- identifying their human health and environmental credentials,
- identifying their strengths and weaknesses against these criteria,
- making comparisons of the relative preferability of different options in terms of their relative potential effects against these criteria, and
- aiding decision making within these contexts.

Secondly, they demonstrate the commitment and action taking place within the case study organisations (at the scales concerned and for the process chains studied) to:

- reduce their overall potential human health and environmental impacts,
- identify areas of weaker performance against these criteria,
- target and improve these areas and to monitor this improvement, and
- develop life-cycle and site and assessment management approaches based on human health and environmental criteria.[11]

The case studies also demonstrate the use of new and modified techniques to meet specific needs.

5. Estimating the Fates, and Human Health and Environmental Effects of Emissions

The case studies contain a series of quantitative and qualitative assessments of the potential human health and environmental effects of different lead industry operations when analysed at different scales. To provide such assessments, it is necessary both:

- to identify those effects which are most relevant to the needs of making decisions about the industry at the different scales of concern and
- to consider how uncertainties in the dosimetric, toxicological, epidemiological and environmental study evidence underlying them limits the reliance which may be placed in estimations which may be made from such data..

These are reviewed in section 5.1 and sections 5.2 and 5.3 respectively.

5.1. Uncertainty and the Estimation of Fates and Effects

The behaviour of compounds emitted into the environment is summarised in Figure 5.1. Emissions have physical, chemical and biological characteristics as well as emission

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[11] Impact pathway approaches are also required. Due to time restrictions in the preparation of this thesis, it has not been possible to include a case study operating at this level. However, the need for such approaches is discussed in the Overall Conclusions and Recommendations (chapter 11).

characteristics (such as quantity and timing of the release, the initial compartment of release (air, water, soil) and their location and source type) which will influence both their fate(s) and impact(s) within the environment. These characteristics along with the physical, chemical and biological characteristics of the ambient environment also influence the dispersion and distribution and hence the fate(s) of the emissions. Whether they will produce an impact or not will also be dependent upon the magnitude, type and sensitivity of the targets to the exposure. Assessment of the potential effects of emissions, therefore, entails an estimation of these various factors which determine fate and effect. Thus, fate factors attempt to identify, measure and model the routes by which an emission effects humans or non-human biota, whilst effect factors attempt to identify, model and measure the consequent effect upon a target or community of such an emission [Clift et al. (1998)]. Emissions may, therefore, affect two different groups of targets i.e. human targets (causing human health effects) and non-human targets (causing environmental effects). Hence, these two groups are considered explicitly in this thesis.

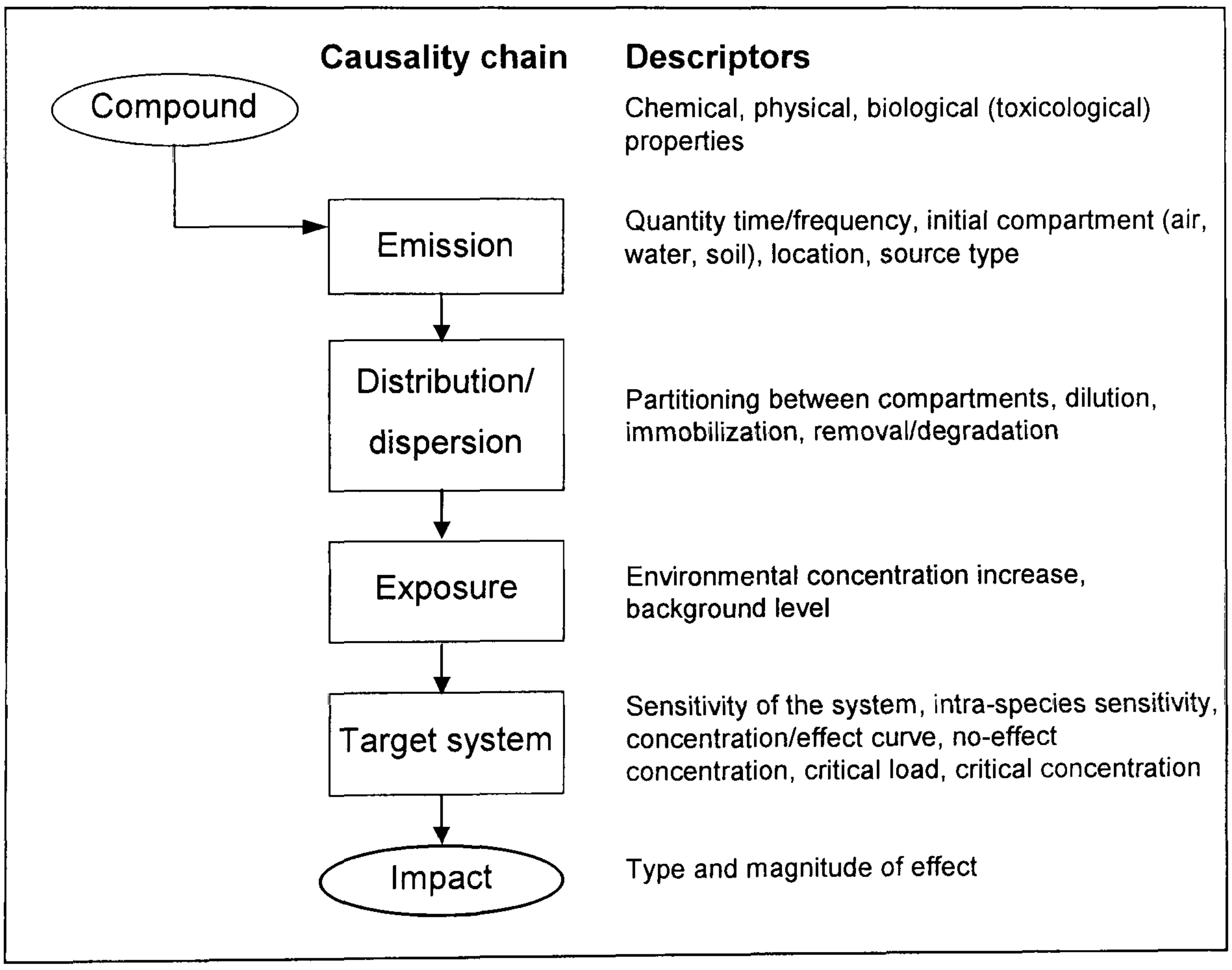

Figure 5.1: The behaviour of an emitted substance in the environment [Modified from Jager et al. (1994)]

Substances emitted into the environment will give rise to an increased concentration of these substances in that environment. These increased concentrations cause changes which may be called primary effects, and one emitted substance may give rise to several such primary effects. These primary effects may themselves cause further changes in the environment which may be called secondary effects. In turn, these may give rise to tertiary effects. In principle, the chain could be never ending. Feed backs may also occur, where higher order effects could be both the cause of lower order effects and the cause of the initial emissions themselves. The interactions between these different orders of effects are summarised in Figure 5.2.

In addition, the geographical occurrence of such effects may range from being local (e.g. acidification of a particular lake) to global (e.g. "greenhouse" warming).

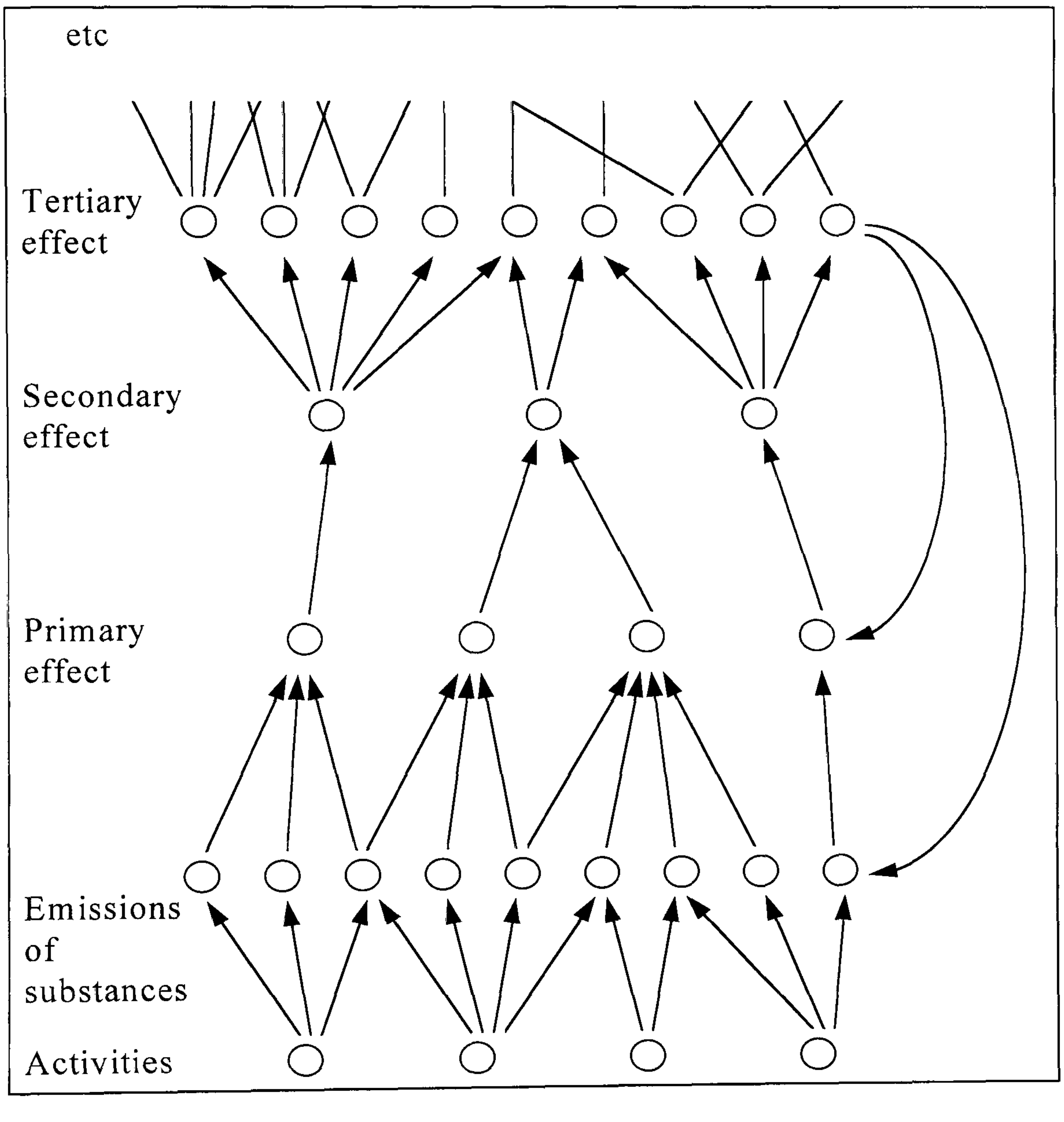

Figure 5.2: Schematic description of the effect cause-chain for emissions [Finnveden et al. (1992)]

The author considers that for the arguments set out in this thesis to be relevant to the needs of the decision makers, the potential human health and environmental effects of the pollutants need to be characterised into a defined list of commonly accepted human health and environmental problems. Assessment of the contributions to such problems requires the adoption of one or more effect endpoints which characterise that effect.

For human health effects, the selected endpoint(s) vary according to the source(s) used to provide the evidence. They are generally mortality and indicators of morbidity when based upon epidemiological evidence, and are indicators such as changes in lung function when based directly upon toxicological evidence. It is likely that such endpoints have been chosen to represent the consensus of the interested parties as to what should to be the focus of interest (i.e. the process is thought to be normative). For epidemiological studies, however, the availability of existing data may also have played a significant part in the selection. Thus, to ensure that the Report is fully commensurate with such norms, the authors have used these human health effect endpoints throughout.

For environmental effects, a variety of effect endpoints, each representing different steps along the cause-effect chain for each emission exist. Figure 5.3 below illustrates part of such a chain for a greenhouse gas emission (such as CO_2) which contributes to enhancement of the global warming effect.

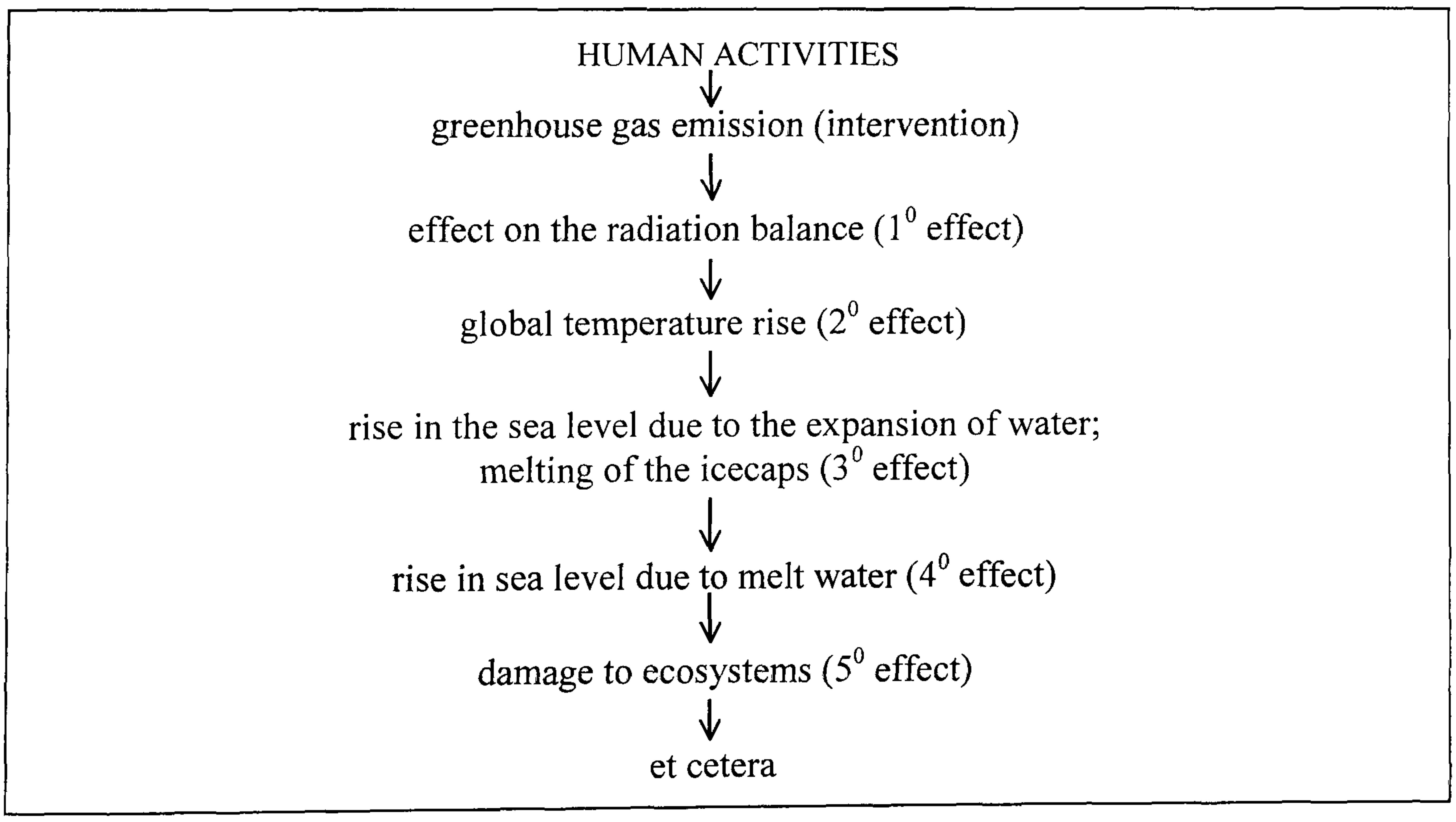

Figure 5.3: Part of the environmental cause-effect chain for an emission contributing to enhancement of the global warming effect [Modified from Heijungs et al. (1992)]

Global warming is but one of a diverse range of commonly accepted environmental effects which are thought to result from human activities. The Society of Environmental Toxicology and Chemistry [SETAC (1991)] has considered this diversity and has produced a comprehensive (and normative) list which has subsequently, been further refined by, amongst others, Heijungs et al. (1992) and Cowell (1997).

The list, according to the modification by Cowell (1997) is shown in Table 5.1.

Resource depletion	**Pollution**	**Disturbances**
Abiotic resources	Global warming	Physical ecosystem degradation
Biotic resources	Ozone depletion	Landscape degradation
Land use	Ecotoxicity	Desiccation
	Human toxicity°°	Direct victims°
	Photochemical oxidant formation	
	Acidification	
	Eutrophication	
	Radiation	
	Dispersion of heat	
	Noise	
	Smell	
	Working conditions°	

° These relate principally to human health effects, whilst the other categories related principally to environmental effects.

Table 5.1: Types and categories of environmental effects [Cowell (1997)]

According to this classification, a maximum of 19 types of potential effects are possible, each of which could be characterised by a number of effect endpoints. For example, in theory, five different effect endpoints could be chosen for global warming (as illustrated in Figure 5.3). In practice, however, when selecting the order in the chain to be used as the endpoint, the predictability of the cause-effect relationship must also be considered. This is because the uncertainties in predicting effects tend to increase considerably as one moves to the higher order effects in the chain. Thus, Heijungs et al. (1992) note that, as a principle, one should use endpoint(s) that are the lowest order at which a clear relationship with the cause-effect chain is apparent [Heijungs et al. (1992)]. In the case of a greenhouse gas emission for example (Figure 5.3), the endpoint which is normally used is its effect on the radiative balance (i.e. their primary effect) since above this order in the chain, predicting the cause-effect relation becomes too uncertain. The authors have taken these consideration into account when selecting the environmental effect endpoints used in this thesis.

With regard to the range of the types of effects listed in Table 5.1, not all are relevant to the emissions in question. Thus, where a type of effect is applicable to one or more of the emissions in question, an explanation of it is provided at the appropriate point in the text. However, those categories which are not considered to be relevant to the emissions in question are considered no further.

In summary, two groups of effects -human health and environmental effects -are identified as being relevant for consideration in this thesis. Within each of these two groups, a variety of different effects are recognised whose identification is thought to be normatively based. Each of these effects could, in theory, be characterised by several endpoints. In practice, however, selection is dictated by two factors, namely the availability of the evidence and the predictability of the effect endpoints in relation to ambient atmospheric pollutants.

These latter factors are now considered for pollutant emissions in general. Since the priorities of the regulators are first and foremost human beings, and then the living (biotic) and non-living (abiotic) components of the environment, the predictability of human health effects from atmospheric emissions in general is considered first to reflect this prioritisation. The reliability of environmental study evidence is then reviewed in section 5.3.

5.2. Estimation of human health effects from dosimetric, toxicological and epidemiological evidence

Evidence for the human health effects of emissions has been derived from dosimetric, toxicological and epidemiological studies. Emission dosimetry may be regarded as the assessment of the degree of exposure of individuals and populations to airborne emission, epidemiology as the study of the health risks associated with such exposures and emission toxicology as the study of the responses to such exposures. All three fields are, therefore, interlinked.

1. Using dosimetric, toxicological and epidemiological study data to id decision making

One of the major uses of such data is to establish the existence (or absence) of dose-response relationships between ambient concentrations of atmospheric pollutants and whether or not there is a threshold before an response (i.e. effect) is detectable. Four main types of possible dose-response relationships exist and are illustrated diagrammatically in Figure 5.4 below.

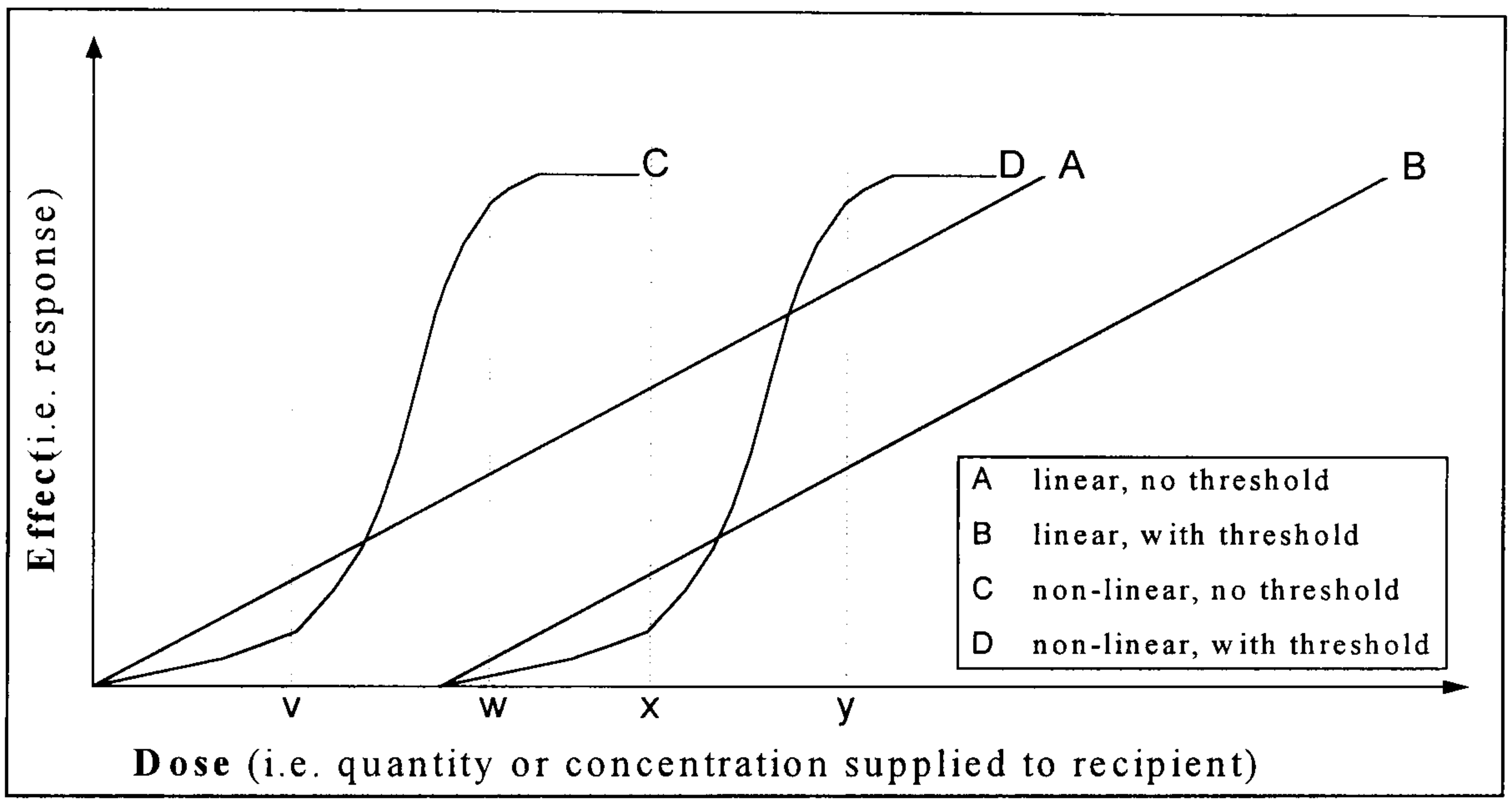

Figure 5.4: Types of dose-response relations which dosimetric, toxicological and epidemiological studies aim to establish

Where a linear relationship is found to exist (as in curves A and B), it is possible to estimate the effects of incremental changes in dose since the changes in response (i.e. the changes in human health effects) will be proportionate. If the function of the sigmoidal curve for non-linear relationships is known, similar estimations would also be possible. However, as an approximation, it may sometimes be possible to assume that the relationship is linear over part of the dose range. This is because there may be an area in the curve which is close to linear (for example between dose values of v and w for curve C and between dose values x and y for curve D in Figure 5.4) or because relatively small changes are considered so that the curve can be linearised locally or approximated by its tangent.

Once such relationships have been established, they may then be used to guide decision makers such as the Agency and local authorities. The calculation of exposure-response coefficients for individual pollutants can serve as an aid to this process. For the atmospheric pollutants of carbon monoxide (CO), sulphur dioxide (SO_2), nitrogen oxides (NO_x) and PM_{10} particulates, the Sub-Group of the Committee on the Medical Effect of Air Pollutants [COMEAP (1998)] have used such coefficients to estimate the number of deaths in the UK (all causes) and the number of UK respiratory hospital admissions (which have been brought forward) attributable to each of these pollutants.

The coefficients may also be used to compare the relative health outcomes (in terms of mortality and respiratory hospital admissions) resulting from changes in the dose. How this might be achieved is indicated in Figure 5.5 below. It illustrates the effect for three different linear dose-response relationships where there is no threshold.

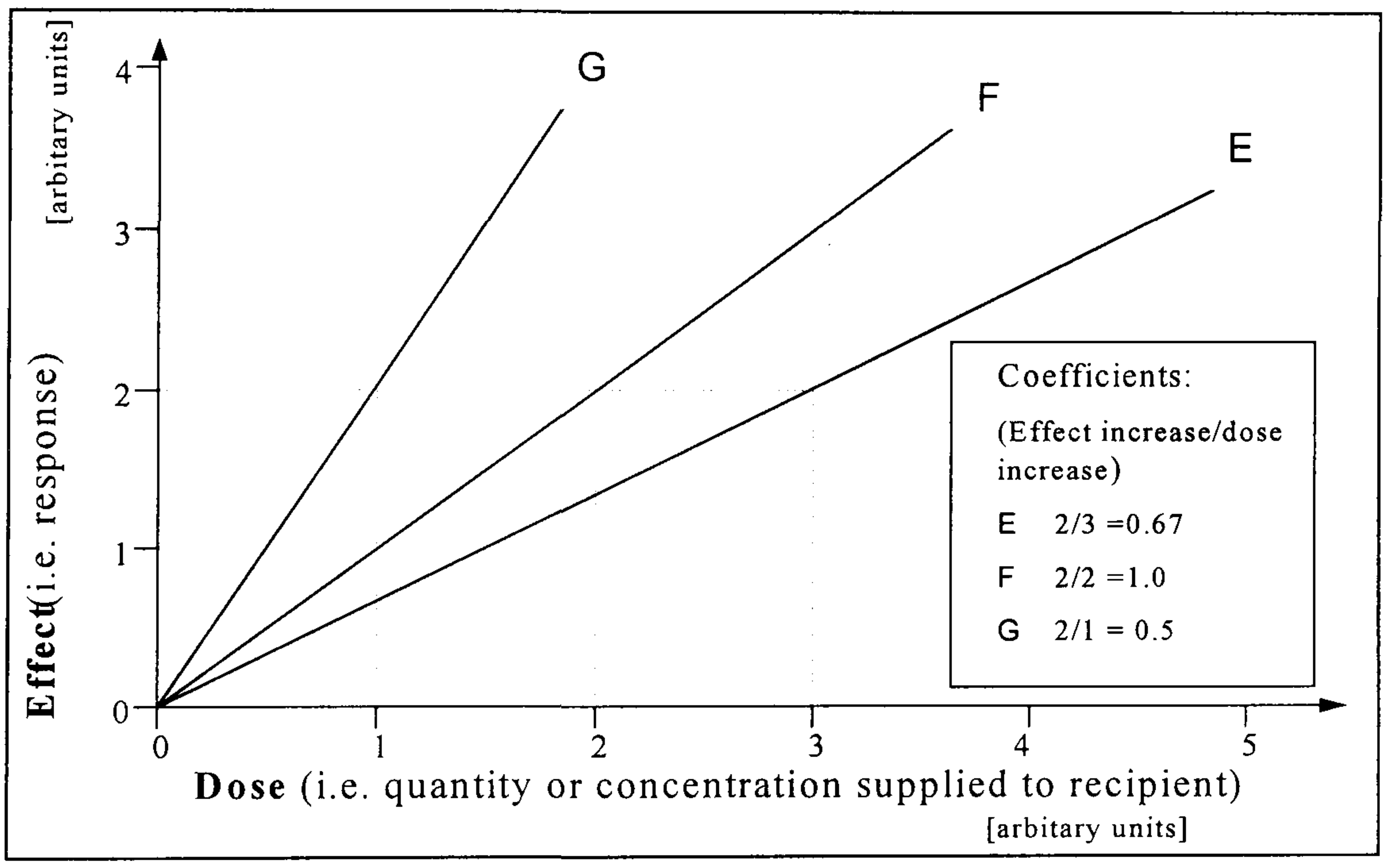

Figure 5.5: Hypothetical example of three linear dose-response relationships with no threshold for response

In the hypothetical example above, stepwise incremental changes in the dose to the recipient would lead to the greatest changes in effect for curve G, less of an effect for curve F and the least effect for curve E. Since the curves represent the dose-response relationships of different emissions, one could conclude that the sensitivity of responses to reductions in emissions would be in the order G, F, and E, with G being the most sensitive. Such arguments would also apply equally to the other types of dose-response relationships illustrated in Figure 5.5.

2. Strengths and limitations

With respect to the general differences between epidemiology and toxicology, Nelsen (1988) notes that the former is the study of human experience of the substances in question, under conditions as well controlled as circumstances permit, whilst the latter is concerned with laboratory experimentation. The main purpose of toxicological investigations, is to secure information supportive of the estimation of risk from one or more external agents [Nelsen (1988)]. (In this thesis, these agents are the emissions from the lead and associated industries.) Because of ethical considerations, toxicological data is derived predominantly, though not exclusively, from animal studies. Thus, one of the major difficulties of toxicology is the handling of the uncertainties resulting from differences in species responses to agents and in extrapolating such data to provide risk factors for human beings. Epidemiological studies, on the other hand, do not suffer this limitation, since they are studying human impacts directly.

Toxicology attempts to translate the results of laboratory investigation to human populations. However, in contrast to epidemiological investigations, where the study population can be very large, the study population will always be limited to the size of the laboratory study group. Thus, anatomical and physiological factors such as gender, age, physique, disease and inter-subject variability, which influence recipient uptake, may limit the reliability of assertions from such evidence. In addition, ambient exposures are normally to complex mixtures of pollutants. Therefore, their combined effects might be synergistic or possibly even antagonistic, However, toxicological studies are normally conducted with single substances or simple mixtures, so such potential phenomena are not accounted for. Thus, the sensitivity of toxicological studies is generally poor.

For different reasons, the sensitivity of epidemiological studies is also poor. The chief cause for this resides in the limitations in the ability of epidemiological investigations to control variables. In contrast, in toxicological investigations this is usually quite straightforward since confounding variables[12] may be readily isolated from the investigation. Therefore, probable causality may be strongly suggested. In epidemiological investigations, however, controlling variables is much more difficult. Therefore, confounding variables can often only be accounted for with extremely careful planning and even then they are rarely handled with complete success. Thus, the establishment of probable causality is often extremely difficult (if not impossible) in epidemiological field studies [Nelsen (1988)]. Table 5.2 summarises some of the major limitations and advantages of the two fields.

Limitations and advantages	**Toxicology**	**Epidemiology**
Relevance	Uncertain	Excellent
Control of variables: exposure, environment, confounding factors	Excellent	Poor
Identifying causal factors	Excellent	Poor
Size of population	Limited	Can be very large
Sensitivity	Poor	Poor
Genetic diversity	Normally deliberately narrow	Broad
Intercurrent disease	Controllable	Not controllable
Study of mechanisms	Easily assessable Uncertain relevance	Ethical hindrances Directly relevant
Diagnostic tests	Unrestricted (except interview)	Severely restricted

Table 5.2: Comparison of the limitations and advantages of toxicology and epidemiology [Nelsen (1988)]

The poor sensitivity of both epidemiological and toxicological investigations is a consequence of the significant methodological and epistemological uncertainties which underlie them. The authors contend, therefore, that they are pursuits which fall beyond the confines of orthodox science, as defined by Ravetz (1997) (This question, the nature of this uncertainty, how it frames questions and its effective management in decision making have been considered in section 4.3.3 of chapter 4.)

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[12] A confounding variable may be defined as follows:

> If a variable x is regarded as the cause of an outcome y, a confounder is a variable which is related to both x and y but is not part of the causal pathway between x and y [COMEAP (1998)].

It has also been argued that whilst it is often possible to produce measures of association such as correlation coefficients and standardised regression coefficients, in general such epidemiological associations do not reflect any causal parameters [Morgenstern and Thomas (1993)]. In an attempt to overcome this problem of establishing causality, an evaluation procedure was developed by Bradford Hill (1965). This attempts to evaluate it on the extent to which studies are able to meet the following nine criteria of association, as shown in Table 5.3.

Criterion:	**Comment:**
strength	Is the magnitude of the association large?
consistency	Is the association observed repeatedly by different persons in different circumstances and places?
specificity	Is the pollutant associated with disease specific mortality or morbidity?
temporality	Does cause precede effect?
biological gradient	Is there a dose-response curve?
plausibility	Is the association biologically plausible?
coherence	Data should not seriously conflict with generally accepted knowledge
experiment	Can it also be demonstrated experimentally?
analogy	It may be fair to judge by analogy with other similar effects.

Table 5.3: The Bradford Hill criteria for the establishment of causality in epidemiological studies [Bradford Hill (1965), Gamble and Lewis (1996)]

Whether such an approach is able to demonstrate causality or merely the strength of the association is debatable. Therefore, it could be argued that the most prudent approach would be to reserve judgement. It is believed that this follows for toxicological studies too, since they also suffer from poor sensitivity.

Epidemiological investigations of the effects of atmospheric pollutants have focused on their chronic and acute effects. Acute effects relate to short-term exposures to high concentrations. Acute effect studies aim to identify and describe any relationships which may exist between same-day or previous-day levels of air pollution and are concerned principally with effects upon population morbidity. Chronic effect studies aim to identify the effects of long-term exposures to pollutants (particulates) in encouraging the development of disease (morbidity) and in increasing the death rates (mortality) within communities. Chronic effects studies, therefore, relate to exposures which may take place over many years (i.e. they are long term).

As has already been noted, the key problem limiting the sensitivity of epidemiological investigations is the influence of confounding variables. In acute effects studies, confounders such as temperature, relative humidity and weather need to be allowed for. In chronic effects studies, similar confounders exist. However, the problems caused by them are considered to be even more difficult to handle than in acute effects studies [COMEAP (1995)]. Also, other factors, which tend to be poorly recorded, such as life-time experience of acute illness, the availability of medical services and smoking habits can influence the development of chronic disease. Therefore, these factors can create additional problems with the interpretation of chronic effects studies [COMEAP (1995)].

Problems also exist due to the fact that the endpoints of different studies tend to vary. Thus, whilst mortality studies use by definition the same endpoint (death), in morbidity studies it may differ quite considerably (for example increases in hospital admissions, increases in general symptoms and changes in the incidence asthma cases have all been used). Therefore, the question of the transferability of results across different endpoints is raised. In addition, even where the outcomes of studies are the same, there may be differences between studies, between geographical regions, between subgroups within studies and in how the endpoint itself is assessed.

Thus, to summarise, the dosimetric, toxicological and epidemiological evidence for human health effects from atmospheric pollutants suffers from low sensitivity. It is also debatable as to whether or not it is able to show causality and there are doubts over the transferability of the conclusions between different studies.

Despite these limitations, such studies have provided evidence for potential human health effects from atmospheric pollutants present at current ambient UK concentrations. For those emissions deemed significant in the case studies, this evidence is considered in chapters 9 and 10 for the individual emissions and for when they occur combination with each other and with other ambient pollutants.

5.3. Estimation of environmental effects

Assessment and estimation of the environmental effects of emissions has been conducted using direct and indirect measurements and modelling studies.

Direct measurement approaches include dosimetric, toxicological and epidemiological investigations. Such studies are subject to the strengths and limitations discussed in the previous section. However, the ethical considerations which have restricted direct toxicological studies on humans do not exist to the same degree for animal studies. Thus, the uncertainties caused by the extrapolation of data from test animals to the target in question tend to be less, as in many cases testing on the animals can be carried out directly. However, extrapolations between different species and even between different species groups may be necessary. In such circumstances, the problems caused by anatomical and physiological differences and other inter-species variabilities may be even more extreme than for human studies.

Indirect estimations may be conducted using indicators. For example, incremental reductions in the growth of plants can be correlated with ambient atmospheric pollutants such as NO_X and O_3 and this feature could be used as an indirect measure. Also, incremental changes in the species structure of lichen epiphyte communities on the trunks and branches of woody plants in deciduous woodlands may be correlated with ambient SO_2 concentrations and so this could also be used as an indirect measure.

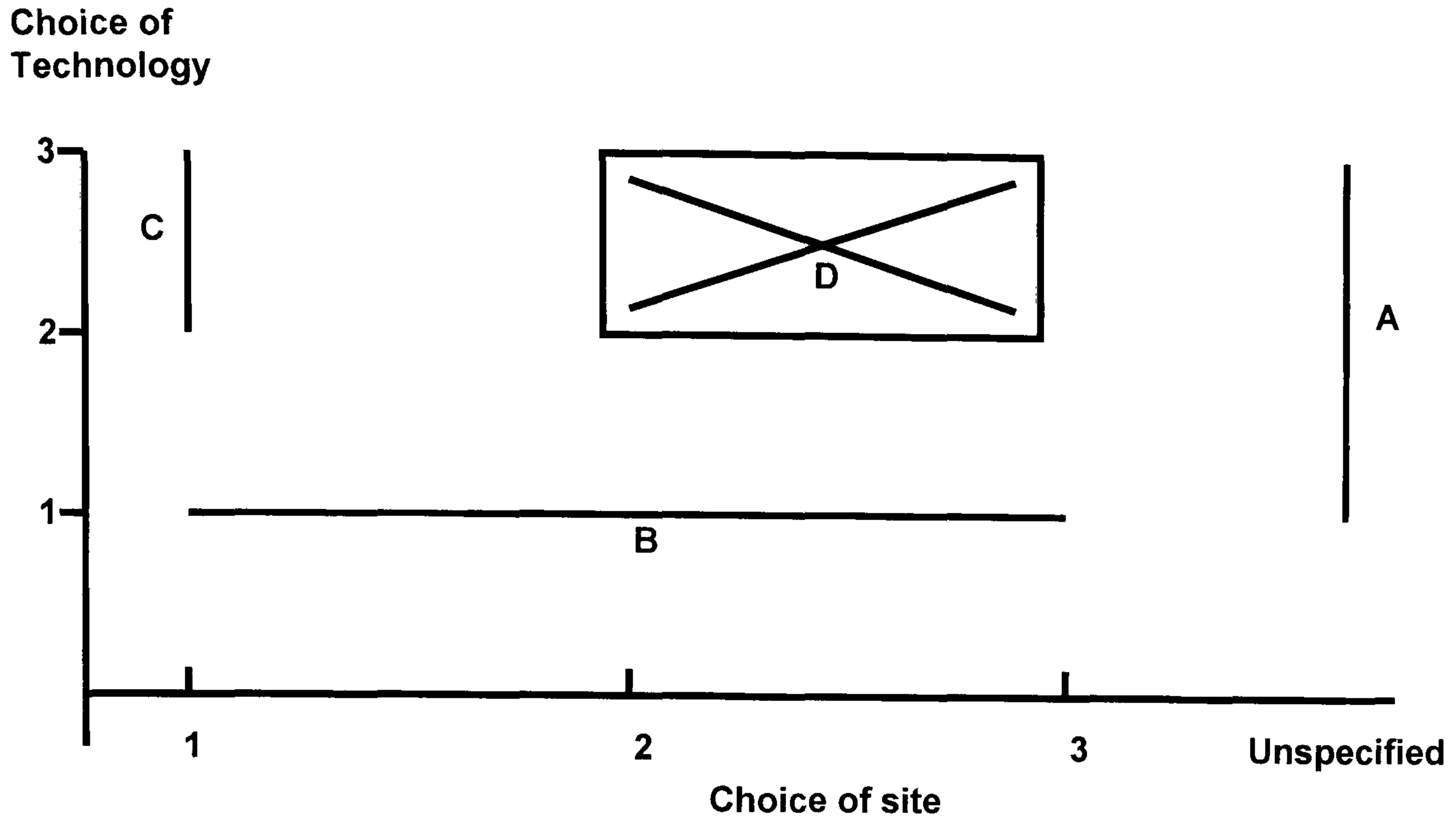
Disadvantages of such methods include the fact that exposures to ambient pollutants are to a complex mixture rather than just to the emission of concern and it is difficult (sometimes impossible) to extrapolate the effects of the emissions of concern from those of the other ambient pollutants. Also, the influence of confounders may be difficult (or impossible) to control under ambient environmental conditions.

The environmental effects of atmospheric emissions have also been estimated using fate and effect modelling approaches. The currently accepted practice is for environmental effects to be classified into a number of effect types such as those illustrated in Table 5.1. Some of these effect types, such as Global Warming and Ozone Depletion are global in nature. Others such as Ecotoxicity tend to be more localised and site-dependant. For the global effect types, effect modelling calculations only are necessary. However, for the non-global types of effects, both fate and effect modelling calculations are necessary. The aim of the fate and effect modelling approaches are to assess the contributions of the emissions in question to each of these effect types.

Substantial uncertainties exist in fate calculations. However, the effect calculations (which are based on dosimetric, toxicological and epidemiological evidence) appear to be even more uncertain. One significant source of this uncertainty is thought to be the fact that the contributions of different pollutants to individual effect types are assumed to be both linear (i.e. proportional to the quantity emitted) and additive (i.e. independent of other pollutants) [Clift et al. (1998)]. This may not necessarily be the case as synergistic or antagonistic interactions might occur.

Both time and site-independent and dependant approaches are used to model and estimate such effects. Life Cycle Assessment (LCA) is a time and site-independent approach which considers the globalised (i.e. generalised impacts) resulting from a product, service or system over the whole of its life cycle (from its cradle as raw materials, through construction and use to its grave as wastes). Thus, it is considered to be an appropriate tool to support strategic decisions on technology selection [Clift (1997)]. On the other hand, time and site-dependant approaches, which may be termed Impact Pathway Analysis (IPA) techniques, involve time and site-specific modelling of the dispersion of emissions and their effect on living organisms [Clift (1997)]. Thus, they are considered to be appropriate techniques for supporting decisions regarding the local impact of specific projects. The stack emission dispersion modelling software ADMS (the Advanced Dispersion Modelling System) and the US EPA ISC (Industrial Source Complex) modelling system are examples of fate modelling approaches which may be used in IPA.

The uses of IPA and generalised approaches (such as LCA) in supporting decision making differ as illustrated in Figure 5.6 below.



A	Established application of Life Cycle assessment (LCA)
B	Established application of Impact pathway assessment (IPA)
C	Requires both LCA and IPA
D	Requires both LCA and IPA

Figure 5.6 Application of site-independent approaches such as LCA and of site-specific approaches such as IPA [Clift (1997), Clift et al. (1998)]]

LCA and other generalised approaches have come to be used in case A type decisions which are concerned with comparing the environmental performance of alternative technological routes for a service such as waste management. In this case, only the technologies themselves are considered, not their location. IPA (case B) has come to be used for considering the impacts from a specific project at different times or at different locations. Thus, case A type decisions are concerned with technology choices and case B decisions are concerned with locational choices. However, in cases C and D both the choice of location and the technology are involved in the decision making. Case C would be a decision context involving technology choice to be used in a defined area or at a known location and case D would be decision context involving selection of both site and the technology (and the scale of the operation). In both, IPA and LCA would be applicable in combination [Clift (1997)].

Figure 5.6 suggests that the assessments in the case studies are expected to contribute to decision making as indicated in Table 5.4.

Decision scale	Decision type	Enclosed as a case study in this thesis?
1. BRM emission source(s)	Case B	No[13]
2. BRM site	Cases C and D	Yes
3. MIM Lead (cradle-to-gate)	Case A	Yes

Table 5.4: Decision types likely to be supported by the scales of the case studies

Thus, Figure 5.6 and Table 5.4 are suggesting the best decision approaches for dealing with emission source scale problems would be site-specific IPA approaches and for cradle-to-gate scale problems site non-specific LCA approaches would be most appropriate. These are indeed the approaches which have been applied. For site scale problems, they are suggesting that combinations of IPA and LCA approaches would be most appropriate. One such approach is the ICI Environmental Burden methodology [ICI (1996)]. It is a hybrid approach which aims, at least in part, to bridge the gap between the IPA and LCA decisions approaches. The potential implications of these assertions for decision making situations, likely to result from the use of the data generated by the case studies, are discussed in the Overall Conclusions and Recommendation (chapter 11).

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[13] Due to time restrictions in the preparation of this thesis, it has not been possible to include a case study operating at this scale. However, preparatory work for such a case study, employing the ADMS tool, has been conducted. The need for such approaches to be included in an integrated strategy, to meet the environmental challenge, is discussed in the Overall Conclusions and Recommendations (chapter 11).

6. Types of Consumptions, Emissions and Effect Categories Associated with the Lead Industry and the CaseStudies

6.1. Lead Industry Consumptions and Emissions

In this thesis, the lead industry is considered to be comprised of those businesses actively involved in the production of refined lead and/or lead alloys from primary (i.e. virgin) or from secondary (i.e. recycled) sources. In the terminology applied in the BRM and MIM case studies (chapters 9 and 10), these are called the 'foreground processes'. As with all industrial production processes, those operated by the lead industry consume raw materials and result, inevitably, in the release of emissions into the environment surrounding its facilities. The chemical and physical nature of these primary (i.e. as released) emissions, are dependant upon the processes from which they are emitted.

Emissions which may be attributed to the lead industry also arise from the 'background processes'. These arise from:

- the upstream ancillary material and energy inputs (from all stages in their production sequences), right back to their respective cradles as raw materials,
- the downstream further processing of wastes (through to their respective final disposal in the environment) and
- the downstream further processing of co-products and by-products, where there are any (until the point where they leave the boundary of the system being considered).

Foreground lead industry processes may themselves be classified into:

- processes particular to the lead industry (e.g. lead smelting, lead refining etc.) and
- processes operated by the lead industry, but which are also conducted in industrial operations associated with other industry sectors (e.g. on-site electric power generation, the transport of materials etc.).

Thus, the proportion of the emissions arising from the later type processes tend to be similar physically and chemically to those from other industry sectors, whilst the proportion arising from the former, tend to be particular to the lead industry.

Chapters 7 and 8 review the potential human health and environmental effects, which may be attributed to the major types of emissions from lead industry processes. These reviews are limited to those potential human health and environmental effects directly associated with the lead industry (i.e. the foreground processes). Background processes are not considered explicitly in these reviews. This is because the aim is to identify and characterise the potential effects particular to:

- the lead industry, in general, and
- BRM and MIM lead life-cycle processes (where sufficient information are available for this to be possible).

However, they are considered, where appropriate, in the case studies. This is because, the industry will also need to account for effects associated with its background processes, if is to meet the environmental challenge in the most effective manner.

The reviews are not intended to be exhaustive, but are restricted to those substances:

- deemed of most concern by the lead and other associated industries (such as the lead-acid battery production industry), the scientific community, regulators and the public in general, and
- consumed from or released into the environment in significant quantities.

Both criteria are normatively based, since concerns regarding resource consumptions and substances emitted into the environment and views on what quantities are significant will vary with time. The approach also implies a certain set of ethical beliefs, views on society, boundaries and decision making rules. It should be noted, that other approaches may be equally valid and should not be discounted simply because they are based on different standpoints. Hence, in adopting this approach, the author is making an 'expert' judgement. Thus, it has similar strengths and suffers from similar shortcoming to those discussed in section 4.4.2.1 of chapter 4.

The most common ore from which lead (Pb) is extracted is galena (lead sulphide, PbS). However, this is often mixed with sphalerite (ZnS). Associated with these ores, are a variety of other trace elements. Thus, from lead smelting and refining operations the following elements may be recovered, but there is probably no single smelter or refinery which recovers them all: lead (Pb), silver (Au), copper (Cu), zinc (Zn), antimony (Sb), bismuth (Bi), nickel (Ni), arsenic (As), sulphur (S), cadmium (Cd), indium (In), tin (Sn), selenium (Se), tellurium (Te), platinum (Pt) and possibly others. [Dodge et al. (1964)] Hence, it is inevitable, a proportion of all of them will be released into the environment. Not all of these substances, however, meet one or more of the criteria for them to be included in the reviews in chapters 7 and 8. Table 6.1 indicates those substances, cited in industry literature for primary and secondary smelting and refined lead production processes, which do (by them either occurring in significant quantities or by a notification to the regulatory authorities being required of their release quantities).

Process	Significant emissions	Notes	Reference source
Britannia Refined Metals (BRM) Limited: Primary and secondary lead refining	*To air:* Pb, Zn, Sn, Sb, Cd, Cu, Te, As, Se *To water:* Pb, Zn, As, Cu, Ni, Ag, Cd, Hg *To land:* Pb, As	BRM is required to report on the annual releases of these substances, as part of its authorisation to operate under Integrated Pollution Control (IPC).	"Application for IPC Authorisation under Section 6 of the UK Environmental Protection Act (1990)" [BRM (1995)]
UK: Primary Pb/Zn smelter	*To air:* Pb, Zn, Cd	There is only one such facility in operation currently in the UK. This is the Britannia Zinc Limited (BZL) facility at Avonmouth. (N.B. BZL operations have been modelled as part of the MIM case study.)	"Pollution Ccontrol for Primary Lead Production" HMIP/RR/93/049 [HMIP (1993)]
Secondary lead recycling	*To air:* Pb *To water:* Pb, Zn, Sb, Cd, Cu, As, Ni, Cr *To land:* Pb, Zn, Sb, As, Sn, Cd	-	"Pollution Ccontrol for Secondary Lead Production" HMIP/RR/93/046 [HMIP (1993a)]

Table 6.1: Significant emissions from primary and secondary lead smelting and refining

The list in Table 6.1 does not include metal and metalloid emissions from mining and concentrations operations. However, since the process materials used in lead smelting and refining operations have passed previously through the mining and concentration stages, the species of the emissions from them are expected to be similar. The following metal and metalloid releases are considered, therefore, to be the most significant for all of the lead industry:

1. Emissions to the air: Pb, Zn, Sn, Sb, Cd, Cu, Te, As, Se
2. Emissions to water: Pb, Zn, As, Cu, Ni, Ag, Cd, Hg, Cr
3. Emissions to land: Pb, Zn, Sb, As Sn, Cd, Ag

Chapters 7 and 8 consider all of these substances. However, whilst all are considered to be significant, published literature has tended to focus on some of the elements only.

The research (and review) efforts of individuals and organisations serve to develop their own paradigms (i.e. conceptual models, as defined in chapter 4). These paradigms have a spectrum similar, generally, to those currently present in society, since they are themselves part of that society. Research and review efforts tend, therefore, to be normative in their focus. This review is similarly normative. Thus, whilst all the elements deemed significant are considered, the extent to which are examined reflects the extent of current concern.

Resource consumptions, plus non-metal and non-metalloid emissions, are also considered, as appropriate.

6.2. The Case studies: MIM Holdings Limited and Britannia Refined Metals Limited

Two case studies, providing assessment approaches for the lead industry, each operating at different scales within the risk management hierarchy (identified in Figure 4.3 of chapter 4), are developed in this thesis. These are:

1. The MIM case study:

This considers and identifies the resource consumption and environmental emissions inventory, as well as the potential environmental impacts, attributable to the production of primary refined lead and refined lead alloys by companies operating under the umbrella of MIM Holdings Limited.

It provides these assessments at various scales, ranging from the cradle-to-gate down (at the highest scale), down to the individual process scale. Hence, the assessments range from the fully site-independent, to the process and hence fully site dependent (as identified in Figure 5.6 of chapter 5).

2. The Britannia Refined Metals Limited (BRM) case study:

This considers and identifies the resource consumption and environmental emissions inventory, as well as the potential environmental impacts, attributable to operations at the following scales; the whole site and the individual streams of throughput through the site.

Hence, these assessments are fully site-dependent (in the sense identified in Figure 5.6 of chapter 5). The BRM company site is in Northfleet, Kent, UK.

6.3 Emissions and Effects

In both the BRM and MIM case studies, raw material and energy consumptions, and emissions are estimated. The potential human and other potential environmental effects are calculated from these emissions. The main types of such effects, associated with lead industry emissions, are reviewed in chapters 7 and 8. However, whether the emissions, identified in the case studies actually give rise to any of these effects and reliability of predictions regarding their magnitude are, of course, dependent upon the considerations outlined in the previous chapter.

7. Potential Human Health Effects from Lead Industry Emissions

7.1 Effects from Atmospheric Emissions

This discussion considers the dosimetric, toxicological and epidemiological evidence for effects upon human beings of ambient exposures to the atmospheric emissions of concern with respect to the lead industry i.e.

- particulates,
- lead, other metals and metalloids,
- NO_x and related compounds,
- SO_x and related compounds,
- CO and
- other significant emissions.

Particulates are examined prior to lead, other metals and metalloids. This is necessary because the latter are emitted and transported in the atmosphere, either as particulates themselves or are adsorbed onto other types of particulates. Hence, an understanding of the potential human health effects of particulates in general is needed as primer, if the potential effects of the latter are to be appreciated fully.

The review has had to concentrate mainly upon primary effects because, for most emissions, this is the highest order in the cause-effect chain at which there is a significant amount of information available and at which the responses to such exposures may be predicted. However, to ensure that the review is as comprehensive as possible, the authors have considered higher order effects related to the emissions in question where sufficient information is available. Thus, separate consideration has been given to tropospheric ozone (O_3) since it is known to be able to produce significant human health effects at current ambient concentration and its creation is dependant upon the modification of the tropospheric photochemistry (a primary effect) resulting from the release of NO_x into the atmosphere. Also, for Pb, plus the other significant metals and metalloids, total exposures are normally a combination from more than one route of entry into the human body. Hence, a separate review of such potential effects is also provided.

The BRM site is in the UK, and the sites associated with the production of lead and lead alloys (considered in the MIM case study), are situated both in the UK and Australia. The Australian MIM operations, directly involved with the production of primary lead, are in:

- Mount Isa and Townsville in northern Queensland, and
- McArthur River and Bing Bong in the Northern Territory.

The activities at Townsville and Bing Bong are quayside warehouse handling operations, which consist of:

- unloading from delivery vehicles (at Townsville these are from rail cars mostly but some are from road containers and at Bing Bong they are from road trains),
- quayside or warehouse storage, and
- subsequent loading onto sea going vessels.

The practices have designed to ensure emissions from the materials being handled are minimal. Hence, any impacts from such activities are expected to be mostly transport and energy consumption related and not to the lead containing substances themselves.

The main processing sites (i.e. McArthur River and Mount Isa) are both remote from large population centres. However, Mount Isa itself is a sizeable town (with a population of 21,750 approximately), whilst at McArthur River the population is restricted to those in the camp (which contains only personnel associated directly with the mining, processing and ancillary processes at the site). UK operations take place at Avonmouth and at Northfleet in Kent. Both occur within major urban areas and are close to substantial population centres (i.e. the cities of Bristol and London respectively).

Data on the actual ambient atmospheric concentrations of emissions in the town of Mount Isa and in the McArthur River camp have not been obtained. However, substantial data concerning UK ambient concentrations are available for all of the emissions of concern. The discussion below has been based, therefore, on such data. Whilst data on ambient concentrations of emissions would be desirable, particularly for Mount Isa, this is not considered to be a major shortcoming, as numbers of people potentially liable to exposure at both locations will be low compared with the UK.

Nevertheless, whilst the focus below is on UK ambient concentrations (and indirectly, therefore, on the emissions from BRM and UK based MIM lead life-cycle processes), the emission categories, and the effects discussion are applicable generally for the whole of the lead industry. It is only the extent of these potential effects which cannot be predicted without ambient concentration data.

This chapter does not attempt an exhaustive survey of the research in the field, even for the UK, since this has been dealt with in documents such as those published:

1. *For non-metal and non-metalloid emissions*:

   by the Quality of Urban Air Review Group (QUARG), the Committee on the Medical Effects of Air Pollutants (COMEAP), the Advisory Group on the Medical Aspects of Air Pollution Episodes (MAAPE), the Expert Panel on Air Quality Standards (EPAQS), plus in miscellaneous review papers.[14]

2. *For metal and metalloid emissions*:

   by the Ministry of Agriculture Fisheries and Food (MAFF), the Expert Panel on Air Quality Standards (EPAQS), plus in miscellaneous review papers and books.[15]

In addition, European and world wide reviews have been provided by organsiations such as the World Health Organisation (WHO), the Organisation for Economic Cooperation and Development (OECD), the United States Environmental Protection Agency (US EPA), and in miscellaneous papers.

Rather, the aims of the chapter are to identify the salient assertions currently being made in the fields and to identify, as much as is possible, the uncertainty underlying them. This information serves as the basis for the case studies (chapters 9 and 10), which provide assessments regarding the relative potential human health and environmental effects of the systems being studied.

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[14] See references in chapter 12, section 12.1, numbered 138, 31 - 33, 90 - 93 and 45 - 48 respectively.

[15] See references in chapter 12, section 12.1, numbered 94 - 98 and 49 respectively.

7.1.1. Particulates

This section considers the following in relation to airborne particulates in general and, where appropriate, to the particulates from the emissions from BRM and MIM lead life-cycle processes in particular:

- the physico-chemical nature of airborne particulates,
- the problems inherent in particulate dosimetry, particulate epidemiology and particulate toxicology and
- the evidence for human health effects from ambient UK concentrations of airborne particulates.

A summary of the main conclusions drawn by the authors is provided at the end of the section.

1. <u>The physico-chemical nature of airborne particulates</u>

Ambient atmospheric particulates are a complex mixture which varies both in the size of the particles as well as in their composition. In addition, there is a diversity in the manner by which they have been characterised [COMEAP (1998)]. QUARG (1993) and COMEAP (1995) have reviewed the diversity of this terminology, as shown in Table 7.1 below, and these terms have also been adopted in this thesis.

Suspended Particulate Matter (SPM):	a general term embracing all airborne particles.
Aerosol:	a suspension of particles in a gas.
Total Suspended Particulates (TSP):	a term describing the gravimetrically determined mass loading of airborne particles, most commonly associated with use of the United States (US) high volume air sampler in which particles are collected on a filter for weighing.
PM_{10}:	particulate matter less than 10µm aerodynamic diameter (or, more strictly, particles which pass through a size selective inlet with a 50% efficiency cut-off at 10µm aerodynamic diameter).
Smoke:	particulate matter, < 15µm, derived from the combustion of fuels.
Black Smoke:	non-reflective (dark) particulate matter, associated with the smoke stain measurement method.
Inhalable Particles:	(also termed inspirable), particles which may be breathed in - "inhalability" is the orientation-averaged aspiration efficiency for the human head.
Respirable Particles:	particles which can penetrate to the unciliated regions of the deep lung.
Thoracic Particle Mass:	describes that fraction of the particles which penetrates beyond the nasopharynx and larynx.

Table 7.1: Terminology used to characterise airborne particulates [COMEAP (1995), QUARG (1993)]

The existence of such overlapping terminology, is due to the fact that much of it is dependent upon the method of measurement which has been employed. The total suspended particulate (TSP) measure is obtained from one of the simplest methods. To measure the TSP, a known volume of air is drawn through a filter. The TSP measure is the mass of the collected particulates. In the smoke stain method, a low flow rate is used to avoid collecting the larger particles and the concentration of the fine particles (i.e. the Black Smoke) is estimated from the blackness of the filter paper. The PM_{10} measurement, on the other hand, is obtained using a size selective sampler. This selects 50% of particles with a 10µm diameter (i.e. cut size[16]), more than 95% of particles with a 5µm diameter and less than 5% of particles with a 20µm diameter. A $PM_{2.5}$ measurement may be obtained by varying the size of the inlet to the collector or more generally by changing the design of the collector. (For the $PM_{2.5}$ measure, the sampler will collect 50% of particles with a diameter

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[16] The cut size is the size range over which greater than 50% of the particulates are collected and less than 50% are not collected.

of 2.5μm.) [EPAQS (1995a)]. The PM_{10} and $PM_{2.5}$ characterisations have become popularly used measures for airborne particulates in recent years. Hence, they are the main measures referred to in this report.

For those wishing to explore the physico-chemical characteristics of ambient atmospheric particulates, a review has been provided in section 1.3.1 of document 24, in volume 5 of the Portfolio . It notes that particulates with a diameter of less than 2.0μm are generally classified as fine particulates and that this category is itself sub-divided into nucleation range particulates (which have a diameter of <0.2μm) and accumulation range particulates (which have a range of 0.2 to 2.0μm) [COMEAP (1995), QUARG (1993)]. It also notes that particulates with a diameter of less than 2.5μm are generally classified as being respirable [COMEAP (1995)]. Data on the size range of particles emitted from lead industry processes have not been obtained by the author. However, low temperature, mechanically generated particles tend to be coarser than particles generated by high-temperature processes such as smelting. Also, metals released from high temperature processes tend to be associated with the sub-micrometer size range [Rasmussen (1996)]. Hence, lead industry dusts, created by mechanical processes such as ore mining and ore crushing, are likely to be coarser than those created by high temperature processes and a significant proportion of the particles will be classified as non-respirable. On the other hand, the bulk of the particles created from high temperature pyrometallurgical processes, such as lead smelting, will be fine, respirable and tend to be in the accumulation size range. These particle will, however, tend to grow in size by the process of aggregation. Prior to discharge via process plant stacks, the air or exhaust containing the particulates is cleaned (usually by filtration in a baghouse). Inevitably, however, a small proportion of the particles will, pass through the abatement system. Such particles would be expected to be mostly in the accumulation range, since the larger size particles would be more efficiently removed.

Thus, it appears the high temperature created particles are being emitted as accumulation range particles. However, the process of aggregation is believed to continue even as they rise up the stack. Whether they subsequently remain in the emitted size range, or grow subsequently in the atmosphere through agglomeration, to form particles with a diameter of greater than 2.5μm (which would not be respirable) is unknown. However, it is thought likely this occurs. Nevertheless, once they are emitted into the atmosphere, the rate of particle growth is expected to be much slower. This is because the process agglomeration is a second order process; i.e. the rate of agglomeration is dependant upon the square of the concentration [Zebel (1966)]. This suggests that though particulates may agglomerate readily in plant ducting and in the immediate 'plume' from stacks (due to the relatively high particle concentrations), once they are fully dispersed, the agglomeration rate is likely to be much slower.

2. <u>The problems inherent in particulate dosimetry, particulate epidemiology and particulate toxicology</u>

Thus far, the general capabilities and limitations of dosimetric, toxicological and epidemiological assertions have been considered. Their specific limitations in relation to ambient particulates will now be analysed.

The COMEAP (1995) report on 'Non-Biological Particles And Health' notes that the degree of exposure to environmental particles (i.e. the received dose) should not just involve a consideration of the airborne concentration of particles or of their constituents but should take into account the appropriate fraction of that aerosol. Estimation of the received dose of an individual will also be dependent upon the characteristics of the particle itself, the local environment within which it exists and the recipient (individual or population) within that local environment.

With regard to the characteristics of the recipient, uptake will be influenced by the anatomical and physiological factors discussed in section 5.5. The characteristics of the particles themselves, will vary according to the size, shape, solubility and the chemical composition of their components and the characteristics of the local environment will vary according to factors such as temperature, humidity, ventilation and lifestyle [COMEAP (1995)].

Confounding variables exist which lead to uncertainty in the risk assessments which may be developed from dosimetry investigations. The following confounders have been identified:

1. Exposures to indoor aerosol concentrations are thought to be increasing in importance in developed countries, due to the fact that modern buildings codes effectively seal the air within rooms. However, outdoor aerosols may still penetrate indoors, with sub-micrometer particle transfers being particularly efficient. Aerosols generated from the combustion associated with the use of gas cookers, wood burning and tobacco smoke may also add to exposure. Due to the amount of time people spend indoors, such exposures may prove to be significant.
2. UK ambient particles, due to their high salt and sulphate components, are generally hygroscopic. This means that hygroscopic particle growth may occur with inhaled particulates. As a result, penetration into the lungs of such particles might be reduced, though data are currently insufficient to prove or disprove this assertion.
3. Sub-micrometer particles may exhibit increased toxicity relative to larger particles. However, size-selective monitoring data of ambient aerosols in the UK is lacking.
4. Ambient particulates may also act as carriers of volatile organic compounds (VOCs) into the lungs, though there is currently insufficient data either to prove or disprove this assertion.
5. Personal exposure monitoring to ambient particulates is limited (and based mostly on US data). Evidence suggests that personal exposure measurements exceed static area sampling measurements by a significant amount [QUARG (1963)]. Also, static area sampling measurements are usually conducted on an intermittent basis. Hence, they are thought unlikely to be able to represent adequately the actual exposures of individuals within a population [EPAQS (1995)].

Knowledge of the concentration of inhaled particles is considered to be insufficient to determine the dose. This is because a large proportion of that which is inhaled may not be deposited in the alveoli but may be exhaled instead. However, due to technical difficulties, few studies have been undertaken on humans to measure the delivered doses of particulates. Instead, deposition has been estimated using animal studies and lung models such as the ICRP 66 Lung Model, the latter of which has been developed by the Task Group for Lung Dynamics of the International Commission for Radiological Protection [ICRP (1991)]. Cautious extrapolation of findings from such studies to humans is essential, however, since inter-species airway geometry varies considerably and lung models are greatly simplified from reality.

When particulates are inhaled, their ability to exert a toxicological response will depend upon three groups of factors. These are the particles themselves (i.e. their size, number, concentration and chemical and physical characteristics), the anatomy of the respiratory tract (i.e. airway structure, cell distribution, age and species differences) and differences in respiratory physiology (i.e. breathing rate, tidal volume and respiratory manoeuvres including breath holding) [COMEAP (1995)]. Thus, unsurprisingly, many of the factors which will influence the distribution of the particulate dose to the tissues are the same as for the dosimetry investigations discussed earlier.

The measure most commonly used for respirable aerosols is the PM_{10}. However, it has been suggested that this an inadequate measure upon which to base either an assessment of dosage or of toxicity [COMEAP (1995)]. To compensate, the $PM_{2.5}$ has also been introduced. Unfortunately, there is evidence to suggest that it too is inadequate. This is because during pollution incidents, the number of sub-micrometer particles often increase significantly [COMEAP (1995)]. Thus, they could provide a significant contribution to the received dose. However, neither the PM_{10} nor the $PM_{2.5}$ measures would be able to detect such changes. This might prove to be a significant omission, since ultra-fine particles could represent a significant hazard as they are believed to be able to produce inflammatory responses in the pulmonary interstitium of the lungs. Clearance times of these particulates are also considerably prolonged compared with larger particulates. Thus, it has been postulated that prolonged, low grade exposure could lead to a chronic inflammatory response [COMEAP (1995)]. There is a need, therefore, for research to identify both the chemical composition and the toxicological effects of ultra-fine particulates on human pulmonary tissue [COMEAP (1995)]. Despite this criticism, there is currently no good understanding of what really is the most biologically relevant index of ambient airborne particulates or of how to measure it [COMEAP (1995)].

The presence of metal ions and other chemical species adsorbed onto the surface of particulates are also thought to be significant influencers of toxicological effect. It has already been noted that atmospheric particulates may act as transporters of VOCs into the lung. However, atmospheric particulates may also transport significant quantities of toxic trace elements such as lead, arsenic, cadmium, beryllium, and chromium as well as gases such as NO_x and SO_x into the lung. The chemical nature of these adsorbates will also depend, to a significant extent, upon the emissions source. Particulates emitted from lead processing operations would be expected, therefore, to have adsorbates of lead and of the other species present in the materials being processed. Hence, it is unsurprising, the lead content of particulate emissions from primary lead refining operations ranges from 20 to 65% by mass. [Data from: US EPA (1998)]

The quantities of adsorbed species per particle will be dependent upon the characteristics of the individual particle such as its internal and external surface area, the hydrophobicity of its surface, its adsorption /desorption characteristics (called the adsorption isotherms) and their kinetics. The presence of other adsorbates, particularly water, may also have a significant influence.

To be able to predict and assess the contribution of adsorbed chemicals to toxicological response, the particles would have to be characterised in much more detail than merely reporting size and gross composition. However, in heterogeneous aerosols, such as those found in rural and urban environments, due to the complexity involved, it has not yet been possible to produce realistic estimates of such contributions [COMEAP (1995)].

Problems also exist due to the fact that different studies may have used different measures of particulates and/or may have assessed different endpoints. With respect to particulate measures, whilst it has been suggested that the PM_{10} measure may be the best choice for a common metric [COMEAP (1995)], other measures, such as TSP and Black Smoke have also been used. Thus, conversion factors have had to be applied to try to equate such measures to the PM_{10} measure. These assume that the dose-response relationship between the original measure and that of the PM_{10} is linear (i.e. proportionate). This may not necessarily be the case, however, since it is believed that relationships may vary according to factors such as the season and the cities being examined [Council of the European Union (1996)]. Therefore, any conversion will, at best, be an approximation and, at worst, may be misleading.

From the foregoing discussion, it is apparent that respirable particulates may transport significant quantities of chemical species which have become adsorbed onto them deep into the lung but that neither the extent of such transport or its variability have yet been determined. It has also been noted that airborne particulates are not only a complex mixture but that that they also exhibit a great variability in their constituents. This variability exists not only geographically but also temporally since substances emitted into the environment undergo complex and dynamic transformations with various secondary particulates being created and removed or destroyed over time. Epidemiological studies are, *de facto*, therefore not only examining the effects of the particles themselves but also those of any adsorbates which may be attached to them. Antagonistic, additive or synergistic effects may occur both between the particles and their adsorbed species and between the species themselves. These constitute additional variables which at the worst may obscure interpretation. At the very least, they will reduce the sensitivity of any epidemiological assertions.

In the case of the emissions from lead industry processes, this means that chemical species which have become adsorbed onto the particulates (such as lead and other substances) will be transported deep into the lungs, since the bulk of the particulates appear to be within the respirable size range. Whether these adsorbates will exert any significant human health effects will depend upon their physico-chemical nature.

3. The human health effects of UK ambient airborne particulates

Extensive reviews of the literature regarding the potential human health effects of ambient UK concentrations of air pollutants have been published. For particulates, the reports of major relevance, used as the primary reference sources for this section of the Report, are:

- Quantification of the Effects of Air Pollution on Health in the United Kingdom' COMEAP (1998),
- 'Urban Air Quality in the United Kingdom' QUARG (1993),
- 'Non-Biological Particles and Health' COMEAP (1995), and
- 'Particles' EPAQS (1995a).

Other references are also cited where appropriate.

It appears that there is clear evidence of associations between the concentrations of ambient airborne particulates (at levels similar to those encountered in the UK) and changes in a number of indicators of both

- morbidity (such as changes in lung function, increased hospital outpatient visits [Xu et al. (1995)], increased cardiac and respiratory admissions [Burnett et al. (1995)], increasing respiratory symptoms [Pope and Dockery (1992), Dockery and Pope (1994)] and increases in respiratory disease [Pope et al. (1995)]), and
- mortality [COMEAP (1995), Dockery and Pope (1994), Katsoyanni et al. (1997), Schwartz and Marcus (1990), Anderson et al. (1996), Dockery et al. (1993)].

Dockery et al. (1993) and Pope et al. (1995a) also identified an association between particulates and cardiopulmonary and lung cancer. Despite this, [COMEAP (1995) contend that there is little evidence to show that particulates contribute significantly to the burden of cancer in the UK. The associations with mortality also appear to be the stronger than with morbidity [COMEAP (1995)]. However, it should be borne in mind that, though these associations are thought to be statistically significant, they are generally weak [Gamble et al. (1996)], the effects are small [Wordley et al. (1997)] and not all studies confirm the existence of associations between particulates and mortality [for example, Styer et al. (1995)]. Nevertheless, COMEAP claim confidence in this assertion, despite the fact that the day-to-day variations in particulates are small compared to other uncontrolled variables such as seasonal variations, weather effects and variations in temperature. It is also believed that health effects are not restricted to specific types of particles, since the effects are apparently detectable with particles emanating from a wide range of sources (such as from motor vehicles, industrial sources and aerosols derived from gaseous emissions such as NOx and SOx). In terms of the nature of the association, it has been suggested that for populations it may be linear and that there may be no threshold for effects [COMEAP (1995), COMEAP (1998), EPAQS (1995a)].

COMEAP (1995) consider the associations between mortality and particulates to be so strong that they are likely to be causal. This finding is corroborated by other reviews of the field such as Katsoyanni et al. (1997) and Stewards (1994). However, COMEAP (1995) did note that they were making this assertion despite the fact that there is no established mechanism of action. They postulated, therefore, that particulates may generate some sort of non-specific reaction whose cause might be ultrafine particulates (i.e. particulates with a diameter of <0.05μm). This assertion is corroborated by the findings of Seaton et al. (1995). The assertion of causality, however, is not universally accepted. For example, Gamble (1996) applied the Bradford Hill (1965) criteria to assess whether such associations were likely to be causal and came to the conclusion that there was insufficient evidence to show that they are.

Despite the fact that epidemiological studies also appear to show that the only factor which correlates with the aforementioned effects is the level of particulates, it is still not known whether the effects of particulates are species specific (i.e. a result of their physico-chemical nature, as described earlier and in section 1.3.2 of Document 24, in volume 5 of the Portfolio) or are a generalised phenomenon [COMEAP (1995)] (possibly due to the effects of ultrafine particulates, as suggested earlier). This is because no determinations of particle size distributions or changes in particle number distributions as pollution indices increase, have yet been reported [COMEAP (1995)]. Despite the lack of knowledge, it is probable that the toxicity of particles will vary according to their chemical composition and surface properties. In terms of effects relating to chemical composition, it has been suggested that toxicity varies more in relation to the acidity of the particles than to their solubility [EPAQS (1995a)]. However, acidity alone is not thought be the primary component defining particulate air pollution toxicity [Brauer et al. (1995)].

Experimental challenge studies have examined the toxic effects of substance typically found in ambient airborne particulates. Such studies have indicated that some particulate species may exhibit greater effects than others. This observation has been derived from challenge studies where individual substances were tested upon volunteers. Where combination challenges have been conducted, however, (which would provide a better simulation of ambient exposure conditions) the results have been inconsistent [COMEAP (1995)]. Nevertheless, challenge investigations using, in isolation, sulphuric acid, ammonium sulphate, ammonium bisulphate and ammonium nitrate at doses of between 100 to 1000μg m^3 have appeared to show no consistent effects between studies on the lung function of normal subjects. However, at high doses (i.e. 1000μg/m^3), ammonium bisulphate has been shown to increase bronchial responsiveness to the chemical methacholine. At such doses, sulphuric acid also has been shown to cause changes in lung function (in the form of reduced forced expiratory volume and mid-expiratory flow). However, this effect has not been demonstrated consistently with all studies [COMEAP (1995)]. In asthmatics, responses have been demonstrated with ammonium bisulphate but not with other sulphates [COMEAP (1995)]. This appears to corroborate the findings of

Amdur et al. (1978), who demonstrated the metal ion composition to be critical to the biological effects of sulphate salts. They suggested that the irritant potencies of sulphates could be ranked as shown in Table 7.2 below.

Compound	**Relative potency**
Sulphuric acid	100
Zinc ammonium sulphate	33
Ferric sulphate	26
Zinc sulphate	19
Ammonium sulphate	10
Ammonium bisulphate	3
Cupric sulphate	2
Sodium sulphate	0.7

Table 7.2 The relative potency of sulphates [Amdur et al. (1978)]

In terms of who might be affected by ambient airborne particulates in the UK, COMEAP (1995) have concluded that there is no evidence that healthy people will experience any effects from current UK ambient airborne particulate concentrations. However, it has been argued that people with predisposing factors, such as neonates [Woodruff et al. (1997)] and those with a history of chronic heart disease, respiratory problems or asthma are at risk of experiencing deleterious effects from UK ambient airborne particulate concentrations [COMEAP (1995)]. It also appears that such effects have been most strongly correlated with acute studies. The evidence for long-term (i.e. chronic) effects is less well developed and the problems of dealing with confounding variables appears to be even more severe. However, COMEAP advises that it would be prudent to consider the apparent association which has been observed between mortality and morbidity indicators and long-term exposure to be causal [COMEAP (1995)].

4. Summary

Section 7.1.1 may be summarised as follows:

1. Ambient atmospheric particulates are a complex mixture which varies considerably in both the size composition and range of the particles themselves and their physico-chemical composition.
2. The sensitivity of particulate dosimetry, toxicology and epidemiology for estimating the human health effects of such ambient particulates is low due not only to this complex physico-chemical nature, but also due the fact that the anatomical and physiological characteristics of the recipients to the particulates varies and that confounding variables exist which cannot always be fully controlled.
3. Chemical species adsorbed on respirable particulates (i.e. with a diameter of 2.5μm) will be transported deep into the lung where they may exert their own human health effects, in addition to any which may be exerted by the particulates themselves.
4. The particulates emitted by high temperature pyrometallurgical lead industry processes are thought to be largely respirable. It is inevitable, a proportion of these particles will be also emitted to the atmosphere. A significant proportion of these particles are either composed of lead and other associated substances or have these substances adsorbed onto them. Such substances have the potential to exert their own human health effects, in addition to those attributable to particulates in general. However, from the data obtained by the author, it has not been possible to ascertain whether or not they give rise to any. Nevertheless, even if they do, their effects in Australia and the UK would probably be minor, as the industry is regulated and employs abatement technologies and management techniques aimed at ensuring this is the case.
5. The particulates emitted from mechanical activities, such as ore mining and ore crushing, are likely to be coarser.
6. With regard to the acute human health effects of particulates themselves, there appears to be clear evidence of associations between ambient UK airborne particulate concentrations and various indicators of morbidity [Xu et al. (1995), Burnett et al. (1995), Pope and Dockery (1992), Dockery and Pope (1994), Pope et al. (1995)] and mortality [COMEAP (1995), Dockery and Pope (1994), Katsoyanni et al. (1997), Schwartz and Marcus (1990), Anderson et al. (1996), Dockery et al. (1993)]. It has also

been suggested that this dose-response relation may be linear and that there may be no threshold for the effects [COMEAP (1995), COMEAP (1998), EPAQS (1995)].

7. With regard to chronic effects human health effects, the evidence is less well developed. However, apparent associations between mortality and morbidity indicators have been observed.
8. Some species of particulates have been shown to exhibit a greater irritancy than others and sodium sulphate is considered to have a very low, possibly negligible, irritant potency. However, particulates are believed to cause human health effects irrespective of species type. It has been suggested that this generalised response may be caused by ultra-fine particulates.
9. In terms of whose might be affected by current ambient UK concentrations of particulates, healthy people are considered unlikely to be affected. However, neonates [Woodruff et al. (1997)], people with predisposing factors such as asthmatics and people with chronic heart disease may experience some acute effects [COMEAP (1995].

7.1.2 Lead, Other Metals and Metalloids

This section considers the following in relation to airborne lead, other metals and metalloids in general and, where appropriate, to those emitted from BRM and MIM lead life-cycle processes in particular:

- the evidence for the contribution of anthropogenic atmospheric emissions to environmental concentrations and fluxes, and
- the evidence for human health effects from UK and Australian ambient atmospheric concentrations of these elements.

The emissions of concern from the lead industry were identified, in section 6.1 of chapter 6, as being: Pb, Zn. Sn, Sb, Cd, Cu, Te, As and Se. However, some are of more concern currently than the others. This is reflected in the emphasis given to each element in the review.

Since lead, other metals and metalloids are emitted and transported in the atmosphere, either as particulates themselves or are adsorbed onto other types of particulates, readers requiring a complete overview are advised to study the previous section first (since it reviews particulates).

1. Evidence for the contribution of anthropogenic atmospheric emissions of lead, other metals and metalloids to environmental concentrations and fluxes:

Substantial and significant uncertainties exist in attempts to determine the contributions of anthropogenic atmospheric emissions to global metal fluxes. Thus, for example, Jaworowski et al. (1981) measured the concentrations of ^{137}Cs, ^{210}Pb, ^{226}Ra, U, V, Pb, Cd and Hg in firn and glacial ice, deposited during the three decades preceding the 1980s and during pre-industrial times, in various locations around the globe. They were unable to find a significant difference in the rate of metals deposition between the two periods, but concluded their samples could have been contaminated by contemporary fall out migrating to deeper (pre-industrial) parts of the glaciers. They also concluded that natural processes dominate flows and the anthropogenic contribution is only small fraction of the total. Thus, for example, they estimated the anthropogenic contribution of Pb to be 7.8%, Cd to be 0.4% and Hg to be 0.8% of the total annual flows. They also noted that natural atmospheric Pb concentrations may vary, spatially, by several orders of magnitude, due to volcanic eruptions and other geological anomalies. This latter factor, highlights the crucial importance of (and the problems in) obtaining representative samples. Thus, Rasmussen (1996) noted, that Nriagu's (1989) estimate of the global flux of Pb from volcanoes ranges by over 4 orders of magnitude, due to the difficulty in obtaining temporally representative samples, which is in turn due to the episodic nature of volcanic eruptions.

The problem of obtaining reprentative data has led to considerable debate in the literature. [Rasmussen (1996)] [See: Pattersen (1983) and Jaworowski et al. (1983).] Thus, Pattersen (1983) contended the Pb and other concentrations indicated by Jaworowski et al. (1981) were too high and were the result of sample contamination, combined with a low sensitivity in the measurement technique. However, whilst it is true the "avoidance of sample contamination is of extreme importance, ... it is also true that orders-of-magnitude variations in atmospheric Pb concentrations can arise for reasons other than erroneous data". [Rasmussen (1996) p. 6] Hence, even with excellent data quality, a global estimate will only be correct, providing the data used to derive it also are spatially and temporally representative. As a result of such problems, there are significant uncertainties when comparing natural and anthropogenic emissions, at the local, regional and global scales.

Despite such difficulties, Pacyna et al. (1989) have modelled the long-range transport of As, Sb, V and Zn, at the European scale, and have concluded that it is possible to correlate emission measures for Europe with concentration measurements at remote locations in Norway. Hence, for Europe at least, they have contended that source apportionment for anthropogenic emissions is possible. Pacyna (1983) has also made estimates of the emissions of 15 elements from various anthropogenic sources in Europe. (The elements include all of the elements deemed to be of concern to the lead industry in this thesis, except for Sn and Te.) For As, Cu, Cr, Mn and Cd emissions, Pacyna also estimated their European spatial distributions.

At the global scale, Nriagu (1990) and Nriagu and Pacyna (1988) have estimated the atmospheric emissions of trace metals from natural and anthropogenic sources, using emission factors for various industrial source categories. Whilst at the UK scale, Hutton and Symon (1986) have estimated the quantities of Cd, Pb, Hg and As entering the UK environment. They noted that the major atmospheric sources (at the time of publication) were:

- Cd and Hg: refuse incineration,
- Hg: the chlor-alkali industry,
- Pb: petrol combustion, and
- As: coal combustion.

Since the time of publication, however, the addition of Pb to petrol has been largely phased out and the contribution of coal combustion to grid electricity power generation (which at the time was the major source of coal combustion in the UK) has also declined significantly. Hence, the relative importance of these two sources will also have changed significantly.

In chapter 2 it was noted that task groups have been formed by a number of international agencies to address the question of whether trans-boundary atmospheric movement of metals from industrial sources has actually occurred. It was also reported, that the conclusion of the UN-ECE (1995) is there is now sufficient evidence it has. This view is in full consort with much of the aforementioned published work in the field. However, Rasmussen (1995), who has conducted a detailed literature view to evaluate the evidence in support of such long range transport, has concluded that further research is still required. The reason cited being, that in view of the uncertainties due to the significance and variability of naturally occurring metals in remote areas, a systematic overestimation of the anthropogenic component of the global metal cycle may occur. The following sources of significant uncertainties, also cited by Rasmussen (1996), lend support to this assertion. These are that:

- natural emission estimates are extrapolated from a very sparse data set,
- source apportionment models are limited in their ability to distinguish between airborne metals of anthropogenic rather than natural origin, and
- there are critical information gaps associated with attempts to quantify the natural and anthropogenic components of metals in soils, vegetation and lake sediments.

Hence, a difference of opinion exists within the field, which has yet to be resolved, as to whether there is, as yet sufficient evidence, to confirm trans-boundary atmospheric metal transport. On the balance of the evidence, it appears probable that it has (at least for some of the elements of concern). However, further research is required.

2. Evidence for human health effects from UK and Australian ambient atmospheric concentrations of lead, other metals and metalloids of concern:

Unlike the other major categories of atmospheric emissions, where human health effects may be apportioned to atmospheric exposures alone, exposures to lead, plus other metals and metalloids are normally a combination of exposures from more than one route of entry into the human body. The principal routes of entry are through the lungs (by breathing Pb contaminated air) and through the stomach and intestines (by drinking and eating Pb contaminated materials). Exposures are generally a combination from all routes, because for several of the substances (e.g. Pb and Cd), the effects are cumulative. Hence, exposures from all of the major routes of entry to the human body have the potential to contribute. It is inappropriate, therefore, to try to identify the contribution of atmospheric emissions to human health effects in total isolation of the other exposure routes. Hence, the combined effects of exposures vai all routes are considered explicitly in section 7.4.

The purpose of this section, is simply to review the known and postulated human health effects of Pb, plus the other metals and metalloids of concern and to identify how ambient atmospheric concentrations in the UK and Australia might contribute to such effects. However, reflecting the fact that some are of more concern currently than others, the emphasis in the review has been tiered as indicated in Table 7.3.

Emission	**Features**	**Assumed level of relative concern**	**Emphasis adopted in review**
Pb	Greatest attention and greatest mass emissions	Highest	Highest
Cd, As, Cu, Zn	Less attention and less mass emissions	Intermediate	Intermediate
Sn, Sb, Te, Se	Least attention and/or least mass emissions	Lowest	Lowest

Table 7.3: Assumed levels of relative concern and emphasis adopted for each type of emission to the atmosphere by the lead industry

By far the most attention and the greatest mass emissions are from Pb. Hence, it has its own separate section in the review. The other emissions are reviewed together, with the emissions of intermediate significance being examined in more detail and prior to those of lowest significance.

a) Lead:

Extensive reviews of the literature regarding the human health effects and ambient atmospheric concentrations of Pb in the UK and other countries have been published. Major reports of relevance are:

- 'Lead' EPAQS (1998),
- 'Air Quality Guidelines for Europe' WHO (1987),
- 'The Relationship Between Air Lead and Blood Lead in Children: A Crritical Review' Brunekeef (1986),
- 'Air Quality Criteria for Lead' US EPA (1986),
- 'Co-operation on Existing Chemicals: Risk Reduction Lead Country Report on Lead' OECD (1991), and
- 'Lead and Health' Cole et al. (1993).

Therefore, they have been used as the primary reference sources. However, other references are also cited, where appropriate.

Humans are exposed to Pb directly through the inhalation of Pb emitted to air. However, air also transports Pb to other media, including dust, soil, food and water. Deposited Pb may then also become re-entrained into the air with wind blown dust [OECD (1991)]. It is estimated that, globally, approximately 332,350 tonnes of Pb per year are emitted into the atmosphere from mobile and stationary anthropogenic sources [Nriagu and Pacyna (1988)]. Of this total, releases to the atmosphere from industries associated with the lead industry and from the lead itself, are estimated to contribute as follows:

- the mining of non-ferrous metals ores for 1,700 to 3,400 tonnes,
- Pb production for 11,700 to 31,200 tonnes,
- Cu-Ni production for 11,050 to 22,100 tonnes,
- Zn-Cd production for 5,520 to 11,500 tonnes, and
- mobile sources for 248,030 tonnes [Nriagu and Pacyna (1988)].

Therefore, these processes account for around 89%[17] of the total global annual anthropogenic emissions to the atmosphere of Pb.

The mobile sources are releases as a result of the use of leaded fuels in road vehicles, and the stationary sources as a result of refining, manufacturing and incineration operations [OECD (1991)]. In recent years, however, there have been actions taken, globally, to reduce and eliminate the use of Pb additives in fuels. Therefore, the contribution of mobile emissions to the global total is likely to have been reduced significantly. Stationary source emissions are both from point sources, via chimney stacks, and from fugitive releases.

The main sources of contamination by Pb of dust (both inside and outside) are from vehicle emissions and from the flaking of lead based paint. Even though the use of Pb in interior paints has been banned or restricted in many OECD countries [OECD (1991)], the huge reservoir of contaminated material in existing housing, constitutes a significant potential hazard, especially for children. This is because, they tend to indulge in pica activities i.e. they may chew of pieces of flaking paint, or they may ingest house dust or garden soil, which has been contaminated by flaking lead paint.

Tables 7.4 and 7.5, on the following pages, summarise the lowest observed effects levels for key lead-induced effects in adults (plus foetuses) and children respectively. These tables provide reference points for the discussion which follows, and the discussion explains the meaning of the terminology used in the tables.

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[17] Calculated using median values, estimated by the author from the above data of Nriagu and Pacyna (1988), for non-ferrous metals ore mining, and for Pb, Cu-Ni, and Zn-Cd production. These estimated median values are: 2,550, 21,450, 16,575, and 8,510 tonnes respectively.

Lowest Observed Effects Level (PbB)	Haem Synthesis and Haematological Effects	Neurological Effects	Renal System Effects	Cardiovascular Effects	Reproductive Effects: Women or Foetus	Reproductive Effects: Men
80-100 µg/dl		Encephalopathic signs and symptoms	Chronic nephropathy (aminoaciduria, etc.)	Heart Injury		Infertility
						?
70 µg/dl	Frank anaemia					Testicular dysfunction
60 µg/dl		Peripheral neuropathies			Pregnancy complications	
50 µg/dl		↓ ?			↓	Increased Abnormal and Lowered Sperm Count
40 µg/dl	Reduced haemoglobin synthesis	Peripheral nerve dysfunction (slowed NCV's)				
	Elevated coproporphyin	CNS cognitive effects (IQ deficits etc.)			Premature birth	Chromatid and Chromosome Aberration
30 µg/dl		↓ ?	Vitamin D metabolism interference	Elevated blood pressure		?
15 µg/dl	Erthyrocyte protoporphyrin elevation	Altered CNS electrophysiological responses			↓	
10 µg/dl	ALAD-D inhibition		↓ ?		Decreased growth	
	PY-5-N activity inhibition	↓ ?		↓ ?	↓ ?	
	↓ ?					

Key to abbreviations:

PbB = blood lead concentration

ALAD-D = delta-aminolaevulinic acid dehydrase-enzyme

PY-5-N = pyrimidine-5'-nucleotidase enzyme

NCV = nerve conduction velocity

CNS = central nervous system

[Sources: OECD (1991); Modified from US EPA (1986) and Winder (1989)]

Table 7.4: Summary of lowest observed effect levels for key lead-induced effects in adults

Lowest Observed Effects Level (PbB)	**Haem Synthesis and Haematological Effects**	**Neurological Effects**	**Renal System Effects**	**Gastrointestinal Effects**
80-100 µg/dl		Encephalopathic signs and symptoms	Chronic nephropathy (aminoaciduria etc.)	Colic, other overt gastrointestinal symptoms
70 µg/dl	Frank anaemia			
60 µg/dl		Peripheral neuropathies		
50 µg/dl		?		
40 µg/dl	Reduced haemoglobin sysnthesis	Peripheral nerve dysfunction (slowed NCv's)		
	Elevated coproporphyrin	CNS cognitive effects (IQ deficits, etc.)		
	Increased urinary ALA			
30 µg/dl		?	Vitamin D metabolism interference	
15 µg/dl	Erythrocyte protoporhyrin elevation	Altered CNS electrophysiological responses		
10 µg/dl	ALAD-D inhibition		?	
	PY-5-N activity inhibition	?		
	?			

Key to abbreviations:

PbB = blood lead concentration
ALA-D = delta-aminolaevulinic acid- deydrase enzyme
PY-5-N = pyrimidine-5'-nucleotidase enzyme
NCV = nerve conduction velocity
CNS = central nervous system

[Sources: [US EPA (1986), OECD (1991)]

Table 7.5: Summary of the lowest observed effect levels for key-induced health effects in children

The human health effects of Pb may, to some extent, be explained by the fact that Pb inactivates enzymes, by binding with the SH-groups of their proteins or by displacing other essential metal ions. Hence, practically all organs, and organ systems, may be viewed as potential targets for disruption by Pb [WHO (1987)]. Indeed, a wide range of effects have been documented. As indicated in Tables 7.3 and 7.4, these may be divided into:

- effects on haem synthesis and haematological effects,
- neurological effects,
- renal system effects,
- cardiovascular effects,
- reproductive effects, and
- gastrointestinal effects.

The effects of Pb may also be classified according to the 3 major groups of subjects who may be exposed. These are effects on:

- adults,
- children, and
- the foetus.

Pb which has been absorbed into the body, is spread around and accumulates in various tissues but particularly in bone, teeth, skin and muscle. Only about 2% remains in the blood. However, it is this fraction which is biologically active and which gives rise to the harmful effects [EPAQS (1998)].

Whilst acute effects may be observed occasionally, in individuals who have been exposed accidentally to high levels of Pb contamination, exposures in the general population are normally long-term and low-level (i.e. chronic). For the latter, the most critical effects are generally haem biosynthesis disruption, erythropoiesis, nervous system effects and increased blood pressure [WHO (1987)]. Workers in the lead industry, and in other industries handling lead-based materials (in forms able to enter the human body), may receive higher overall exposures, due to these additional occupational Pb sources. They may, therefore, experience some additional effects. The possible effects experienced both by the general population and those with additional occupational exposures are discussed below. The former are discussed first.

The normal process of haem biosynthesis and its interference by Pb are well understood [WHO (1987)]. It is known, that the initial stages occur in the mitochondria of erythrocytes and the final in their cytoplasm. Pb is known to interfere with the process in three main ways, namely:

1. it stimulates, indirectly, the mitochondrial enzyme delta-aminolaevulinic acid synthetase (ALAS-D)
2. it inhibits, directly, the cytoplasmic enzyme delta-aminolaevulinic acid dehydrase (ALAD-D)
3. it interferes with the normal functioning of intramitochondrial ferrochelatase, which inserts Fe(II) into the protoporphyrin ring of haem [WHO (1987)].

Stimulation of ALAS-D has been found in lead workers at blood Pb levels of 40 μg/dl [Meredith et al. (1978)]. Inhibition of ALAD-D, which catalyses the conversion of aminolaevulinic acid (ALA) to porphobilinogen [US EPA (1986)], leads to the accumulation of ALA in blood, plasma and urine. The threshold of urinary ALA elevation is widely accepted as being at blood Pb levels of 40 μg/dl [US EPA (1986), WHO (1987)], though some studies have demonstrated inhibition of ALAD-D activity, in human and other mammalian species, at blood Pb levels below this even, and with no clear threshold being evident [US EPA (1986)]. Interference with the insertion of Fe(II) into the protoporphyrin rings of haem, is evidenced by increased levels of erythrocyte protoporphyrin (EP) and zinc protoporphyrin (ZPP) in the blood and urine. ZPP is formed due to the fact that the porphyrin acquires Zn in lieu of Fe(II), which is blocked by the Pb. The ZPP, which remains tightly bound to the erythrocytes for their entire life (about 120 days), is used as a common index of Pb exposure in medical screening [US EPA (1986)]. Studies of pooled data, indicate a threshold, for an increase of EP, of about 20 μg/dl for adults and 10 to 20 μg/dl for children [WHO (1987)].

One of the more direct signs of damage to the haemopoietic system, is an increase in erythrocyte (red blood cell) destruction. In severe cases, anaemia may result. (Normally, such cases would only occur in cases of acute exposure.) The reduction in haem in the body gives rise, in turn, to a variety of impacts on numerous systems, at both the cellular and organ levels, including the renal-endocrine system and the liver. As a result, vitamin D activation may be disrupted, thereby causing complications in calcium homeostasis [OECD (1991)].

Encephalopathy has been observed in adults with blood levels exceeding 120 µg/dl and children with levels of 80 to 100 µg/dl. It is frequently fatal in children and those who survive often demonstrate irreversible neurological and neuropsychological sequelae. However, some controversy exists as to whether blood Pb levels less than 70 µg/dl are detrimental to the structure and function of the peripheral nervous system and/or the central nervous system (CNS) [WHO (1987)]. This is because, damage to the peripheral nervous system has been examined using nerve conduction velocity (NCV) as an indicator. Although such studies have shown a reversible effect in Pb exposed workers, at blood Pb levels between 30 to 80 µg/dl, the effect is small and reversible, and the clinical significance of the findings is considered to be uncertain [WHO (1987)]. With regard to CNS effects, it has been shown that children are at greatest risk because CNS development is at its peak in the first few years of life [Robertson (1994)]. One measure of CNS effects, which has been studied extensively, is the effect of blood Pb and Intelligence Quotient (IQ) in children. Whilst few studies appear to have taken account of the influence of confounders, EPAQS (1998) argue that, taken together, the data suggest an inverse relationship between blood Pb and intelligence in populations (i.e. the higher the blood Pb the lower the average IQ in the population). However, some controversy surrounds this finding, due to the possible influence of the confounders. Indeed, Pocock et al. (1994) who have conducted a systematic review of the subject, suggest other explanations need considering. For example, some studies have indicated a possible relationship between poor diet and lowered IQ [Cole et al. (1993)]. Behavioural habits and environment might also result in lower scores in IQ tests. It could be the case, therefore, that children of lower average intelligence might be more likely to suffer exposure to Pb, because of their behavioural habits, environment and/or diet, rather than Pb itself causing the lowering of the average intelligence. Hence, it may be the case, the relationship between blood Pb and IQ is actually not causative [EPAQS (1998)]. The statistical association is also small i.e.

only between 1 and 3 IQ points, for a 10 µg/dl increase in blood level [Cole et al. (1993)]. It also appears from various critical reviews of the literature, that there is no convincing evidence of cognitive deficits in children with blood Pb levels below 40 µg/dl, though at higher levels concern is warrented [WHO (1987)].

It has been shown from epidemiological survey data, from the UK and USA, and from animal study data, that a statistically significant positive dose-response relationship between diastolic and systolic blood pressure and blood Pb levels exists. However, the relationship in the UK was much weaker than in the US study. Whilst survey data cannot prove definite causality, the agreement of the animal study data with the survey data, suggests that it is. The relationship also appears to be without an obvious threshold [WHO (1987)]. This is significant for the general population as it suggests the blood pressure effects from Pb may be widespread.

Occupational exposures in the lead industry and other industries handling lead containing materials in forms able to enter the human body, can raise the blood Pb levels in some workers to the extent that additional effects could be experienced. One area, which has been the subject of investigation, is the influence of occupational exposure on male fertility. However, Cole et al. (1993) note that current data on the Pb effects from workplace exposures appear to be inconsistent, though testicular and seminal effects (such as decreased sperm motility and count and increased numbers of abnormal sperm) appear to be associated with blood Pb levels above 40 µg/dl.

With regard to effects on the female reproductive system, the effects at blood Pb levels lower than 30 µg/dl are inconsistent. Both human and animal study data indicate, however, that high blood Pb levels (i.e. clinical poisoning following high level exposure) has been associated with decreased fertility, still births and spontaneous abortions. Nevertheless, no association between blood Pb levels in the general population and spontaneous abortions have been found, although premature births have been reported [Cole et al. (1993)].

Acute Pb poisoning in both human and animal studies, produces structural and functional changes in the proximal tubular lining cells in the kidney and these changes may progress to nephropathy with tubular atrophy. However, evidence for significant pathological and functional renal system impairment has not been demonstrated convincingly in the general population. Indeed, such impairment does not appears to be associated with blood Pb levels below about 62 μg/dl. This does not preclude the possibility of such effects in the general population, only that further clarification is required [Cole et al. (1993)].

Gastrointestinal effects may also be experienced, following high level exposure in adults, where the blood Pb level is raised to in excess of 100 to 120 μg/dl. However, in children, colic and other overt gastrointestinal symptoms present down to a blood Pb level of at least 60 μg/dl.

Typically, 90 to 95% of the Pb in a person or animal is stored in the skeleton, where it stays for many years. However, the ageing process (which gives rise to osteoporosis) and some situations of body stress (such as pregnancy), cause demineralisation of bone. This might cause damaging effects in the elderly and to the developing foetus, where the body burden of Pb is high.

Possible carcinogenicity has also been investigated. Certain Pb compounds, notably lead acetate, subacetate and phosphate, may give rise to benign or malignant kidney tumours, when administered to rats and mice in high doses. However, epidemiological evidence that Pb is carcinogenic in humans is inconsistent.

With respect to average ambient atmospheric Pb concentrations, the WHO (1987) estimated them to be below 0.5 $\mu g/m^3$ in European non-urban areas and in European kerbside urban areas to be between 0.5 and 3 $\mu g/m^3$. In the UK, during approximately the same period (i.e. the mid-1980s), the annual average levels at a busy kerbside monitoring site in west London were around 1.4 $\mu g/m^3$ and at urban sites, in general, were in the range of 0.15 to 0.8 $\mu g/m^3$. Hence, the UK urban measurements were in the middle of the range cited by WHO (1987) for Europe. By 1996, however, they had fallen to between 0.009 and 0.038 $\mu g/m^3$in various rural locations, to between 0.118 and 0.171 $\mu g/m^3$ in kerbside urban areas, and to between 0.030 and 0.148 $\mu g/m^3$ in urban areas in general [EPAQS (1998)]. Such reductions are a result of the lowering in the maximum permissible Pb content in petrol from 0.4 to 0.15 g/litre and the introduction of vehicles using unleaded petrol. From the start of 2000, the sale of unleaded petrol is due to be banned, except in a limited number of specific cases. As a result, urban and background rural ambient atmospheric concentrations will probably drop still further.

With regard to Australian ambient atmospheric concentrations, in many rural areas they are also likely to be well below the 0.5 μg/m^3 indicated as a typical European concentration in WHO (1987). Also, since the continent of Australia is relatively sparsely populated and the majority of its population live close to its coasts, in places distant from industrial activities and urban populations, concentrations are likely to be close to the natural background. Hence, in many areas, concentrations will certainly be much lower than the WHO European urban estimate. In areas close to where the mining and processing of Pb containing materials takes place, however, the Pb concentration may to be significantly higher than the WHO estimate.[18]

With regard to UK ambient atmospheric concentrations around lead and associated industry premises. These have to comply with the current national air quality standard (NAQS) for non-ferrous metal works, which is 2.0 μg/m^3 as an annual average not to be exceed at housing in the vicinity. Estimates of the numbers of people who might be affected by this NAQS are unknown to the author. There are proposals to lower the NAQS to 0.5 μg/m^3. Its benefits are also unknown to the author.[19]

In order to assess the extent of the potential human health effects attributable to atmospheric emissions from the Pb industry in general, and BRM and MIM Pb life-cycle processes in particular, it is necessary to identify how incremental changes in the one affect the other. The health effects associated with differing blood levels have been characterised (as explained above). Therefore, the health effects attributable to these emissions could be estimated in theory, if the relationship between changes in ambient atmospheric Pb exposures and human blood Pb levels is known.

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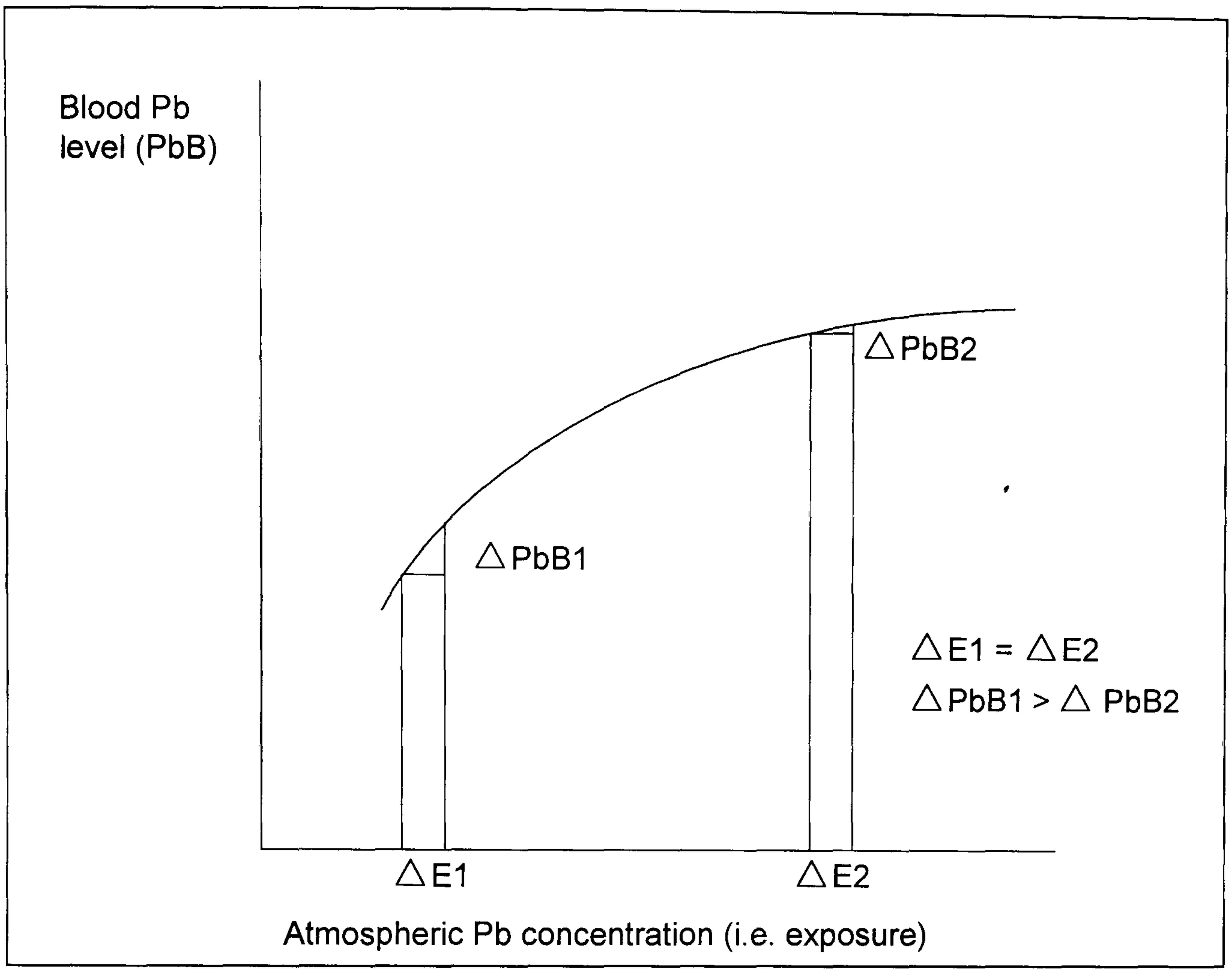
[18] Data on typical ambient Australian atmospheric Pb concentrations in rural and urbanised areas in general and around Australian MIM Pb life-cycle processes in particular, plus the numbers of people exposed to these concentrations, have not been identified. These are areas for further research.

[19] Both are areas for further research.

Studies examining this issue, have identified a relationship which at high exposures is downward curvilinear but at lower levels of exposure is close to linear [WHO (1987)]. This is shown diagrammatically in Figure 7.1.

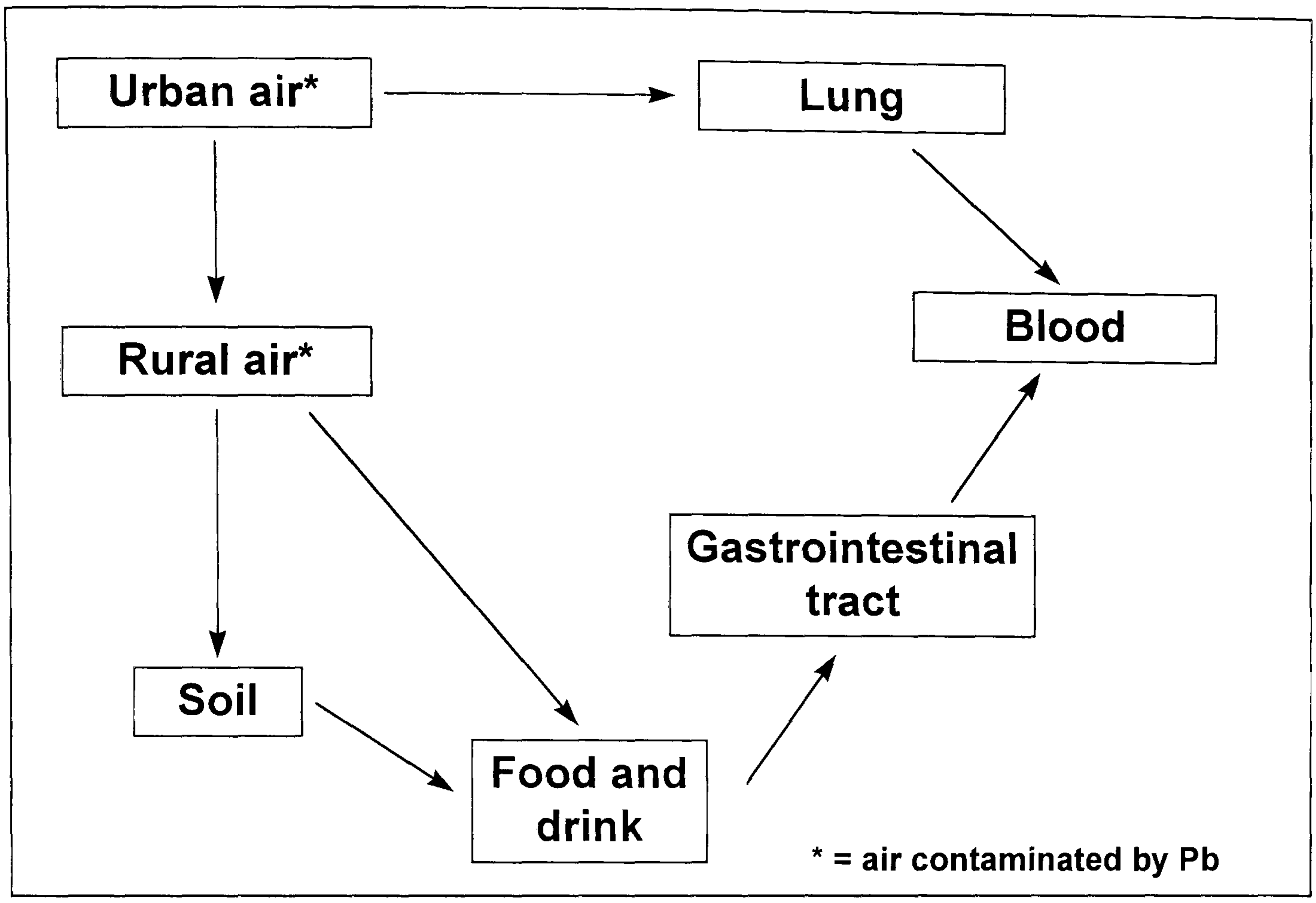


[Source: Modified from Brunekreef (1984)]

Figure 7.1: Curvilinear dependence of blood Pb on atmospheric Pb exposure

Experimental inhalation studies in adults, at atmospheric Pb levels of 3.2 µg/m^3, have given typical slopes of 0 to 3.6, with a typical value of 1.64 ± 0.22 [US EPA (1986)]. The US EPA (1986) have estimated the slope for children to be at least as steep and provide an estimate of 1.92. However, Brunekreef (1984), in a critical review of the subject, noted that most values centre on 3 to 5 rather than 1 to 2. Brunekreef also noted, that adjustment for confounders has been absent or incomplete, in most if not all studies, so the estimates should be "viewed with caution".

A significant proportion of atmospheric Pb does not enter the body directly via the lungs, but instead, indirectly, via the gastrointestinal tract. This is because atmospheric Pb becomes adsorbed onto surfaces and absorbed into other media (i.e. soil, water etc.). It is then able to enter the body through the gastrointestinal tract when materials, themselves contaminated by such media are ingested. The main pathways by which atmospheric Pb contributes to blood Pb levels are indicated in Figure 7.2.



[Source: Modified from WHO (1987)]

Figure 7.2: Main pathways by which atmospheric Pb contributes to blood Pb levels

It is apparent, therefore, that if estimates of the overall blood Pb levels in the general population and the possible influence of BRM and MIM Pb life-cycle processing operations on these levels are to be attempted, the contributions of these other exposure routes also need to be included in the assessment. This is examined further in section 7.4.

b) Other metals and metalloids:

Table 7.1 indicates that the atmospheric emissions of Cd, As, Cu and Zn are considered to be of intermediate relative concern for the lead industry, and those of lowest relative concern to be Sn, Sb, Se and Te. For the substance of intermediate concern, the approximate global annual atmospheric emissions have been estimated to be as follows:

- Cd 7,570 tonnes,
- As 18,820 tonnes,
- Cu 35,370 tonnes, and
- Zn 131,880 tonnes [Nriagu and Pacyna (1988)].

Whilst for the substances of lowest relative concern, they have been estimated to be:

- Sn 6,140 tonnes,
- Sb 3,510 tonnes, and
- Se 6,320 tonnes [Nriagu and Pacyna (1988)].

No data were provided by Nriagu and Pacyna (1988) for Te.

For the metals and metalloids of intermediate relative concern, Table 7.6 indicates the relative mass contributions of the major categories, associated with the lead and associated industries, to total annual global anthropogenic emissions to the atmosphere.

Emission category	**Estimated annual emissions to the atmosphere (tonnes)**							
	Cd		**As**		**Zn**		**Cu**	
	Range*	**Median ****	**Range***	**Median ****	**Range***	**Median ****	**Range***	**Median ****
Mining of non-ferrous metal ores	0.6 - 3	1.8	40.0 - 80	60	310 - 620	465	160 - 800	480
Pb prod'n.	39 - 195	117	780 - 1,560	1170	195 - 468	332	234 - 312	273
Cu-Ni prod'n.	1,700 - 3,400	2,550	8,500 - 12,750	10,625	4,250 - 8,500	6,375	14,450 - 30,600	22,525
Zn-Cd prod'n.	920 - 4,600	2,760	230 - 690	460	46,000 - 82,800	64,400	230 - 690	460
Total		**5,429**		**12,315**		**71,572**		**23,738**
Total global emissions	7,570		18,820		131,880		35,370	
Fraction of total attributed to the lead & associated industries	**72%**		**65%**		**54%**		**67%**	
Notes:	* Data source: Nriagu and Pacyna (1988). ** Calculated by the author.							

Table 7.6: Relative contributions of the major categories of the lead and associated industries to emissions of intermediate relative concern

Hence, the various processes associated with the lead and associated industries account for 72, 65, 54 and 67% of the total anthropogenic emissions to the atmosphere of Cd, As, Zn and Cu respectively. The Cu-Ni and Zn-Cd production emissions of Cd, As, Zn and Cu have been included in the calculations in Table 7.5, since they are either co-products or by products of these processes. However, it may be argued with some justification, that not all of the Cd, As, Zn or Cu emissions, from either non-ferrous metal ore mining in general or from Cu-Ni and Zn-Cd production in particular, should be attributed to the lead and associated industries. Thus, it could be argued, a proportion of them some ought be attributed instead to the Cu, Ni, Zn, and Cd production industries. To achieve this, some form of partitioning (i.e. allocation) would be needed. The question of its applicability and the most appropriate apportionment approaches to apply are considered in some detail, with respect to the specific partitioning issues encountered in the BRM and MIM case studies (chapters 8 and 9 respectively).

With respect to cadmium (Cd), volcanic action appears to be the largest source of natural emissions, whilst the steel industry, and Zn-Cd production appear to be the largest sources of anthropogenic emissions [WHO (1987)]. In rural areas, the yearly means of Cd in air concentrations range from <1 to 5 ng/m^3, in urban areas from 5 to 15 ng/m^3 and in industrialised areas from 15 to 50 ng/m^3 [WHO (1987)].

At concentrations of 50 ng/m^3, the average daily Cd intake would be more than 1 μg [WHO (1987)]. As this represents the highest end of the typical atmospheric concentration range, the daily Cd intake from the air alone will be less than this for most of the general population. Cigarette smoking may also contribute significantly to Cd intake. Each cigarette may contain from 0.5 to 3 μg/g of tobacco. When a person smokes, 10% of this Cd is emitted into the mainstream smoke and 10 to 20% of this Cd may be then absorbed by the lungs. Thus, a person smoking 20 cigarettes per day will have a daily intake of up to 3 μg [WHO (1987)].

In long-term low-level (i.e. chronic) exposures, which are the norm for the general population, the kidney is regarded as the critical organ. This is because renal dysfunction may occur when the Cd concentration in the kidney cortex reaches about 200 mg/kg wet weight [WHO (1987)]. Damage is characterised by proteinuria, amino aciduria, glycosuria and an increase in Cd excretion (accompanied by a decrease in its half-life in the body). Such changes are thought to be a result of damage to the nephron tubule epithelial cells. The damage also appears to be irreversible and progressive [Robertson (1994)].

Other effects of chronic exposure, identified from human and animal studies, are:

- decalcification and lesions in the bones (due to damage to the osteoclasts and osteoblasts),
- hypertension (due to arteriolar constriction),
- increased atherosclerosis and cardiovascular disease,
- immunotoxicity (due to the inhibition B-cell activation by $CdCl_2$, which may, therefore, influence the ability of B-cells to produce specific antibodies),
- testicular atrophy,
- reduction in life span and rate of growth,
- anaemia,
- placental damage,
- neurbiological and behavioural effects,
- teratogenicity, and
- interference with DNA and various enzyme systems [Robertson (1994)].

However, such effects will be prevented if the renal burden of Cd is kept low [WHO (1987)]. Indeed, the monitoring of urinary Cd is an effective way of detecting the risks of renal and other dysfunctions. The critical concentration in the urine is 10 μg/g of creatinine, though 5 μg/g of creatinine is recommended as a "safe limit" [WHO (1987)].

Acute respiratory effects (presenting as an acute chemical pneumonitis) are associated with Cd fume in air concentrations above 1 mg/m^3. Chronic respiratory effects are also associated with occupational exposures of 20 $\mu g/m^3$ for about 20 years [WHO (1987)].

Existing Cd in air levels are believed to give a large margin of protection, with regard to toxic effects through inhalation on the lungs and kidneys in the general population. However, atmospheric Cd becomes precipitated into soil, is readily taken up by plants and by this means becomes ingested by humans [WHO (1987)]. Thus, combined exposures are the norm. These are examined in section 7.4, along with the problems the build of Cd in the food chain may pose to future generations.

Cd has been classified by the International Agency for research on Cancer (IARC) as a Group 2B carcinogen, on the basis that there is evidence of it being carcinogenic to humans [WHO (1987)]. However, estimates of the cancer risks posed by Cd are very uncertain, because data based on human studies are problematic. In this context, WHO (1987) state that, whilst existing Cd levels in the urban and rural environment may be tolerated, an increase in existing levels should not be permitted.

Arsenic (As) is released into the atmosphere from both natural and anthropogenic sources. The former are principally from volcanic activity and the latter from the smelting of metals, the combustion of coal (especially brown coal) and the use of pesticides [WHO (1987)]. Representative background concentrations of As in air, in rural areas range from 1 to 10 ng/m^3, but in some cities can reach several hundred ng/m^3, and in the vicinity of some non-ferrous metal smelters and power plants may exceed 1000 ng/m^3, depending on the As content of the fuel being combusted [WHO (1987)]. Therefore, assuming a breathing rate of 20 m^3/day, the uptake rates from atmospheric As particulates in rural areas are 0.0006 to 0.006 μg/day, and in urban areas are 0.06 to 1 μg/day [WHO (1987)]. Cigarette smoking may also contribute to As intake, especially when the tobacco plants (used to make the cigarettes) have been treated with lead arsenate insecticide. It is estimated that about 6 μg of As may be inhaled per pack of cigarettes smoked, and about 2 μg may be retained in the lungs [WHO (1987)].

The major routes of As uptake are through inhalation and ingestion. Renal clearance also appears to be the major route of excretion of absorbed As in humans and animals [WHO (1987)]. Occupational exposures occur primarily in workers:

- in metal-smelting industries,
- at some coal burning plants, and
- using or producing pesticides [WHO (1987)].

Lung cancer is considered to be the critical effect following inhalation. Indeed, an increased incidence of lung cancer has been identified in several occupational groups exposed to inorganic As compounds. A positive interaction between AsO_3 and benzo [*a*] pyrene has also been suggested, but not proven conclusively [WHO (1987)]. As is carcinogenic because it has been shown to be clastogenic and induces sister chromatid exchanges in a variety of mammalian cells, at least in vitro. At relatively high exposure levels, it has also been shown to be teratogenic in a number of mammalian species [WHO (1987)]. A moderately increased risk of cancer has also been identified in a number of populations living near inorganic As emission sources, such as smelters [WHO (1987)]. No known safe threshold exists, with respect to cancer induction resulting from As exposure. Therefore, a conservative estimate of the risk of cancer occurring per member of population, at an air concentration of 1 $\mu g/m^3$ over a life time, is 3×10^{-3} [WHO (1987)] (i.e. a 1 in 333.33 increase from the normal background incidence). This exposure level is well above those found normally in rural and urban areas, though unprotected workers in significant As emitting industries could be thus exposed. It also, of course, underlines the need for effective respiratory protection measures for such workers. Fortunately, such measures exist, at least within the OECD group of industrialised countries and, to some extent, in other countries too.

Zinc (Zn) and copper (Cu) emissions to the atmosphere are considered to be of intermediate relative concern for the lead and associated industries, on the basis that significant quantities of both are released. However, unlike Pb, Cd and As which have no known useful function in the human body, Zn and Cu are essential as micro-nutrients. Indeed, much of the literature on the human health effects of Zn and Cu concerns their deficiency rather than overload [Robertson (1994)].

Zn is required by all plant and animal life. It is incorporated into over 20 different metalloenzymes and over 100 enzymes require it to function. Indeed, after Fe, it is the most commonly occurring metal in human tissue. The main routes of uptake of both Zn and Cu are via food and drink, rather than via the air. However, significant occupational inhalation exposures, might occur in the non-ferrous metal processing industries.

Cu overload leads to stomach distress, cramps, nausea, vomiting and diarrhoea. However, the body is able to eliminate excess Zn through the gastrointestinal system. Hence, there is little risk of long-term accumulation in the body [Robertson (1994)]. Nevertheless, where daily total exposures are 150 mg/day and greater, an increased risk of heart disease may occur. In addition, reductions in liver and kidney function and some impairment of the immune system may occur as a result of sub-acute Zn intoxication [Robertson (1994)].

Whilst Cu is essential for life, it is toxic to living systems. However, even though Cu toxicosis can be fatal, it normally only ever occurs in human beings, who are unable to excrete the excess, due an inherited defect in one or more of their homeostatic mechanisms [Robertson (1994)]. Therefore, not only is Cu toxicosis rare, but is usually not serious. Where it does occur, the initial symptoms are nausea and vomiting, which may then be followed in severe cases by a deepening coma, oligouria, abnormal hepatic mitochondrial respiration, abnormal cytochrome C oxidase activity, hepatic necrosis and death. In addition, cytological investiagtions have identified that Cu overload results in alterations to lysosomal morphology and pH, increased lysosomal fragility, decreased membrane fluidity and alterations to membrane fatty acid composition [Robertson (1994)].

Finally, since this section is intended to provide a review of all of the atmospheric emissions deemed of current concern for the lead and associated industries, those viewed to be of minor significance (i.e lowest relative concern in Table 7.2) also need to be considered. Their total annual global emissions to the atmosphere and the relative mass contributions to these totals, of the major categories associated with the lead and associated industries are indicated in Table 7.7.

Emission category	Estimated annual emissions to the atmosphere (tonnes)					
	Sn		Sb		Se	
	Range*	Median**	Range*	Median**	Range*	Median**
Mining of non-ferrous metal ores			18 - 176	97	18 - 176	97
Pb prod'n.			195 - 390	293	195 - 390	293
Cu-Ni prod'n.	425 - 1,700	1,063	427 - 1,700	1,064	427 - 1,280	854
Zn-Cd prod'n.			46 - 92	69	92 - 230	161
Total		**1,063**		**1,523**		**1,405**
Total global emissions	6,140		3,510		3,790	
Fraction of total attributed to the lead & associated industries	**17%**		**43%**		**37%**	
Notes:	* Data source: Nriagu and Pacyna (1988). ** Calculated by the author. Bank spaces denote an insignificant contribution to that category. No data were provided for Te emissions to the atmosphere.					

Table 7.7: Relative contributions of the major categories of the lead and associated industries to emissions of lowest relative concern

Thus, the estimated global annual atmospheric emissions of Sn, Sb and Se individually are less than those for Cd, are an order of magnitude less than for As and Cu, and are two orders less than for Pb and Zn. The percentage contribution to the estimated total global annual atmospheric emissions of Sn, Sb and Se are also less than all of the other significant metals and metalloids of concern. In addition, it may be inferred from the lack of data by Nriagu and Pacyna (1988) on Te, that anthropogenic emissions were not perceived to be a global problem at the time.

Ambient atmospheric concentrations of Sn have been shown to vary from 0.002 to 0.3 μg/m^3 in the USA, and Sb to vary from 0.29 to 4.8 ng/m^3 in the UK. These represent daily intakes of 0.03 μg to 0.45 μg for Sn and 4.4 to 4.8 ng for Sb (assuming a mean inhalation volume of 15m^3/day) [MAFF (1985)]. Estimates of the uptake of Sn and Sb from food and water vary from country to country and according to diet. However, MAFF (1985) report the intake of Sn from food in the UK, from various surveys, to be between 2 and 15 mg/day. For Sb, the intake in the UK, is 29 μg/person/day (upper bound) and 2μg/day (lower bound).[20], whilst in New Zealand it is reported to be less than 12 μg/day, and in Germany to have a mean of 23 μg/day. Values for Sb intakes from food in the USA have been reported, which are about ten times higher [MAFF (1985)]. Uptakes from water, are assumed in MAFF (1985) to be 9 μg/person/day for Sn, and 15 μg/person/day for Sb.

With respect to Se, concentrations in ambient air are generally very low (with an average concentration believed to be below 0.07 fg/m^3), whilst Se uptake from food is estimated to be between 0.07 to 0.152 mg/day [ATSDR (1989)]. Therefore, it may be concluded, that the major route of human exposure to Sn, Sb and Se is via food and to a lesser extent drink, rather than via inhalation.

Inorganic Sn, which is the main form of Sn emitted by the lead and associated industries, is relatively non-toxic to humans. MAFF (1985) reports that the Food and Agriculture Organisation of the World Health Organisation (FAO/WHO) Joint Expert Committee on Food Additives have set a provisional maximum tolerable daily intake of 2 mg/kg of body weight which, for an adult with an average weight of 70 kg, corresponds to an intake of 0.14 g/day. Hence, total intake of Sn in the general population is well below the tolerable limit. Therefore, the direct human health effects upon the general population from the Sn emitted to the atmosphere by the lead and associated industries are expected to be negligible.

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[20] The 'upper bound' value has been calculated on the assumption that individual values less than the limit of detection of the measurement technique are equal to that value, whilst the 'lower band' values assume that individual values less than the limit of detection are zero.

The US EPA have established Reference Doses (RfD) for oral exposure, for both Sb and Se. The RfD is an estimate of the daily oral exposure to the human population (including sensitive groups) considered likely to be without an appreciable risk of deleterious effects during a lifetime. For Sb it is 0.0004 mg/kg of body weight/day and for Se it is 0.015 mg/kg of body weight/day (i.e. 0.028 mg/day for Sb and 1.05 mg/day for Se, assuming a typical average body weight of 70 kg). However, inhalation Reference Concentrations (RfC) have not been established for either element. In addition, neither element has been shown to be carcinogenic. Indeed, Se is an essential element in human nutrition and has recommended daily intakes of 0.07 to 0.55 mg/day for men and women, and 8.7×10^{-4} mg for infants [ATSDR (1989)]. Thus, total exposures in the general human population to Se appear unlikely to give rise to any detectable effect. For Sb, however, it is possible that some chronic human effects may be experienced, since the upper bound estimate of the daily intake from food is 29 µg/day (i.e. 0.029 mg/day) and the uptake from drinking water may be 15 µg/person/day (giving a total oral uptake of about 0.044µg/day), whilst the RfD is slightly less at 0.028 mg/day.

Animal studies of the effects of oral exposures have indicated that chronic Sb exposure can give rise to gastrointestinal effects and effects on the blood, liver, and central nervous system (CNS) [ATSDR (1992)]. Whether these are actually experienced in the general population, as a result of total Sb exposures is unknown.[21] However, since the typical uptakes of Sb from the atmosphere are between 4.1 and 4.8 ng/day, the contributions of atmospheric Sb to any such effects are expected to be very minor.

In conclusion, therefore, environmental exposures to atmospheric Sn and Se are not expected to give rise to significant human health effects in the general population. Also, any human health effects from atmospheric Sb, in addition to those which may already be occurring from dietary Sb, are likely to be negligible. For Te there is insufficient information to ascertain whether atmospheric emissions give rise to any significant human health effects. However, the world refinery output is currently only 90 tonnes [Bourge (2000)]. Therefore, human health effects are likely to be very localised and, for the general population, to be negligible.

3. Summary:

Section 7.1.2 may be summarised as follows:

1. Whilst substantial and significant and substantial uncertainties exist in attempts to determine the contributions of anthropogenic atmospheric emissions to global fluxes, it appears, on the balance of the evidence, that trans-boundary metal transport has occurred on a global scale.
2. Exposures to Pb, plus other metals and metalloids are normally a combination from more than route of entry into the body. (The effects of combined exposures are reviewed in section 7.4.)
3. The atmospheric emissions of some metals and metalloids, from lead industry processes, are considered to be more significant than others. Pb is considered to be of greatest relative concern, since it is the greatest mass emission and has tended to receive the most attention (by the lead and associated industries, the scientific community, regulators and the general public). Cd, As, Zn and Cu are of intermediate concern, since their mass emissions are less and they have received less attention. Whilst Sn, Sb, Te and Se are of the least relative concern, since their mass emissions are the least and/or they have received the least attention.
4. Anthropogenic Pb emissions to the atmosphere are from both stationary and mobile sources. The mobile sources are from the use of Pb-based additives in petrol. Such additives are currently either being phased out altogether or reduced in most OECD countries. Hence, globally, this source is likely to show a significant decline in relative importance. Stationary sources are from both point sources (i.e. chimney stacks) and fugitive releases.
5. The main sources of Pb in dusts, currently, (both indoors and outside) are from vehicle emissions and the flaking of Pb-based paints.
6. The human health effects of Pb have been studied extensively. Pb gives rise to a variety of effects. Lowest observed effect levels have been identified for different categories of effects, at different blood Pb concentrations in adults, children and in the foetus.
7. The human health effects of Pb may, to some extent, be explained by the fact that it inactivates enzymes by binding with the SH-groups of their proteins, or by displacing other essential metal ions.

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[21] This is an area for further research.

8. Pb has been shown to be able to exert effects on haem synthesis and haematology, the nervous system, the renal system, the cardiovascular system, the reproductive system, and the gastrointestinal system. However, it has not been demonstrated conclusively that Pb is a carcinogen.
9. Pb exposures in the general population are generally long-term and low-level (i.e. chronic).
10. A significant proportion of the Pb entering the human body tends to be via the gastrointestinal system, as well as via the lungs. Hence, estimates of the possible influence of the lead industry in general, and BRM and MIM Pb life-cycle processes in particular, on the general population, can only be attempted if these other sources are also considered. (See section 7.4.)
11. Whilst existing Cd in air levels give a large of margin of safety, with regard to toxic effects through inhalation, atmospheric Cd becomes precipitated. It is then taken up readily by living organisms, passes along food chains, and thereby becomes ingested by humans.
12. Cigarette smoking can also make a significant contribution to personal Cd exposure.
13. Cd is recognised to be a carcinogen. The WHO (1987) argue, that while current levels in the urban environment may be tolerated, an increase in existing levels should not be permitted.
14. As is also carcinogenic and with no known threshold. Lung cancer is considered to be the critical effect of As exposure. It is possible that the current As in air concentrations may make a small contribution to the current incidence of cancer in the general population.
15. Zn and Cu are essential nutrients. Their main routes of uptake are via food and drink rather than the air. Indeed, much of the literature is concerned with deficiency rather than overload. Cu toxicosis is also rare. Therefore, overload from inhalation exposure is considered unlikely in the general population
16. Sn and Se total exposures, by all routes of entry into the body, are unlikely to give rise to any significant human health effects in the general population. In addition, the human health effects of Sb exposures, via inhalation, are likely to be insignificant compared with any possible effects which might be associated with current levels of exposure from food and drink.
17. There is insufficient information to ascertain whether Te gives rise to any significant effects in the general population. However, since the global production of Te from

refineries is only about 90 tonnes [Bourge (2000)], such effects, if any, will probably be localised and minor.

7.1.3. Sulphur Dioxide and Related Compounds

1. Introduction

Sulphur compounds leave the BRM stacks and those plants associated with MIM's Pb life-cycle, as part of the particulate atmospheric emissions (as sodium sulphate (Na_2SO_4) and sulphuric acid (H_2SO_4)), and as SO_2. The human effects of the particulate, metal and metalloid components of these emissions were considered in the previous sections of this chapter. This section is concerned solely with the human effects of atmospheric SO_2 emissions and of the secondary and higher order substances it forms in the environment. The physico-chemical nature of these emissions and the secondary and higher order substances which they form are reviewed in section 1.3.1.1 of Document 24, in volume 5 of the Portfolio.

This section considers the evidence for human health effects from ambient UK concentrations of atmospheric SO_2 (and related compounds) in relation to atmospheric sulphur dioxide (SO_2) and related compounds in general and, where appropriate, to the SO_2 and related compounds emitted from MIM's lead life-cycle and BRM processes in particular.

A summary of the main conclusions drawn by the author is provided at the end of the section.

2. The human health effects of ambient UK and Australian concentrations of atmospheric SO_2 (and related compounds)

Extensive reviews of the literature regarding the potential human health effects of ambient UK concentrations of air pollutants have been published. For SO_2 and related compounds, the reports of major relevance are:

- 'Quantification of the Effects of Air Pollution on Health in the United Kingdom' COMEAP (1998),
- 'Urban Air Quality in the United Kingdom' QUARG (1993),
- 'Sulphur Dioxide, Acid Aerosols and Particulates' MAAPE (1992) and
- 'Sulphur Dioxide' EPAQS (1995b).

Therefore, they have been used as the primary reference sources for this section of the thesis. However, other references are also cited where appropriate.

Epidemiological investigations of the effects of SO_2 on humans have focused on acute and chronic effects. In such studies, the relationship between concentrations and daily variations of SO_2 with mortality and morbidity indicators (such as health service usage) have been investigated. Experimental challenge studies to identify the toxic effects of SO_2 on panels of patients and healthy volunteers have also been conducted.

With respect to the interpretation of acute epidemiological studies, difficulties (i.e. uncertainties) exist due to the presence of confounding factors, such as season and temperature, which may cause SO_2 to accumulate in some weather conditions and disperse in others [COMEAP (1998)]. Problems also exist due to the fact that, under ambient conditions, humans are liable to be exposed simultaneously to several pollutants. In addition, SO_2 and particulate emissions often arise from the same sources and so tend to be correlated. This means that it is often difficult to distinguish between their respective effects [COMEAP (1998)]. The situation is further complicated by the fact that SO_2 also contributes to the formation of sulphates which are secondary particulates [COMEAP (1998)]. Other factors which can complicate assessments are that:

- levels of activity and the type of breathing will have a significant effect on the dose of the pollutant received,
- indoor sources of SO_2 may also exist which would enhance exposures and
- people may move in and out of the area during the course of a study [MAAPE (1992)].

Despite the interpretational difficulties, exposure-response relationships of SO_2 with mortality [Ryder et al. (1980)] and with morbidity (through endpoints such as hospital admissions [Ryder et al. (1980)], respiratory disease [Ryder et al. (1980)] and health impacts [Katsoyanni et al. (1997)]) have been detected. Furthermore, it has been argued that these appear to be detectable at levels well below the air quality guideline value (of $350\mu g/m^3$) which is recommended by the World Health Organisation [Xu et al. (1995)]. Nevertheless, the Advisory Group on the Medical Aspects of Air Pollution Episodes argues that available evidence seems to indicate, that individuals who do not suffer from respiratory disease will be unaffected by elevated episodes of SO_2 at the concentrations likely to occur in the UK [MAAPE (1992)]. Asthma patients, on the other hand, appear to be more sensitive to SO_2 [MAAPE (1992), COMEAP (1995a)]. It is reported that in parts of the UK, levels of SO_2 regularly exceed those at which effects of clinical significance, such as coughing, wheezing and tightness of the chest may be experienced. Such effects though acute are reversible [MAAPE (1992)]. However, it is recognised that, at times, average concentrations of SO_2 might exceed 400 p.p.b. (i.e. $1144\mu g/m^3$). If this occurs, it is thought that asthmatics may experience symptoms of a severity where it is advisable to limit their exposure and possibly increase their treatment. However, for non-asthmatics, the effects are thought likely to be mild [MAAPE (1992)]. It also appears that only fairly small changes appear to be necessary to give rise to changes in health effects such as in the amount of wheezing, bronchodilator use and reduced peak expiratory flow rates [COMEAP (1998)].

Apparently, there also appears to be some evidence of chronic effects. However, as with particulates, the uncertainties caused by confounders such as temperature and seasonal variations (as well as factors such as smoking) tend to be greater than for acute effect studies. After allowing for such variables, the size of these effects appears to be small. Where they do occur, however, they appear to affect susceptible people such as those with problems of wheeze [MAAPE (1992)].

COMEAP note in conclusion, that there is little doubt that SO_2 both causes and aggravates symptoms, especially for those with pre-existing asthma. They also point out that exposure to SO_2 and particulates in association appears to increase mortality both in the short and long term. However, they note that it is uncertain which element is responsible for this or whether the effect is truly synergistic [EPAQS (1995a)].

Thus, under certain atmospheric conditions (such as during a temperature inversion) when the stack plume is forced down to ground level, it is possible that ambient concentrations resulting from any of the stacks venting SO_2 emissions from BRM and other MIM lead life-cycle processes could reach levels where acute effects may be visible. Whether this occurs in practice or not is unknown. The severity of such events could be estimated using dispersion modelling software such as ADMS and ISC. However, the short-term concentration estimations provided by such approaches are considered by the authors to be highly uncertain and do not provide a reliable enough of an estimate upon which to base judgement.

As far as potential chronic human health effects from the SO_2 emitted from either BRM, or from any other of the MIM lead life-cycle processes are concerned, these are also unknown, though they too could be predicted using emission dispersion modelling.[22] In the UK, however, chronic effects are minor. Therefore, their contribution is also expected to be minor. The author has not ascertained the situation in Australia. However, since it is an OECD country, the effects are assumed to be broadly similar.[23]

It should also be borne in mind that in the above discussion, the primary effects only of SO_2 (which is itself a primary pollutant) have been considered. However, the SO_2 emissions are transformed progressively in the environment. It was noted, in section 4.3.1 of chapter 4, that secondary, tertiary and higher order effects may also occur at each stage in this process. For example, SO_2 is oxidised to the secondary pollutant H_2SO_4 which then contributes to acidification of the environment. If severe enough, this could result in the environmental mobilisation of various metals which may then be taken up by humans. Potentially at least, this could cause deleterious effects. (These effects would be secondary human effects in relation to the initial SO_2 emission.) Ideally, all transformations and effects should be considered if assessments of the impacts of emissions are not to be an underestimation. However, as was also noted, substantial uncertainties exist in our current abilities to both model and track these higher order effects. For this reason, assessments are

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[22] Due to time constraints in the preparation of this thesis, it has not been possible include this as a case study. However, preparatory work for an emission source scale case study has been conducted. The need for such assessments in an integrated strategy to meet the environmental challenge, is considered in the Overall Conclusions and Recommendations (chapter 11).

[23] This is an area for further research.

often limited to the effects of primary pollutants (as is the case with SO_2). In some cases, however, as in the effects of NO_x (which will be discussed in the next section) the effects of secondary pollutants may be of equal or of even greater significance than the effects of the primary pollutants.

3. Summary

Section 7.3 may be summarised as follows:

1. Exposures to SO_2 tend to occur simultaneously with other pollutants and it has been difficult to distinguish the influence of the SO_2 from these other pollutants. Interpretation is further complicated by the facts that SO_2 is oxidised in the atmosphere to form sulphates (secondary particulates) and that confounding factors exist which are difficult or impossible to eliminate.
2. Despite such difficulties, an exposure-response relationship for acute effects with both mortality and morbidity indicators has been demonstrated [Ryder et al. (1980)].
3. As with particulates, healthy people are considered unlikely to be affected by current ambient UK concentrations of SO_2. However, people with predisposing factors such as asthmatics and people with chronic heart disease may experience some acute effects [MAAPE (1992)].
4. It is not known whether the emissions from any of the MIM lead processing operations or from BRM processes ever give rise to significant acute human health effects. However, it is theoretically possible that under certain environmental conditions they might do so. On the other hand, chronic effects, if any, would be expected to be minor.

7.1.4. Nitrogen Oxides and Related Compounds

1. Introduction

The NO_x emissions from processes involved in the life-cycle of MIM's Pb and from the BRM site originate mostly as a result of the high temperature combustion processes occurring within it. Their source will be from the oxidation of the nitrogen in the combustion air and/or from any of the nitrogen compounds in the fuel or from any other material being burned. NO is the major oxide initially emitted from such processes. The amount of NO_2 emitted from such combustion processes is a function of the temperature, combustion conditions and other factors [QUARG (1993)].

The physico-chemical nature of NO_x emissions and their behaviour in the atmosphere are reviewed in section 1.2.1.2 of Document 24, in volume 5 of the Portfolio.

This section considers the following in relation to atmospheric nitrogen oxides (NO_x) in general and, where appropriate, to the NO_x and related compounds emitted from processes involved in the life-cycle of MIM's Pb and the BRM site in particular:

- the evidence for human health effects from ambient UK and Australian concentrations of NO_x, and
- the human health effects of secondary pollutants dependant upon NO_x.

A summary of the main conclusions drawn by the author is provided at the end of the section.

2. The human health effects of ambient UK and Australian concentrations of atmospheric NO_x

Extensive reviews of the literature regarding the potential human health effects of ambient UK concentrations of air pollutants have been published. For NO_x, the reports of major relevance, used as the primary reference sources for this section of the thesis, are:

- 'Quantification of the Effects of Air Pollution on Health in the United Kingdom' COMEAP (1998),
- 'Urban Air Quality in the United Kingdom' QUARG (1993),
- 'Oxides of Nitrogen' MAAPE (1992a) and
- 'Nitrogen Dioxide' EPAQS (1995).

Other references are also cited where appropriate.

As with SO_2 and particulates, epidemiological investigations have focused on acute (short term) and chronic (long term) effects and have attempted to relate them to endpoints such as mortality and indicators of morbidity. Experimental challenge studies, to try to identify the toxic effects of NO_2 on healthy and asthmatic humans, have also been conducted. The focus of these studies has been almost exclusively on NO_2, as it is an irritant gas and is known to have serious and sometimes fatal effects when inhaled in very high concentrations. NO, on the other hand, has not been a focus of attention because it is produced naturally by living cells and is considered to have no harmful consequences when inhaled by humans at the concentrations likely to occur in the ambient atmosphere [EPAQS (1995)]. MAAPE note, however, that data on the health effects of NO_2 is limited because epidemiological studies have tended to concentrate upon other pollutants [MAAPE (1992a)].

In common with epidemiological studies in general, problems have existed in interpreting such studies due to the fact that under ambient conditions, humans will normally be subject to a cocktail of pollutants rather than just to NO_x. Therefore, synergistic, additive or even antagonistic effects on human health might occur as a result of interactions between the various components. This is thought to be a significant problem, since the composition of urban air can vary significantly diurnally, seasonally and during episodes of heightened atmospheric pollution [MAAPE (1992a)]. Indeed, COMEAP (1998) consider that the control of such variables has not been adequate in many studies which have reported an association between human health effects and NO_2.

Similarly, confounders (such as temperature, weather conditions and tobacco smoking) and other difficult to control variables have increased the uncertainty (and hence reduced the sensitivity) of such studies. These other variables include the following.

1. Unflued natural gas or propane gas appliance can give rise to high and intermittent indoor exposures.
2. Steep gradients tend to occur between the sources and sites of exposure,
3. There is uncertainty regarding the relative importance of peak versus average exposures.
4. Both indoor and outdoor exposures may be highly variable in time and space [MAAPE (1992a)].

Indeed, COMEAP (1998) consider that, of the major air pollutants, the effects of NO_2 are the most difficult to assess at the moment due to its dependency upon such a wide range of modifying influences.

Controlled animal studies of prolonged exposure at high concentrations have demonstrated that NO_2 induces damage to epithelial cells lining the lung and that such exposure appears to increase susceptibility to bacterial and viral infections. However, when inhalation exposures have been conducted on healthy humans, at concentrations comparable with or above those found in the ambient environment during air pollution episodes, little or no effects have been demonstrated [MAAPE (1992a), EPAQS (1995)]. Also, even for asthmatics the effects appear to be trivial and inconsistent [MAAPE (1992a)]. Thus, there is little evidence that asthmatics are more sensitive to NO_2 than normal individuals [MAAPE (1992a)].

MAAPE have argued that where epidemiological studies have been able to demonstrate apparent associations there has been an unwillingness to attribute it to NO_x. They argue, that this might be due to the presence of such experimental evidence and because of the above mentioned interpretational problems inherent in epidemiological studies.[MAAPE (1992a)]

The problem of trying to identify the human effects of NO_2, has been heightened by the fact that epidemiological studies have not been able to demonstrate consistent effects at ambient concentrations [COMEAP (1998)] and that there does not appear to be a linear dose-response relation [MAAPE (1992a)]. However, there is some epidemiological evidence to suggest that short term effects may occur at concentrations well below the World Health Organisation guideline levels [MAAPE (1992a), EPAQS (1995)]. However, whether this is really due to NO_2 alone or in conjunction with other pollutants is unclear [EPAQS (1995)].

MAAPE conclude that current evidence suggests that, in terms of acute effects, individuals who are not suffering from respiratory disease will be unaffected by the type of episodic periods of elevated NO_x concentrations such as occur currently in the UK. However, persons suffering from respiratory disorders (including asthma) may experience a worsening of their symptoms during such periods [MAAPE (1992a)]. The author has not ascertained the situation in Australia. However, since it is an OECD country, it is assumed to be broadly similar.[24]

Chronic effects have been even more difficult to assess, since much of the data are contradictory. However, there is some evidence to suggest that prolonged exposure to elevated levels of NOx may enhance susceptibility to allergens such as the dust mite. It has also been suggested, though not confirmed in controlled tests, that it might also reduce lung function [EPAQS (1995)].

NO_2 is an oxidising agent and it is thought that this may be the reason for its effects which are damage to cell membranes and proteins. Also, whilst it is known that NO_2 could react with amines in the body to produce nitrosamines (which are known to be carcinogenic), carcinogenicity has not been demonstrated under ambient conditions.

As with SO_2, under certain conditions, the NOx emissions from stacks serving BRM and/or MIM Pb life-cycle processes might be brought to the ground without dispersing substantially. Under such conditions, acute effects might be experienced by susceptible individuals. However, since no consistent dose-response relationship has been demonstrated for NO_x, whether any effects could be attributed to the NO_x alone is highly uncertain.

3. The effects of secondary pollutants dependent upon NO_x.

Introductory

It has been noted that many potentially reactive nitrogen oxides exist in the atmosphere and that some species are considered to be relatively innocuous whilst others are recognised as having the ability to produce deleterious human effects. However, epidemiological study data and/or controlled exposure test data for species other than NO_2 tends to be either limited or non-existent. Therefore, assessment of the potential effects of these substances in this report has not been possible.

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[24] This is an area for further research.

Some of the atmospheric nitrogen oxide species are primary pollutants (i.e. released in that form), whilst others are secondary (i.e. produced via atmospheric photochemistry). Still others, such as NO_2, may be both primary or secondary, as some of it may be released directly from emission sources whilst the bulk is secondarily created. Regardless of their origin, all nitrogen oxide species will be progressively transformed in the environment. Each of the new species created may, potentially at least, affect human health in some way. However, as was noted in the previous sections, substantial uncertainties exist in our abilities both to model the behaviour of species and to track their effects. Nevertheless, if one is aiming to provide a comprehensive assessment of the potential environmental effects of emissions (as is the purpose of this document), focusing purely on the primary pollutants might lead to a significant underestimation of the overall human effects of emissions.

This would certainly be the case with NO_x, since the primary pollutants are considered in this report to be the NO and the small amount of NO_2 which may be released. Since, NO is considered to be relatively innocuous, an assessment which simply focused on primary pollutants would grossly underestimate the impact of such emissions. Thus, the potential health effects of all of the secondary pollutants ought to be considered if a comprehensive assessment of the potential effects of the emissions is to be provided. Ideally, this assessment ought to extend beyond the first order secondary pollutants (i.e. those formed initially from primary pollutants such as the bulk of the NO_2), through all of the higher order pollutants until they finally decompose into a quasi-natural state. With current modelling and tracking methodologies this is obviously not possible, at least not to any extent of reliability because the uncertainties are just too great. However, where sufficient knowledge about a higher order pollutant exists, such that it may be identified as being potentially significant, it ought to be included in the assessment.

The production of tropospheric O_3 is a secondary effect of anthropogenic NO_x emissions since its generation is dependant upon them. It is thus also an example of a higher order secondary pollutant. It is appropriate to consider it in this Report, since it is a significant pollutant whose production is dependent upon NO_x. Therefore, a brief review of its potential human effects is provided here.

The human effects of tropospheric O_3

As with the pollutants discussed previously, extensive reviews of the human health effects of ambient UK concentrations of tropospheric O_3 have been published. Those of particular relevance to this report are:

- 'Quantification of the Effects of Air Pollution on Health in the United Kingdom' COMEAP (1998),
- 'Ozone' MAAPE (1991) and
- 'Ozone' EPAQS (1994).

Therefore, they have been used as the primary reference sources for the section below.

As with the pollutants discussed previously, epidemiological investigations have focused on acute (short term) and chronic (long term) effects and have attempted to relate them to endpoints such mortality and indicators of morbidity. Experimental challenge studies, to try to identify the toxic effects of NO_2 on healthy and asthmatic humans and animal studies have also been conducted.

The animal studies have shown that, at concentrations which are of a similar order to those found in ambient UK air, histological changes to the respiratory tract occur. (These changes include damage to the cilia lining the respiratory tract and the stimulation of an inflammatory response). However, these changes appear to be reversible [MAAPE (1991)]. It might be postulated, therefore, that such changes would also occur in humans.

Controlled exposure chamber studies on human volunteers have indicated that the dose-response curve appears to be non-linear and that there is no threshold for effects on lung function.[MAAPE (1991)] However, it appears that there is considerable variation in the lung function responses of individuals exposed to O_3 doses which are similar to those found in elevated pollution episodes. There is, apparently, also no evidence that smokers, people with chronic obstructive pulmonary disease, the elderly or asthmatics are any more susceptible to the effects of O_3 than healthy individuals [MAAPE (1991), EPAQS (1994)].

Studies of short term effects have focused on morbidity indicators (such as changes in lung function, respiratory symptoms, effects on hospital attendances and admissions) and mortality data [COMEAP (1998)]. However, as with the pollutants discussed earlier, epidemiological studies of the effects of O_3 upon health have been difficult to interpret due to the fact that O_3 is usually inhaled as part of a pollutant cocktail and that variables and confounders which are difficult or impossible to control will have served to increase their uncertainty (and hence to reduce their sensitivity). Despite this, MAAPE was able to conclude that statistically significant changes in lung function would be expected in people taking vigorous exercise during elevated O_3 pollution episodes and that people who are sensitive to O_3 might show symptoms such as coughing and discomfort when breathing deeply [MAAPE (1991)].

The NO_x emissions from the high temperature combustion processes at BRM and those involved in MIM's lead operations certainly contribute, to some extent, to the formation of tropospheric O_3. However, the actual contributions depend upon the additional presence of other atmospheric pollutants (as explained in section 1.3.1.2 of Document 24, in volume 5 of the Portfolio). Therefore, it is not possible to predict the actual contributions of their emissions.

With regard to the chronic effects of O_3, EPAQS (1995) note that evidence is scanty. Therefore, it is not possible to make an assessment.

4. Summary

Section 7.4 may be summarised as follows:

1. The sensitivity of dosimetric, toxicological and epidemiological studies is generally low. This is due to the fact that ambient NO_x exposures normally occur simultaneously with other pollutants and the presence of confounding factors hampers interpretation.
2. Epidemiological studies have not been able to demonstrate consistence effects at ambient concentrations [COMEAP (1998)] and there does not appear to be a linear dose-response relation [MAAPE (1992a)]. However, there is some epidemiological

evidence to suggest that effects may occur at concentrations well below the World Health Organisation guideline levels [MAAPE (1992a), EPAQS (1995)].

3. In terms of acute effects, it appears that individuals not suffering from respiratory disease will be unaffected by the elevated NO_x episodes as occur currently in the UK. However, those with respiratory disorders (e.g. asthmatics) may experience a worsening of symptoms [MAAPE (1992a)]. The author has not ascertained the situation in Australia. However, since it is an OECD country, it is assumed to be broadly similar.
4. Much of the evidence for chronic effects is contradictory. However, it has been suggested that it may enhance susceptibility to allergens such as dust mites and might reduce lung function [EPAQS (1995)].
5. The potential human health effects of the NO_x emitted from MIM lead processing operations and from BRM processes are unknown, though they are likely to be small.
6. The NO_x emissions from from MIM lead processing operations and from BRM processes also contribute to the generation of tropospheric O_3. From animal studies, O_3 is known to cause damage to the respiratory tract. However, the dose-response relationship appears to be non-linear and to have no threshold [MAAPE (1991)].
7. In common with the other pollutants, interpretation of the human health effects of O_3 has been hampered significantly by the presence of difficult or impossible to control variables (including confounders) and the fact ambient O_3 exposure is usually part of a complex mixture.

7.1.5. Carbon Monoxide

Carbon monoxide (CO) emissions from foreground processes involved in the life-cycle of MIM's Pb and from the BRM site in particular, and from the lead industry in general, originate from combustion processes. These may be divided into emissions from site:

- production processes,
- vehicles,
- heating facilities, and
- power generation plant.

This section considers the evidence for human health effects from ambient UK and Australian concentrations of CO, originating from these processes, both in general and, where appropriate, from the processes involved in the life-cycle of MIM's Pb and BRM in particular.

A summary of the main conclusions drawn by the author is provided at the end of the section.

2. The human health effects of ambient UK and Australian concentrations of atmospheric CO

Extensive reviews of the literature regarding the potential human health effects of ambient UK and global concentrations of air pollutants have been published. For CO, the reports of major relevance, used as the primary reference sources for this section of the thesis, are:

- 'Quantification of the Effects of Air Pollution on Health in the United Kingdom' COMEAP (1998), and
- 'Air Quality Guidelines for Europe' WHO (1987).

CO is one of the most commonly produced and widespread of air pollutants. The total emissions of CO are believed to equal (or even exceed) those of all other pollutants combined. Indeed, estimates of anthropogenic emissions range from 350 to 600 million tonnes per annum. The largest group of sources of anthropogenic CO are from incomplete combustion processes involving carbon containing compounds. These are principally from vehicles, industrial processes, heating facilities and incinerators. However, it is also produced by some non-combustion industrial processes. In addition, CO is widely generated inside homes by combustion appliances (such as gas and oil fired boilers and gas cookers), especially where they are used in poorly ventilated rooms [WHO (1987)].

For cigarette smokers in the general population, however, by far their most importance source of CO is cigarette smoke. It can also act as a significant source of exposure for other people in the same room as a smoker ("passive smoking") [COMEAP (1998)]. Some widespread natural and non-biological sources have also been identified. However, their contribution to CO concentrations in urban areas is considered negligible [WHO (1987)].

The only important route of exposure to CO is via inhalation. CO reacts with haemoglobin (Hb) to form carboxyhaemoglobin (COHb). Hb has an affinity for CO which is 200 times greater than that for oxygen. The binding of Hb to CO, gives rise to a condition known as hypoxia. The exact mechanisms by which CO causes hypoxia are unknown. However, the most widely accepted theory is that once Hb has bound to CO, it is no longer able to pick up oxygen in the lungs and transport it to respiring tissue cells. Hence, the overall oxygen carrying capacity of the blood becomes reduced. The presence of COHb is also thought to be impede the dissociation of oxygen from Hb in the capillaries [WHO (1987)].

Natural background levels of CO are reported to range from 0.01 to 0.23 mg/m^3 [WHO (1987)]. In urban areas, however, traffic is often the major source of CO. Concentrations of CO in the urban air tend to vary according to the weather and traffic density. They also vary according to distance from the emission sources. The 8-hour mean concentrations are generally less than 20 mg/m^3. However, 8-hour mean concentrations up to 60 mg/m^3 have been recorded [WHO (1987)]. CO concentrations tend to be expressed in terms of 8-hour mean concentrations because fluctuations in air CO levels are only slowly reflected in blood CO levels. This is because, it takes 4 to 12 hours for approximate equilibrium to occur between air CO levels and blood CO levels [WHO (1987)].

The average COHb concentration levels in the general non-smoking population are from 1.2 to 1.5%, and in cigarette smokers from 3 to 4%, whilst heavy smokers may reach 10%. However, it needs to be noted that healthy people have endogenous levels of COHb of between 0.5 and 1.0% [WHO (1987)]. Occupational exposures to CO can also be considerable. People particularly likely to be thus exposed, include traffic wardens, traffic police, garage workers, employees at metallurgical, petroleum, gas and chemical plants, and fire fighters. In some workplace situations, air CO levels may exceed 115 mg/m^3, and in garages may even exceed 570 mg/m^3. This may result, in some non-smoking workers having COHb concentrations up to five times higher than the general population [WHO (1987)].

The human health effects of CO, at different % COHb concentrations, are indicated in Table 7.8.

Carboxyhaemoglobin concentration (COHb)	**Effects**
2.3 - 4.3	Statistically significant decrease (3 - 7%) in the relation between work time and exhaustion in exercising young healthy men.
2.9 - 4.5	Statistically significant decrease in exercise capacity (i.e. shortened duration of exercise before onset of pain) in patients with angina pectoris and increase in duration of angina attacks.
5 - 5.5	Statistically significant decrease in maximal oxygen consumption and exercise time in young healthy men during strenuous exercise.
<5	No statistically significant vigilance decrements after exposure to CO.
5 - 7.6	Statistically significant impairment of vigilance tasks in healthy experimental subjects.)
5 - 17	Statistically significant diminution of visual perception, manual dexterity, ability to learn, or performance in complex sensorimotor tasks (e.g. driving)
7 - 20	Statistically significant decrease in maximal oxygen consumption during strenuous exercise in young healthy men

[Source: WHO (1987)]

Table 7.8: Human health effects associated with low-level CO exposure -lowest observed effect levels

Table 7.8 indicates, that among healthy people, a decreased oxygen uptake and a resultant decrease in work capacity is apparent starting at 5% COHb. However, in some studies, small but significant decreases in work time to exhaustion, are apparent down to COHb levels of 3.3 to 4.3%, though maximum aerobic capacity appears not to be diminished. No adverse effects have been reported below 2% COHb [WHO (1987)].

There have been reports of associations between an increased risk of ischaemic heart disease and chronic exposure to elevated CO [COMEAP (1998)]. In addition, CO has been associated with acute effects, such as increased hospital admissions for cardiovascular disease, and congestive heart failure [COMEAP (1998)].

For quantitative hazard evaluation purposes, air CO levels and consequent blood CO levels need to be related. This is achieved using the Coburn formula (indicated in Table 7.9), which predicts the %COHb levels for different air CO exposure times and concentrations, for those engaged in sedentary, light and heavy work. The predictions are for people previously unexposed [WHO (1987)].

CO air concentration		**Exposure time**	**Predicted COHb level (%) for those engaged in**		
ppm	**mg/m^3**		**sedentary work**	**light work**	**heavy work**
100	115	15 minutes	1.2	2.0	2.8
50	57	30 minutes	1.1	1.9	2.6
25	29	1 hour	1.1	1.7	2.2
10	11.5	8 hours	1.5	1.7	1.7
[Source: WHO (1987)]					

Table 7.9: Predicted COHb levels for subjects engaged in different types of work

As smoking is a major contributor to COHb levels, exposure limit guidelines tend to be drawn up to protect non-smokers. WHO (1987) recommends a COHb level of 2.5 to 3.0%, for the protection of the population, including sensitive groups (such as those suffering from angina pectoris). To this end, the WHO recommend the following exposure limits, which have been calculated using the Coburn formula:

1. A maximum permitted exposure of 100 mg/m^3 for periods not exceeding 15 minutes.
2. Time-weighted average exposures:
   60 mg/m^3 (50 ppm) for 30 minutes,
   30 mg/m^3 (25 ppm) for 1 hour, and
   10 mg/m^3 (10 ppm) for 8 hours.

BRM and MIM Pb life-cycle processes, as well as the lead industry in general, certainly contribute to the production of anthropogenic atmospheric CO. The people most at risk of elevated blood COHb levels and from consequent CO induced health effects, are workers involved in operating combustion process plant and site vehicles. However, providing their exposures are kept within the bounds indicated above, such effects should be largely avoided.

The author has not obtained data indicating the contribution to air CO levels, either by such BRM and MIM Pb life-cycle processes or by the lead industry in general. The extent of possible effects on the work force are also unknown.[25]

3. Summary

1. CO is one of the most common and widespread of atmospheric pollutants.
2. The largest sources of anthropogenic CO emissions are from incomplete combustion processes, such as from vehicles, process plant, heating facilities and incineration.
3. The only important route for CO exposure in humans, is via inhalation.
4. In urban areas, traffic is often the most important source of CO. In such areas, CO concentrations tend to vary according to traffic density, weather, and distance from the emission source(s).
5. Normal background concentrations of CO are believed to range from 0.01 to 0.23 mg/m^3. In urban areas, the 8-hour mean CO concentrations are generally less than 20 mg/m^3 [WHO (1987)].
6. Occupational exposures by traffic wardens, traffic police, garage workers, employees at metallurgical, petroleum, gas or chemical plants, and by fire fighters may be much higher than the general population.
7. CO binds with haemoglobin in the blood to form carboxyhaemoglobin (COHb). The average COHb concentration levels in the general non-smoking population, are from 1.2 to 1.5%, and in cigarette smokers from 3 to 4%. For heavy smokers, they may reach 10% [WHO (1987)].
8. Among healthy people, a decreased oxygen carrying capacity (hypoxia) is apparent starting at 5% COHb. Some studies indicate small but significant effects down to COHb levels of 3.3 to 4.3%. No adverse effects have been reported below 2% COHb [WHO (1987)].

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[25] These are areas for further research.

9. Data indicating the CO emissions, either from BRM and MIM Pb life-cycle processes, or from the lead industry in general, have not been obtained. Therefore, it has not been possible to predict the potential health effects attributable to CO from these processes.

7.1.6. Other Significant Emissions

It needs to be noted, that potential human health effects may also be attributable to periodic fugitive and point source emissions of miscellaneous other inorganic and organic compounds. For most of these compounds, the releases are so small that their effects are likely to be negligible for the general population, even local to the processing facilities concerned. Nevertheless, some, such as evaporative emissions from solvents (used e.g for cleaning equipment), may be of concern in terms of the occupational exposures caused by their use.

However, one group of emissions is worthy of special note. These are the stack and fugitive releases of polycyclic aromatic hydrocarbons (PAH), released as a result of the incomplete combustion of various organic materials, such as fuel oil. The emissions include substances such as benzo [*a*] pyrene, anthracine, and various types of dioxins. The latter occur when the organic compounds being combusted contain chlorine. Many types of PAH's are carcinogenic, including benzo [*a*] pyrene, anthracine, and some types of dioxins (notably 2,3,7,8 tetrachlorodibenzo-p-dioxin (TCDD)). Whilst PAH's have been the subject of investigations by industry, scientists and regulatory agencies during the recent period, data concerning their actual emission quantities, the actual chemical and physical nature of the released materials, and their actual potential human health effects is incomplete. Hence, it is not possible to assess the potential human health effects, attributable to these emissions, either from the lead industry in general or from the BRM and MIM Pb life-cycle processes in particular.[26]

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[26] This is an area for further research.

7.2 Effects from Emissions to Water and to Land

7.2.1 Introduction

Emissions to the atmosphere may be inhaled directly by humans. The scale of the direct effects from such emissions may be global and regional, as well as local. However, the scale of the direct effects of emissions to land tend to be strictly local, whilst those of water emissions tends to be local (and sometimes regional) rather global.

Even though aquatic emissions resulting from industrial activities are discharged into adjacent water bodies (rivers, lakes or seas), human beings in industrialised countries do not drink such water directly or use it directly for the irrigation of human food crops. Also, whilst direct emissions occur to land, significantly contaminated land is not used to grow such crops. This is because water and land, for such uses, are required to be of sufficient quality such that they do not give rise to significant health effects, when either the water is drunk or used for human food crop irrigation, or the land is used to grow human food crops. To achieve this, potable water suppliers try to use water sources which are relatively free from contamination and the use of non-potable irrigation water, tends to be strictly controlled. Most commonly, obtaining water suitable for drinking, is achieved by abstracting from sources where it is filtered naturally through uncontaminated soil and rock strata. This serves to remove the majority of contaminants. In such circumstances, water suppliers only have to conduct a few simple procedures to ensure the water is suitable for distribution to the customer. However, where the availability of suitable water sources makes it impossible to contain suitable uncontaminated water, suppliers are forced to conduct their own purification processes, prior to distributing it to their customers. Also, since the primary effects of individual discharges to water tend to be more localised than for atmospheric emissions, it is possible, in many cases, to site water treatment works so they abstract from water courses upstream of industrial facilities. This avoids, therefore, the direct effects from such discharges. Similarly, in many countries, land designated for the landfill of industrial waste is kept segregated from agricultural land, and measures are taken to ensure the concentration of trace contaminants in soils used for human food crops (and also in the crops themselves) do not exceed levels at which significant health effects might be experienced in the general population.

Hence, whilst emissions to land and water could give rise to direct human health effects, in reality, this does not happen in industrialised countries. This is the case for BRM and MIM Pb life-cycle processes, since those associated directly with the lead industry are restricted to the UK and Australia (apart from the sea shipping of Pb bullion and mixed concentrate between Australia and the UK). Both the UK and Australia are industrialised countries. In some parts of the developing world, however, the general population close to an industrial activity may have to grow human food crops in land, and/or drink and irrigate with water which has been contaminated by industrial activity. In such circumstances, therefore, lead industry emissions to water could give rise to direct effects in humans.

It also needs to be noted, that much of the contamination of the land and water bodies is indirect, rather than as a result of direct controlled disposal and discharge and/or direct uncontrolled surface water run-off from facilities. This is because, contaminated materials deposited in and on the land can become leached into adjacent water bodies. In addition, contaminants in the atmosphere gradually become precipitated out, either directly into water bodies or onto land and may then become leached into them. Indirect contamination also can take place, as a result of the leaching of contaminants from vessels holding or lining water ready to be drunk or used in human food crop irrigation. For example, in plumbosolvent areas, where pipes supplying drinking water are made of Pb, some of it may dissolve in the water, thereby contaminating it.

Since the lead industry produces refined Pb and Pb alloys from both primary (i.e. virgin) and from secondary (i.e. recycled) sources, emissions to the land and water from both need to be considered in this review. Primary refined Pb and Pb alloy production involves a sequence of processes, some of which contribute significantly to emissions to land and water, as indicated in Table 7.10.

Process	Operations involved	Emissions to land	Water emissions
Mining	• Ore extraction from the ground • Ore crushing and screening	• Waste rock • Disturbed ground • Landfill of miscellaneous wastes*	• Mine water* • Wash-out from waste rock
Concentration	• Grinding (i.e. comminution) of ore • Flotation to produce concentrate	• Landfill of miscellaneous wastes*	• Tailings stream
Smelting	• Pyro- or hydrometallurgical smelting to produce bullion	• Slags* • Landfill of miscellaneous wastes*	• Process effluents*
Refining	• Purification of the bullion to produce final refined Pb and Pb alloys	• Slags* • Landfill of miscellaneous wastes*	• Process effluents*

Note: * Depending on the technologies and techniques applied, these may be insignificant in practice.

Table 7.10: Emissions to land and water associated with primary refined Pb and Pb alloy production processes

Mining and ore crushing operations break up the terrain and hence increase the surface area of material exposed to rainfall. In many parts of the world (such as the tropics), this rain tends to fall as intense storms of relatively short duration, which give rise to the risk of flooding. Large tonnages of ore material and waste material (mullock) also have to be mined. Whilst the former are transported to concentration facilities, the latter have to be disposed of. As a result, some "wash-out" (run off) is considered inevitable [Barbour (1994)].

Fortunately, low concentrations only of the desired metals (Zn, Pb, Ag, Cd etc.) are present in the mullock, since they are toxic to varying degrees. However, because the desired materials are present mostly as sulphides (the Pb as galena (PbS), the Zn as sphalerite (ZnS) etc.) and are often associated with significant amounts pyrite (FeS_2), which is also a sulphide, acids may be generated by the oxidation of these sulphides (S^{2-}) to sulphates (SO_4^{2-}). In the case of FeS_2, acid is generated by the following reactions:

$$2Fe^{4+}S^{2-}{}_2 + 2H_2O + 7O_2 \rightarrow 2Fe^{2+}SO_4^{2-} + 2H^+{}_2SO_4^{2-}$$

$$4Fe^{2+}SO_4^{2-} + 2H^+{}_2SO_4^{2-} + O_2 \rightarrow 2Fe^{4+}{}_2(SO_4^{2-})_3 + 2H_2O$$

$$Fe^{3+}{}_2(SO_4^{2-})^3 + 6H_2O \rightarrow 2Fe^{3+}(OH^-)_3 + 3H^+{}_2SO_4^{2-}$$ [Barbour (1994)]

Hence, the oxidation of the sulphide ores in solution, results not only in the production of SO_4^{2-} anions but also of H^+ cations (which lower the pH). Site run off and mine water also tend to contain significant quantities of sulphide (S^{2-}) anions, and/or sulphate (SO_4^{2-}) anions plus H^+ cations, for similar reasons. In addition, many mine workings are below the level of the natural aquifers, and so have to be pumped out continually.

Concentration (which involves milling and flotation) gives rise to waste water. This is despite the fact that substantial recycling is usually applied. Associated with this water are substantial quantities of tailings (waste particles which are mostly non-ore containing). Quantities of the various organic and inorganic reagents used for flotation, may also be present in the waste water. The classes of such reagents are summarised in Table 7.11.

Class	**Use**	**Compound**
1. Collectors	To selectively coat particles with a water-repellent surface, which is attractive to air bubbles	Water-soluble polar hydrocarbons, such as fatty acids, xanthates
2. Modifiers:		
(a) pH regulators	To change pH to promote flotation; either acidic or basic	NaOH, CaO, Na_2CO_3, H_2SO_4, H_2SO_3
(b) Activators and depressants	To selectively modify the flotation response of minerals present in combination	Metallic ions, lime, sodium silicate, starch, tannin, phosphates, sodium cyanide
3. Frothers	To act as a flotation medium	Pine oil, propylene glycol, aliphatic alcohols, cresylic acids
Oils	To modify froth and act as collectors	Kerosene, fuels oils, coal-tar oils

[Source: Modified from [Barbour (1994)]]

Table 7.11: Classes of reagents used in concentration (with examples)

Those materials with a high oxygen demand, however, (such as CN^- ions resulting from the use of sodium cyanide, hydrocarbons, oils etc.) are usually removed, by oxidation, prior to final disposal). Even so, the efficiency of this removal process varies according to the technologies and techniques used. In some cases, they may not take place at all. For example, some of the surface waters in the Rhodopes Region of Bulgaria have been shown to be polluted with significant quantities of CN-, As and S^{2-} and that this results from the lack of plant for conditioning the waste water from non-ferrous mining and concentration operations [Panayotova (1997)].

The tailings stream (which is normally a slurry) is pumped either to a specially dammed for area for settlement, or is discharged into adjacent surface or underground water bodies. Such actions have the potential to contaminate drinking water and water supplies for human food crop irrigation. Indeed, the use of tailings dams gives rise to the potential for disastrous consequences to humans and organisms in adjacent areas, should the dam fail as a result of seismic activity for example.

Primary concentrate smelting, and refining of the Pb which is produced, give rise to water emissions of various metals and metalloids. The species deemed significant are identified in Table 6.1 to be: Pb, Zn, As, Cu, Ni, Ag, Cd, Hb, Cr. Secondary Pb smelting and refining operations, also give rise some of these emissions. However, since modern lead-acid batteries contain sulphuric acid electrolyte, significant SO_4^{2-} water emissions are generally also produced.

In addition to emissions to water, various wastes are consigned to landfill at each step in both primary and secondary Pb production. These vary from specialised slags (produced by some of the pyometallurgical processes) to general rubbish. Contained within these emissions, there are a number of substances deemed to be of significant concern for the lead industry. Figure 6.1 in chapter 6 indicates that, for metals and metalloids, these are: Pb, Zn, Sb, As, Sn, Cd and Ag. They are of concern because, under appropriate conditions, they may become dissolved (and hence form ions) in water seeping through the ground, where they have been placed. As a result, both the seeping water and the aquifer into which the seeping water drains can become contaminated. Once in the ionic form, they may also be taken up by human food crop plants. Other substances such as sulphides (S^{2-}) and sulphates (SO_4^{2-}) may also be deposited on and into the land in significant quantities, and give rise to similar problems.

In the following review, the potential human health effects from the above outlined emissions of concern, are grouped into those associated with:

- anions (i.e. SO_4^{2-}, S^{2-}, and CN^-),
- cations (i.e. H^+, $Pb^{2+ \text{ and } 4+}$, Zn^{2+}, $As^{3+ \text{ and } 5+}$, $Cu^{+ \text{ and } 2+}$, $Ni^{2+ \text{ and } 3+}$, Ag^+, Cd^{2+}, $Hg^{+ \text{ and } 2+}$, and $Cr^{2+ \text{ and } 3+}$ $Sn^{2+ \text{ and } 4+}$, and Sb^{2+}), and
- organic and inorganic suspended matter (i.e. tailings, oils, and miscellaneous other substances).

The review considers the dosimetric, toxicological and epidemiological evidence for effects from ambient exposures resulting from:

- drinking water contaminated by such substances,
- eating crops irrigated with water contaminated by such substances, and
- eating crops grown in soils contaminated by such substances.

Where the toxicological effects of individual substances have already been reviewed in section 7.1, these are cross referenced. For these substances, the review focuses purely on the evidence pertaining to the potential contributions to total exposure from these sources. As with atmospheric emissions, however, the review has had to concentrate mainly on upon primary effects, as this is the highest order in the cause-effect chain at which the responses to such exposures may be predicted.

The overall purpose of the review is to identify the salient assertions currently being made in the fields, and to identify the uncertainty underlying them. It also serves to provide a basis for the BRM and MIM Pb life-cycle case studies (chapters 9 and 10).

7.2.2. Anions

Of the anions of concern from the lead industry:

- emissions of S^{2-} and SO_4^{2-} from primary production operations are, principally, as a result of ore mining and ore concentration, and secondary production emissions are from scrap lead-acid battery scrap smelting,
- whilst CN- ions are emitted from primary production concentration operations only.

MIM Pb life-cycle concentration operations are conducted at Mount Isa, in Queensland, Australia and at McArthur River, in the Northern Territory, Australia. CN^- emissions from these operations, are not thought likely to give rise to any significant human health effects at either site. This is because, the drinking water for the Mt. Isa mine site and town, is collected from a source (principally Lake Moondarra) which is distant from the mine site and tailings run off area. Also, it is tested on a regular basis for the presence of CN^- anions [Buchanan (1998). Similarly, drinking water used at the McArthur River mine and camp, is abstracted from bore holes well away from the tailings dam, and its quality is tested on a regular basis [McArthur River Mining (1998)]. However, non-ferrous mining operations in some countries, may give rise to concentrations of CN^- anions in surface waters, which could, should the water be drunk or used for the irrigation of human food crops, cause human health effects. For example, concentrations of 0.4 mg/dm^3 have been measured in the Arda River, in the Rhodopes region of Bulgaria, about 2 km downstream from an ore-dressing tailings pond facility [Panayotova (1997)]. Since this is in excess of the US EPA maximum contaminant level for drinking water of 0.2 mg/dm^3 [ATSDR (1995)], it can be concluded that this water might give rise to significant health effects, if drunk untreated.

CN^- acts as a potent and rapidly acting chemical asphyxiant, by preventing tissue utilisation of oxygen. It causes this by inhibiting the tissue respiratory enzyme cytochrome oxidase [OSHA (2000)]. The lowest observed level at which chronic effects from exposure in humans have been detected is at intakes of about 30 mg/kg of body weight/day. For a human with an average weight of 70 kg, this is equivalent to 2.1 g/day. Chronic exposures at and above this level, may result in one of more of the following: breathing difficulties, heart pains, vomiting, blood changes, headaches, and enlargement of the thyroid glands [ATSDR (1995)]. At higher levels of exposure, CN^- can give rise to effects such as weakness of fingers and toes, difficulty in walking, dimness of vision, deafness and decreased thyroid function. However, it is not known whether CN- can cause birth defects, and there are no reports that it can cause cancer in people or animals [ATSDR (1995)]. Acute symptoms of CN^- exposure, include weakness, headache, confusion, vertigo, anxiety, dyspnoea, and sometimes nausea and vomiting. Coma and convulsions may also occur in some cases. If a large enough dose of CN^- is absorbed, instantaneous collapse, unconsciousness (with convulsions) and almost immediate death occurs [OSHA (2000)].

It was noted earlier that, S^{2-} anions present in waste mine rock, tailings and mine water are oxidised to SO_4^{2-}, and that their generation is accompanied by the production of H+ cations, which can result in a lowering of the pH. The activity of the bacterium *Thiobacillus ferroxidans* is thought to be a dominant factor in this process, and hence also in the development of such acidity [Barbour (1994)]. Since the solubility of many metals and metalloids vary appreciably with changes in pH, the fall in pH may give rise to significant leaching of such substances, thereby increasing both the hazard potential of the waters and the potential for the uptake of the ions by food crop plants.

In the case of the Mt. Isa concentrator tailings stream, the pH is normally on the alkaline side of neutral, due to the presence of significant quantities of dolomite and limestone rock [Buchanan (1998)]. The McArthur River concentrator tailings stream was also shown in test work to remain at or near a neutral pH [McArthur River Mining (1998)]. Hence, in both cases, the solubility of heavy metals is likely to remain low. The measures discussed earlier which protect the drinking water at both sites from contamination by CN- at both sites, also afford it protection from S^{2-} and SO_4^{2-} anions.

7.2.3. Cations

In addition to H^+, which lowers the pH, the cations of concern are principally those produced from Pb, Zn, As, Cu, Ni, Ag, Cd, Hg, Cr, Sn and Sb.[27] The main routes of entry

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[27] Emissions to the atmosphere of Pb, Zn, As, Cu, Cd, Sn, and Sb are considered to be significant concerns for the lead industry. Hence, the dosimetric, toxicological and epidemiological evidence, for chronic and acute human health effects associated with ambient atmospheric exposures to them have been reviewed in section 7.1.

of the latter substances into the body are via inhalation and the ingestion of food and drink combined. The effects of such combined exposures are reviewed in section 7.2.4.

Lead (Pb):

The average intake of Pb food and drink consumed by an average person in the UK, in the period immediately prior 1982, was estimated to be 94 μg/day approximately in rural, non-plumbosolvent areas, with no Pb pipes or fittings, and 140 μg/day approximately in urban, plumbosolvent areas, with Pb pipes and fittings. In the former case, the daily intakes from drinking water and from food (including the contribution from cooking water) were estimated to be 5.5 and 94 μg/day in the former case, and 30 and 110 μg/day in the latter case [MAFF (1982)]. However, MAFF (1989), note there has been a drop in Pb concentrations in the UK, in most foodstuffs, but especially in canned fruit and vegetable products since 1982. The latter has been attributed to the reduction in the use of Pb in can solders. However, MAFF (1982a) noted that crops grown in areas of high Pb soil content may contain higher concentrations of Pb than crops in general. This is assumed to still be the case. MAFF (1989) also note that direct exposures to Pb in most food groups in Great Britain (excluding tap water), over the period 1982 to 1987, ranged between 0.02 to 0.06 mg/day. This well below the FAO/WHO Provisional Tolerable Weekly Intake for dietary Pb which is 3 mg/week for an adult.

Pb concentrations in drinking water originate primarily from the phenomenon of plumbosolvency. Hence, whilst the water leaving UK treatment works usually has a concentration of less than 10 μg/dm^3, in plumbosolvent areas, where Pb pipes, storage tanks and other Pb-based fixtures are in use, occasional breaches occur of the current UK Pb standard drinking water (which is 50 μg/dm^3 at consumers taps) [EPAQS (1998)]. In the US, concentrations have been shown to range from 10 to 30 μg/dm^3, on average, but can rise to 350 μg/dm^3 in localised areas where Pb pipes, fittings or solders are used [OECD (1991)]. In highly plumbosolvent areas in Ayr, Scotland, concentrations were found to range from 10 to 1000 μg/m^3, with a pH ranging from 4.4 to 5.5 [Davies and Thornton (1989)].

OECD (1991) note that establishing the relationship between exposure to food alone and the blood Pb level is complicated by the fact, it is sometimes difficult to determine the source of the exposure. However, slopes of 0.16 for blood Pb levels to dietary intakes have been identified for children exposed to low ambient levels, whilst slopes of 0.03 to 0.06 have been associated with intake levels exceeding 300 μg/day. For adults, however, whilst slopes of 0.02 to 0.06 have been identified, experimental studies are considered to be more accurate, generally, for estimating blood Pb levels than using such coefficients [OECD (1991)]. The relationship between drinking water Pb concentration and blood Pb concentration at low doses, is also believed to be linear in adults, and with a slope of about 0.06. In children and infants, slopes have been shown in studies to be 0.12 and 0.26 respectively with drinking water below 15 μg/dm^3, and 0.06 and 0.04 respectively above 15 μg/m^3 [OECD (1991)]. However, most studies have been based on exposures to relatively high water Pb levels (between 50 to 2000 ug/dm^3) and the results have then extrapolated for low doses [OECD (1991)].

In theory, such coefficients could be used to estimate the direct potential effects from Pb emissions to land and water, attriibutable to the lead industry in general, and to BRM and MIM Pb life-cycle processes in particular. However, this is not possible, in practice, without modelling to identify that fraction of the Pb in individual drinking water supplies and foods, which is attributable to direct emissions from these sources, rather than from indirect sources (i.e. from secondary, tertiary and higher order sources).[28] This fraction will tend to vary according to food and water type, and origin.

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[28] This represents an area for further research.

Zinc (Zn) and Copper (Cu):

Food is the most significant source of exposure for Zn and Cu in the UK [MAFF (1981)] and globally [Robertson (1994)]. The Joint FAO/WHO Expert Committee on Food Additives reports that daily Zn intakes range from 4.8 to 47.6 mg/day [MAFF (1981)]. In the UK, the daily intake of Zn from food is estimated to be about 10.5 mg, which is within the range considered adequate by the WHO Expert Committee on Trace Elements in Human Nutrition [MAFF (1981)]. Daily Cu intakes of under 1.8 mg/day are reported in MAFF (1981) to be typical in the UK. This is less than the 2 mg/day recommended as necessary for adequate nutrition by the WHO Expert Committee on Trace Elements in Human Nutrition [MAFF (1981)]! The main sources of Zn and Cu in the diet, are meat, cereals and milk [MAFF (1981)].

Water is an important, though lesser source of exposure, to Zn [Robertson (1994)]. Average concentrations of Zn in tap water, have been shown to range from 5.6 to 8.6 μg/dm^3 in different surveys in the UK, and to be 0.11 mg/dm^3 in the US [MAFF (1981)]. Whilst the US data is an order of magnitude higher than the UK, it would still account for less than 10% of the total intake from food. With regard to Cu, surveys in the UK and US, range between 0.014 and 0.36 mg/dm^3 [MAFF (1981)].

Hence, Zn concentrations in food and water are well with the range for normal nutrition, and Cu concentrations (at least in the UK) err on the side of dietary deficiency rather than overload. In addition, Cu toxicosis rarely, if ever, occurs in humans, and excess Zn is excreted efficiently via the gastrointestinal tract [Robertson (1994)]. Therefore, the contributions of BRM and MIM Pb life-cycle processes, and the lead industry in general, to Zn and Cu contamination of waters, would be unlikely to give rise to any significant detrimental health effects in the general population, should such waters be drunk or used for irrigation untreated. Similarly, contamination of foods by such Zn and Cu emissions is unlikely to give rise to any significant detrimental human health effects in the general population.

Arsenic (As) and Cadmium (Cd):

The Total Diet Studies, conducted during the period 1974 - 78, estimated the daily intake of As in the average UK diet to be less than 100 μg/day, whilst in the US, during the period 1973 - 74, it was estimated to be around 100 μg/day [MAFF (1982b)]. Also, WHO (1987) have estimated that a representative intake of As is 40 μg/day, in foods of terrestrial origin, and 80 μg/day in sea foods. Intakes from air and water add very little to the figures [MAFF (1982b)].

The most significant dietary source for As is, therefore, fish and shellfish, and communities, which have a high proportion of fish in their diet, are likely to be subject to considerably increased intakes of As. However, MAFF (1982b) note, that even in such cases, the FAO/WHO maximum acceptable daily load of As, which is 50 μg/kg of body weight/day (equivalent to 3500 μg/day for a 70 kg adult), would not be breached. Therefore, the contributions of the lead industry, and of BRM and MIM Pb life-cycle processes, to non-cancer human health effects, resulting from direct contamination of food and drink, by their As emissions to land and water, are likely to be negligible for the general population.

For non-smokers, various types of food represent the major source of exposure to Cd. This is despite the fact that intestinal absorption is only about 5% [Robertson (1994), WHO (1987)], though absorptions up to 20% have been found in some women with severe iron deficiencies [WHO (1987)]. The daily intake of Cd in food has also been well documented [WHO (1987)]. In European countries and North America, it is 10 to 30 μg/day. However, there may be large variations depending on age and dietary habits.

Drinking water normally contains only very low concentrations of Cd (generally ranging between 0.1 and 2.0 μg/dm^3), though on rare occasions, levels up to 10 μg/dm^3 have been detected [WHO (1987)]. Hence, it is generally not a major contributor to Cd exposure in humans.

In industrialised countries, Cd in food comes principally from the deposition of Cd-containing atmospheric particles onto soil, and from the application of sewage sludge to agricultural land [WHO (1987)]. Hence, Cd discharges by the lead industry, direct to water and land, are not considered to be major contributors to the Cd in food. As a result, the non-cancer human health effects associated with the Cd, in their direct land and water discharges, are expected to be negligible for the general population.

With regard to the risk of cancer induction following exposure to Cd and As, since there is no known threshold for either substance, the subsequent ingestion of a fraction of their As and Cd emissions to land and water, may contribute to the background cancer rate in the general population. However, without detailed modelling, to identify the fraction of Cd and As in drinking water and foods directly attributable to BRM and MIM Pb life-cycle discharges, their contribution to this background rate cannot be established.[29]

Nickel (Ni)

Ni has been shown to be an essential micro-nutrient for some mammalian species, and it has been suggested a person of 70 kg body weight would have a daily requirement of 50 fg/day. However, at higher levels of exposure, Ni and its compounds show varying degrees of toxicity. Nickel carbonyl is the most acutely toxic Ni compound, and insoluble forms, such as Ni powder, are the least toxic [ATSDR (1993)]. The main target organs for Ni, and Ni-containing compounds, are the lungs and kidneys [US EPA (1985)].

The symptoms of acute nickel carbonyl exposure in humans include headache, vertigo, nausea, vomiting, insomnia, and irritability. These are often followed by chest pains, dry coughing, cyanosis, and gastrointestinal symptoms. Indeed, US EPA (1994) classify nickel carbonyl as a high concern pollutant, based on it severe acute toxicity.

The US EPA (1998a) have established a Reference Dose (RfD) for humans, for chronic exposure to the soluble salts of Ni, of 20 μg/kg of body weight/day, based on decreased body and organ weight in rats. For a 70 kg average weight adult, this is equivalent to 1400 μg/day. In addition to decreased organ weight, neonatal mortality and dermatoxicity have been associated with low level chronic oral Ni exposure in humans [US EPA (1998a)].

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[29] This represents an area for further research.

Nickel carbonyl, and Ni in refinery dust, have been classified by the US EPA as carcinogens and probable carcinogens, by inhalation, respectively [US EPA (1998)]. Presumably, oral exposures would also give rise to similar effects.

Food is the main source of dietary Ni, and to a lesser extent water. However, the latter contributes to less than 5% of the total Ni intake in the general population [MAFF (1985)]. The typical dietary intake of Ni ranges from 100 ro 300 μg/day. This is above the suggested minimum for nutrition and below the RfD, by about an order of magnitude. Hence, the non-cancer human health effects associated with the Ni, in direct land and water discharges by BRM and MIM Pb life-cycle processes, are expected to be negligible for the general population. However, in common with As and Cd, Ni itself and many Ni-containing compounds are known carcinogens. Therefore, as with As and Cd, such Ni discharges may, contribute to the background cancer incidence rate in the general population.

Chromium (Cr)

The Total Diet studies, conducted during the period 197-78, estimated the mean daily intake of Cr in the average UK diet to be 16 μg/day (upper bound) or 111 μg/day (lower bound), and in typical western diets in general, the Cr intake is estimated to be between 50 and 100 μg/day [MAFF (1985)]. Whilst no particular food groups appear to dominate, the highest concentrations have tended to be found in the meat, fish, fruit and sugar groups of foods [MAFF 91985)].

Cr ions exist in two forms Cr^{3+} (trivalent) and Cr^{6+} (hexavalent). Cr^{3+} ions are thought unlikely to occur in drinking water, due to the low solubility of hydrated chromic (II) oxide (Cr_2O_3) at pH values >5 [MAFF 91985)]. Hence, Cr^{6+} is the ion normally present. Cr concentrations in the public supplies in some European Union (EU) cities are reported to vary between 1.0 to 5.0 μg/dm^3 (with a mean of 2.0 μg/dm^3) [MAFF 91985)]. Assuming a mean daily intake of 1.5 dm^3/day, the mean daily intake would be 3 μg/day.

Cr metal appears to be biologically inert [Robertson (1994)]. However, neither the levels of exposures nor its potential harmful effects have been well characterised [ATSDR (1989)]. Cr^{3+} ions are required for health because they potentiate the action of insulin, and are thought to act as a moderator in lipid metabolism [Robertson (1994)]. The minimum daily requirements for optimum health are not known, however, a daily ingestion of 0.7 to 3 μg/day per kg of body weight are considered adequate [ATSDR (1989)]. For an adult of 70 kg body weight, this is equivalent to between 49 and 210 μg/day. Normal exposures to Cr^{3+} ions are not considered to be harmful, however, very large doses may be [ATSDR (1989)]. Cr^{6+} ions are believed to be 10 to 100 times more toxic than Cr^{3+} ions when exposure is by the oral route [Robertson (1994)]. Cr in food is mostly in the trivalent (Cr^{3+}) form.

The human health effects, associated with long-term low level (i.e. chronic) exposure, are damage to the liver, kidneys, circulatory system and nerve tissues, and dermatitis [US EPA (1998d)]. Acute human health effects are irritation of the skin and ulceration, either of the gastrointestinal tract [US EPA (1998)] or of the nasal mucosa [ATSDR (1989)], depending on the source(s) of exposure. Cr^{6+} is carcinogenic by inhalation. However, the oral carcinogenicity cannot be ascertained, as there appears to be no available data [US EPA (1998d)].

The main source of exposure to Cr, in the general population, is via food, and to a lesser extent via water. Exposure via inhalation is usually relatively minor. Absorption through direct skin contact may also be significant occupationally.

The oral RfD for Cr^{3+}, in soluble salts (which include chromic (II) oxide (Cr_3O_3) and chromium (III) sulphate ($Cr_2(SO_4)_3$), is 1.5 mg/kg of body weight/day [US EPA (1998d)], and for Cr^{6+} is 0.003 mg/kg of body weight/day. For an adult with a weight of 70 kg, this is equivalent to 105 mg/day. If it is assumed the Cr in food and water intakes is all Cr^{6+} (i.e. the 'worst case scenario'), then typical oral exposures from drinking water and food combined, would be 139 μg/day (when calculated using the EU mean drinking water intake and the UK Total Diet Studies 1978 - 78 upper bound mean), and would be 114 μg/day (when calculated using the EU mean drinking water intake and the UK Total Diet Studies 1978 - 78 lower bound mean). These values are well within the the oral RfD values for Cr^{6+}. Therefore, it may be concluded that, the non-cancer human health effects associated with the Cr, in direct land and water discharges by BRM and MIM Pb life-cycle processes, are expected to be negligible for the general population. However, in common with As, Cd and Ni. Cr^{6+} is a known carcinogen. Therefore, as with As, Cd and Ni, such Cr discharges may, contribute to the background cancer incidence rate in the general population.

Mercury (Hg)

The general population may be exposed to Hg through inhalation of the air, consumption of food and water, and through dermal exposure [US EPA (1997)]. Hg exists in three different states: elemental-Hg, inorganic-Hg and organic-Hg, and the major routes of exposure differ for each type [ATSDR (1992)]. Thus, for elemental-Hg, the major route is through occupational inhalation, and in the general population through the amalgam in dental fillings. The general public are not exposed to inorganic-Hg, to any significant extent these days, because these substances have been banned [ATSDR (1992)]. However, exposure to the general public by organic-Hg (in the form of methyl-Hg) occurs primarily through diet, with fish and fish products being the principal source. The chronic and acute human health effects of Hg vary, according to which of the three types are involved. However, since these days, exposures to organic-Hg, only, are considered significant for the general public, its effects, only, are examined below.

Acute exposure to organic-Hg results in effects on the central nervous system (CNS) (which include blindness, deafness, impaired consciousness and death) [WHO (1990)]. The primary effect from chronic exposure, is damage to the CNS. The oral RfD for methyl-Hg is 0.0003 mg/kg of body weight/day, based on the CNS effects in humans [US EPA (1998e)].

The Hg released by the lead industry to water and the land, is principally in the elemental form. This is converted by methylation into methyl-Hg in the environment. Therefore, the consumption of methyl-Hg, following its bio-accumulation in fish, is a secondary and not a primary effect. It was noted at the start if this chapter (section 7.1) that, as one moves along the cause-effect chain, the uncertainty, with regard to trying to predict and estimate effects, increases. Thus, whilst it may be possible, generally, to predict potential primary effects, this is not usually possible for higher order effects, without additional modelling of the structure and behaviour of the receiving environment for the emissions. The author has, therefore, been unable to ascertain, whether any significant human health effects actually are attributable to organic-Hg in foods, resulting from elemental-Hg emissions to land and water, either from the lead industry in general, or from BRM and/or MIM Pb life-cycle processes in particular.

Tin (Sn)

The Total Diet Studies, conducted during the period 1976 - 81, estimated the weighted mean daily intake of Sn in the average UK diet to be 3.86 mg/day (upper bound) or 3.75 mg/day (lower bound)[30], whilst in the US, values ranging from 1.5 to 17 mg/day have been reported [MAFF (1985)]. With regard to the sources of Sn in the diet, whilst it occurs naturally in some foods, the concentrations are normally less than 1 mg/kg of food. Higher concentrations have been found in cereals. However, the highest concentrations are in canned foods, with the Sn ions having entered the food through the dissolution of the Sn coating on the Sn plate inside the cans [MAFF (1985)]. Nevertheless, the Total Diet Studies indicated, that during the period from 1976 - 81, there was a regular year-by-year fall in the Sn intake in the UK diet. MAFF (1985) suggests this may have been caused by an increase in the proportion of cans which were lacquered.

MAFF (1985) also note that concentration of 6 $\mu g/dm^3$ has been reported in municipal drinking water in the USA. Assuming a mean intake of 1.5 dm^3/day, the daily intake of Sn from water is 9 μg/day. Hence, the intake from water is three orders of magnitude less than the intake from food.

Since the WHO provisional maximum tolerable daily intake is 0.14 g/day (i.e. 140 mg/day), for an adult with an average weight of 70 kg, the daily Sn consumption from water and from food, in the general population, is well within the tolerable limit. Therefore, the human health effects, resulting from the direct contamination of food and drink, by Sn emissions to land and water, from both the lead industry in general, and from BRM and MIM Pb life-cycle processes in particular, are expected to be negligible.

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[30] The 'upper bound' value has been calculated on the assumption that individual values less than the limit of detection of the measurement technique are equal to that value, whilst the 'lower bound' value assumes that individual values less than the limit of detection are zero.

The Total Diet Study, conducted in 1976, estimated the daily intake of Sb in the average UK diet from food to be 29 μg/day (upper bound) and 2 μg/day (lower bound). However, most foods contained Sb at concentrations which were less than the limit of detection of the test technique. In the USA, estimates from monthly tests, conducted between January and August 1967, estimated intakes from food to be between 247 and 1275 μg/day, whilst in New Zealand and Germany, average daily mean intakes have been estimated to be less than 12 μg/day and 23 μg/day respectively [MAFF (1985)].

MAFF (1985) also noted there were very few data available for Sb concentrations in drinking water. However, the 1980 European Economic Community (EEC) Directive 80/778/EEC has set a concentration limit of 10 μg/dm^3. Assuming a mean consumption of 1.5 dm^3/day, the intake of Sb would be 15 μg/day.

Since intakes from food and water are of the same order of magnitude, it may be concluded that both are important sources of exposure to Sb in the general population. Using the upper bound value from the 1976 UK Total diet Study data, and the 1980 EEC Directive concentration limit, the total exposure to Sb via the oral route of entry (i.e. from food and water alone) would be 44 μg/day. As noted in section 7.1.2, this is slightly above the oral RfD set by the US EPA, which is 28 μg/day. Hence, emissions may contribute, possibly, to some chronic effects. However, there are uncertainties associated not only with the estimates of the intakes of Sb from food and water, but also with the oral RfD itself. Hence, further data are needed to ascertain whether any significant human health effects are actually attributable to Sb exposure from food and drink in the general population. In consequence, it is uncertain whether any significant direct human health effects may be associated with Sb emissions to water and land, either from the lead industry in general, or from BRM and/or MIM Pb life-cycle processes in particular.[31]

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[31] This is an area for further research.

Silver (Ag)

Ag has been treated as a significant emission, principally because the annual emissions to water of Ag, within the consented trade effluent discharge from the BRM battery breaking plant at Northfleet, Kent, UK, have to be reported the UK environmental regulator, the Environment Agency (UK EA), as part of BRM's Authorisation to operate the plant under the regime of Integrated Pollution Control (IPC). This discharge is into the River Thames, which is immediately adjacent to the plant.

Inorganic Ag is relatively non-toxic and the estimated emissions to water are thought to be less than 0.03 kg/year. Therefore, they are considered to be of minor significance only, as they are unlikely to give rise to any significant direct human health effects in the general population.

7.2.4 Organic and Inorganic Suspended and Solid Matter

Direct discharges to water include undissolved particulate matter as well as dissolved ions. Solids deposited in the ground, in some instances, will also contain ions in solution. However, in most cases, they are either in a non-ionic, or a non-aqueous from. For substances to be absorbed into the human body, however, they need, generally, to be in solution. This may occur in the environment. For example, in a landfill site, where non-aqueous solid matter can become dissolved in water percolating through it. On the other hand, it may happen within the gastrointestinal tract, in the moist surfaces of the lung alveoli, or on other exchange surfaces, such as the skin.

The discussion in sections 7.2.2 and 7.2.3, addressed the situation where the human body is exposed directly to aqueous pollutants. This section considers exposure to solid matter in food and suspended particulate matter in water, which on contact with the body exchange surfaces, will have to be converted into aqueous forms, if they are to pass across them.

With regard to the sequence of processes involved in the production of refined Pb and Pb alloys, from their cradle, as raw materials, through to the manufacture of the final product(s); undissolved particulate emissions to water and solid wastes to land, arise from all of the processes, as indicated in Table 7.6. Emissions are both fugitive and controlled. They also arise, not only from the raw materials being processed, but also, to some extent, from all of the ancillary materials.

It is not known, whether any of these emissions ever give rise to significant direct human health effects. However, it was noted in section 7.2.1, that in industrialised parts of the world (and in some other parts too), efforts are made to ensure that soils, used for growing human crops, and water supplies, used for drinking water and for human food crop irrigation, are free from such contamination. Therefore, since the operations conducted by BRM and by MIM for its Pb life-cycle processing steps are all within industrialised countries, the direct effects of such emissions are likely to be negligible. Emissions to land and water may, however, in some instances, give rise to significant indirect effects. For example, in Northampton, Western Australia, tailings from Pb ore processing operations, have been used extensively in the town for various purposes, such as for the foundations of houses, covering driveways, streets etc., and it has been suggested this might be responsible for the increased blood Pb levels, detected in children in the town compared with those resident outside of it. [Heyworth et al. (1981)].

Identification of such indirect effects, would require site-specific modelling. Whilst such modelling and assessments have not been included in this thesis (due to time constraints), they are required. Indeed, they are already conducted, where significant effects are suspected. However, if they are to aid the lead industry, in meeting the environmental challenge, in manner which is optimally effective, they need to be integrated fully within an approach, such as is outlined in chapters 1 to 4 of this thesis. This question is considered further in the Overall Conclusions and Recommendations (chapter 11).

7.3. Effects from Combined Exposures

7.3.1. Introduction

So far, the substances deemed likely to be most significant by decision makers have been examined separately. However, under ambient conditions, a mixture (cocktail) of pollutants is the usual form of exposure. Exposures to individuals substances also tend to be combined, in the sense that they are often via more than one route of entry into the body. Therefore, the question arises of whether ambient exposure to mixtures via either one route of entry, and/or via a combination of routes, gives rise to health effects which are any different to the effects of exposure in isolation. The possibilities are that multiple exposures might give rise to additive effects, synergistic effects or antagonistic effects. Also, such interactions (if any) may occur between some or all of the pollutants deemed significant and/or with other agents such as allergens.

The question of exposures by individual types of substances by more than one route of entry, is applicable mainly for metals and metalloids, and has already been considered in sections 7.2.2.and 7.2.3 of this chapter. Hence, the focus of this section is on the possible combinatory effects between different pollutants and/or other agents in the environment.

7.3.2. Via Inhalation

The Advisory Group on the Medical Aspects of Air Pollution Episodes [MAAPE (1995)] has examined this question in their report entitled 'Health Effects of Exposures to Mixtures of Air Pollutants'. MAAPE (1995) note, there is only a limited amount of information on the toxic effects of complex atmospheric mixtures. However, some controlled animal experiments have been conducted. In these studies, some synergistic effects in concurrent administrations have been detected, for example; in rats between O_3 and ammonium sulphate ($(NH_4)_2SO_4$) [Warren et al. (1986)], between O_3 or NO_2 and $(NH_4)_2SO_4$) [Last et al. (1983), Last et al. (1984)], between O_3 and sulphuric acid (H_2SO_4) [Warren and Last (1986)] and between O_3 or NO_2 and acidic aerosols (though it appears that it is the acidity and not the aerosol which gives rise to the synergism) [Last et al. (1986)]. However, MAAPE (1995) consider that risk assessment extrapolation from such investigations to humans is not possible due to the fact that the findings are from simple mixtures (when ambient exposures are to complex mixtures) and the administered doses tend to be relatively high compared to ambient exposures.

Apparently, laboratory studies on humans exposed to atmospheric mixtures, at concentrations equivalent to ambient, have found no clear evidence of synergism. However, MAAPE (1995) suggest that interactions may have been underestimated because of the small size of the studies, their limited duration and the confounding effects of prior exposure to ambient pollutants. Controlled chamber studies on humans also appear to have failed to show clear evidence of interactions. In addition, MAAPE (1995) note that epidemiological studies of atmospheric mixtures are poorly developed and show an incomplete pattern. Thus, evidence for interactions between pollutants in atmospheric mixtures is, at present, equivocal.

With regard to synergism when atmospheric pollutants are in combination with agents such as aeroallergens, MAAPE (1995) note that a limited number of studies have shown that, when human subjects are exposed to relatively high concentrations of air pollutants, an enhanced allergenic response may be observed. However, whether this occurs with ambient concentrations of pollutants is unknown. Therefore, this too is, at present, equivocal.

Thus, in conclusion, on the basis of current evidence, it has not been possible to demonstrate any significant interactions between atmospheric pollutants under ambient conditions. However, this does not mean, necessarily, that they do not occur, only that they have not been demonstrated.

7.3.3. Via Ingestion

Animal study evidence concerning the influence of dietary composition on the toxicity of Cd, Cu, Zn and Pb, has been reviewed by Mills et al. (1980)]. They noted, that Cu intoxication in sheep has been shown to be decreased, by increasing the amount of Zn in the diet, and that Zn achieves this antagonistic effect by restricting the rate at which Cu is accumulated in the liver. Molybdenum (Mo) has also been shown to be a Cu antagonist, by inhibiting Cu uptake. Where diets are high in Mo, it has been associated with Cu deficiencies. However, this action appears to require the presence of sulphur (S) in the diet. This is because, S and Mo appear to work synergistically in their effects. Indeed, when diets are low in S, elevated Mo appears to have little effect on Cu uptake [Mills et al (1980)].

In various animals, chronic exposures to excess Zn or Cd in the diet, have been shown to lead to a marked decline in the Cu content in the blood plasma and liver, and to the development of a poorly mineralised skeleton [Mills et al (1980)] (i.e they are antagonists).

With regard to Pb, experiments using laboratory animals have demonstrated that gastrointestinal (GI) absorption is increased, not only when food intake is reduced, but also when the dietary intakes of protein, calcium (Ca), phosphorous (P), sulphur (S), magnesium (Mg), and iron (Fe) are low [Mills et al. (1980)]. These effects are antagonistic. Synergistic effects regarding GI absorption of Pb have also been demonstrated, when diets are high in fat, phopholipids and lactose (i.e. it is increased) [Mills et al. (1980)].

Animal studies have also shown that: retention of ingested Pb in the gut increases when the Ca, P and potassium (K) contents of diets are decreased, reduced sulphate (SO_4^{3-}), Ca, and phosphate (PO_4^{3-}) has been associated with increased mortality in high Pb containing diets in sheep, increased Ca and PO_4^{3-} markedly depresses Pb retention in bone and increased S depresses Pb retention in the kidneys and heart.

Mushak (1991) has reviewed the research concerning the biophysico-chemical aspects associated with the GI absorption of Pb in children and adults. He notes, there are many known interactions between Pb and other substances in the GI tract, and that the evidence from these effects is derived largely from animal studies (such as those summarised above). However, he points out that the two nutrients, which feature most prominently in the human diet, in modifying GI absorption of Pb, are Ca and Fe. For these two substances, inverse relationships with Pb (i.e. antagonisms), similar to those identified in the review of Mills et al (1980), have been identified. He also points out that PO_4^{3-} and vitamin D are also significant, but that they have not yet been fully characterised epidemiologically.

Vitamin C has also been shown to exert an antagonistic effect on blood Pb levels in humans, and it may be as effective as the Pb chelating agent ethylene diamine tetra acetic acid (EDTA) for the clinical treatment of severe Pb food poisoning [Saryan (1999)].

Finally, it is worth noting also, that the uptake of Pb (and of other metals and metalloids too) from the GI tract, is influenced, additionally, by biophysico-chemical factors such as particle size, the solubility of the particles in the gut, and the reactivity of the ingested Pb-containing chemical matrix with reference to *in vivo* mobilisation [Mushak (1991)].

Thus, antagonistic and synergistic effects have been demonstrated in animals for Pb and Cu, with various metals and anions. Some effects have also been shown to occur in humans. However, whether any of these interactions (or others which are as yet unknown) significantly potentiate or reduce any of the human health effects, attributable to the emissions from the lead industry in general, or BRM and MIM Pb life-cycle processing operations in particular, is uncertain.

8. Potential Environmental Effects of the Lead Industry

8.1. Introduction

As with the previous chapter, dealing with the potential human health effects of the emissions from BRM and MIM lead life-cycle processes, this chapter focuses on their potential environmental effects first in isolation (in sections 8.2 to 8.4). Then, in section 8.5, the potential environmental effects of the emissions when they occur in combination with each other and with other ambient pollutants is considered.

Whilst the focus is on the emissions from BRM and MIM lead life-cycle processes, the emission categories and their effects are generally applicable for the whole of the lead industry. Due to the complexity of environmental processes, in the absence of comprehensive modelling of the fate of emissions in the various environmental media, it is not possible to estimate their actual effects. Their potential to cause such effects, however, can be assessed and such assessments are provided in the case studies in chapters 9 and 10, which consider BRM processes and MIM Pb life-cycle processes respectively. Rather, this chapter serves to identify the types of environmental effects associated with the emissions of concern from the foreground processes of the BRM, MIM Pb life-cycle operations and the lead industry. The chapter (in conjunction with chapter 7) also provides a theoretical basis for the human health and environmental categories, which are used in the case studies, and assessed and discussed in the Overall Conclusions and Recommendation (chapter 11).

The environmental effects, discussed below, have been modified from those introduced in Table 5.1, as follows:

- the categories of 'resource depletion' and 'disturbances' have been amalgamated into a single section (section 8.2),
- the category of 'pollution' has been subdivided into two sections dealing with atmospheric pollution, and aquatic pollution plus land contamination (sections 8.3 and 8.4), and
- within these subdivisions, the types of pollution are discussed for each of the main groups of substances deemed to be of concern.

8.2. Resource Depletion and Disturbances

In common with all extractive industrial operations, the production of refined lead and lead alloys requires the consumption of various environmental resources. These occur not only with regard to the main raw materials (i.e. the zinc-lead-silver ores considered in the MIM case study), but also with other ancillary materials (considered in both the BRM and MIM case studies).

The development and operation of industrial facilities also causes various disturbances both locally and further afield. Local disturbances include:

- land take (which in the case of mining operations may be quite significant),
- land occupation (e.g. by processing facilities),
- visual intrusion (through the siting of facilities), and
- noise and smell generation.

Such primary disturbances, may destroy local plant and/or animal communities. Alternatively, they may alter their structure, reversibly or irreversibly, Some disturbances (such as noise and smells) and their effects could, on occasion, extend further afield. They may also give rise to secondary and higher level effects in a manner similar to that discussed for emissions in chapter 5 section 5.1.

8.3. Pollution

Many of the acute and chronic human health effects detailed in the previous chapter, would be expected to be experienced by other terrestrial vertebrates. However, the ability of exposures via inhalation, ingestion and/or via other routes, to exert a response will vary according to differences in:

- the structure or shape of their respiratory tract, gastrointestinal (GI) tract and/or other exchange surfaces (such their epidermis),
- the physiology of their respiratory tract (such as tidal volume and breathing rate), their GI tract (such as the peristaltic rate) and/or other exchange surfaces,
- the physiology of their bodies (which may differ significantly from humans), and
- the body mass of the organism itself (which may make it more or less tolerant to pollutants).

In addition, the size of the dose, which different organisms receive, will vary according to their living habits and where they live.

The other environmental effects of the pollutants in question are considered below.

8.3.1. Lead, Other Metals and Metalloids

In section 7.1.2 of the previous chapter, it was noted that Pb, plus all of the other significant metal and metalloid emissions from BRM and MIM Pb life-cycle processes, and by the lead industry in general, are toxic to humans. They are also toxic to organisms living in the receiving environment. According to the terminology in Table 5.1, they contribute, therefore, to the category of Ecotoxicity.

The extent of this toxicity varies according to the types of target organisms concerned. Broadly speaking, these may be divided into:

- microbes (bacteria, unicellular algae, microfungi etc.) and macrofungi,
- non-vascular plants (mosses, liverworts, complex algae and lichens etc.),
- vascular plants (non-flowering and flowering),
- invertebrates (arthropods, crustaceans, echinoderms, molluscs etc.), and
- vertebrates (fish, amphibians, reptiles, birds, mammals).

The effects of Pb, plus the other metals and metalloids of concern, upon these different targets organisms, are summarised briefly below. Since they have been reviewed previously by the author in Robertson (1994), it should be referred to, should more detailed information be required.

It also needs to be noted, that variations in the sensitivity to exposures within populations also exist, due to the presence of phenotypic and genotypic variations in individuals. These differences have also been reviewed by the author in Robertson (1994).

1. Effects on microbes (bacteria, unicellular algae, microfungi etc.) and macrofungi

Due to the fact that Pb, plus the other metals and metalloids, are toxic to microbes, a number of physiological effects may be detected as a result of their exposure. These are reductions in:

- the rate of litter decomposition,
- the rate of soil respiration,
- nitrogen mineralisation and nitrification,
- urease, phosphatase and amylase activity,
- microbial biomass, and
- the numbers of colony forming hydrolytic bacteria and fungal length [Robertson (1994)].

Alterations to the community structure may also occur [Robertson (1994)].

These parameters have also been used to measure and monitor the extent of contamination of the environment by these substances.

2. Effects on non-vascular plants (mosses, liverworts, complex algae, lichens etc.)

Large ranges in tolerances to heavy metals have been shown to exist. Bryophytes (i.e. the mosses and liverworts), are extremely capable of accumulating heavy metals from the atmosphere, and some types are also extremely efficient at taking up metal ions from soil and water substrates and accumulating them [Robertson (1994)].

In lichens, the toxic effects of heavy metal contamination under field conditions have been recorded as a sudden or gradual decrease in species numbers and/or diversity along a known gradient of heavy metal exposure [Robertson (1994)].

3. Effects on vascular plants (non-flowering and flowering)

Metals such as Ag, Cd, Cr, Cu, Hg, Mn, Mo, Ni, Pb, Sb, and Zn tend to be strongly adsorbed by soil constituents (especially smectite, illite and vermiculite clays), organic matter and hydrous oxides of Fe and Mn [Robertson (1994)]. In general, metals (except Mo) are more mobile and, in consequence, are more available in conditions which are acidic, reducing and/or where the organic matter content is less than about 2% [Robertson (1994)]. It has also been shown that the average stability of metal-soil complexes to changes in pH tends to decrease with electronegativity, in the order: Pb > Ni > Co > Zn > Cd > Mn [Boekold et al. (1993)]. Hence, soil pH will have a significant influence of the availability of metals to plants.

In addition, it has been shown that at equimolar concentrations, the relative toxicity of metals to vascular plants (as well as for bryophytes and lichens) decreases in the order: Cd > Cu > Pb ≥ Zn. However, the availability of micro-nutrients and other environmental conditions (such as soil pH and the presence of various soil constituents noted above) may modify, greatly, the concentrations needed to bring about such reactions [Tyler et al. (1989)].

a) Effects of Pb:

The bio-availability of Pb in soils, is usually relatively low due to the stability of the metal-soil complexes. In addition, even where it is taken up by plants, translocation is usually limited due to the tendency of Pb to bind to root surfaces and cell walls [Robertson (1994)].

The visible symptoms of Pb toxicity are smaller leaves and stunted growth. Leaves may also become chlorotic and reddish, and the roots turn black [Robertson (1994)]. The physiological effects of Pb on plants include, depression of photosynthesis and respiration. Whilst at a biochemical level, significant effects include, inhibition of chlorophyll biosynthesis and of several enzymes involved in photosysnthesis (especially phosphoenolpyruvate carboxylase (PEPC) in C4 plants such as *Zea mays*), nitrogen metabolism alterations, and chromosomal abnormality induction [Robertson (1994)].

b) *Effects of Cd*

Cd is chemically similar to Pb, and is adsorbed by soil constituents. However, unlike Pb, the Cd metal-soil complexes are highly sensitive to pH changes. Indeed, Boekold et el. (1993) have shown that a reduction of 0.5 of a pH unit gives rise to a halving of the amount of Cd adsorption.

Cd is much more toxic to plants than Pb, and this may be related to its high affinity for the sulphydral groups of several plant compounds involved in plant metabolism. The symptoms of Cd toxicity are stunted growth, and smaller, curled and chlorotic leaves whose margins show a red/brown coloration [Robertson (1994)]. The physiological effects of Cd on plants, are principally on photosynthesis and respiration, and these are as a result of the inhibition of CO_2 fixation within chloroplasts. Whilst at a biochemical level, it is believed that Cd acts by replacing other metals, such as Zn, in metal activated enzymes [Robertson (1994)].

c) *Effects of Cu*

Cu is a micro-nutrient, which is involved in carbohydrate, nitrogen and cell wall metabolism, seed production, disease resistance and water relations. However, at higher concentrations, it can be extremely toxic [Robertson (1994)]. The growth response effects to Cu in toxic concentrations, are stunted growth, small chlorotic leaves and early fall. Concentrations of 0.05 to 0.10 mg/dm^3 are considered to be toxic to non-tolerant plants [Tyler et al. (1989)].

Knowledge of the physiological effects of Cu toxicity on plants has been derived almost entirely from *in vitro* studies. These have shown the principal effect, of Cu intoxication to be, the inhibition of photosynthesis and respiration [Robertson (1994)].

e) *Effects of Zn*

Like Cu, Zn is an essential plant micro-nutrient. Zn is involved in carbohydrate, nucleic acid and lipid metabolism. The growth response effects to Zn, in toxic concentrations, are generally similar to those for Cu [Robertson (1994)]. However, of all of the heavy metals, Zn is considered to be the least toxic to vascular plants [Robertson (1994)].

The biochemical effects of Zn toxicity are upon the activity of several enzyme systems (such as those involved in photosynthetic electron transport and phosphorylation) [Robertson (1994)]. Hence, one of the physiological effects of Zn intoxication, is the inhibition of photosynthesis. Other effects include, interference with carbohydrate metabolism and the restriction of translocation in the phloem [Robertson (194)].

f) Effects of the other significant metals and metalloids

All heavy metals are toxic, to varying degrees, to plants. These effects (including those from Pb, Cd, Cu and Zn) are exerted principally upon various enzyme systems. The first visible sign of damage, from experimental studies, is almost always impeded root development (reduced growth, elongation etc.) [Tyler et al. (1989)]. At higher concentrations, physical effects which are similar, generally, to those described for Pb, Cd, Cu and Zn, are likely to occur.

Acidic soils, such as those which occur in conifer woodlands, provide conditions where heavy metal mobilisation, and hence their availability to plants may be higher than with other soil types. However, Tyler et al. (1989) note, that as an average, vascular plants growing in such soils appear to be less sensitive to heavy-metal pollution than most of the other main groups of soil organisms.

4. Effects on invertebrates and vertebrates

Soil invertebrates are affected by heavy metal contamination, through surface contamination of their bodies, from ingestion of contaminated food, and by food source deprivation (which is itself a product of heavy metal contamination) [Tyler et al. (1989)]. Also, in both invertebrates and vertebrates, some heavy metals may concentrate selectively in certain tissues. For example, they tend to concentrate selectively in the cerebral ganglia, seminal vesicles, and in certain parts of the gut of earthworms [Robertson (1994)]. In addition, earthworms and other soft-bodied invertebrates, and certain arthropod groups, such as oribatid mites, appear to be more sensitive to heavy metal burdens than are other groups [Robertson (1994)].

Neither the trophic level, nor the body weight of organisms, appears to determine the body concentrations of Zn, Cd or of Cu in terrestrial invertebrates. Rather, the body burden of individual metals, is dependent upon the organism's ability to regulate each of the metals [Robertson (1994)]. Thus, for example, evidence points quite strongly to the fact that most terrestrial organisms can regulate Zn quite efficiently, whilst body levels of Cd appear to be only poorly controlled. However, some soil vertebrates do appear capable of controlling their tissue Cd levels quite efficiently [Robertson (1994)]. There is also some evidence to suggest that Cu regulation occurs, though whether Pb regulation also occurs, is considered more controversial [Laskowski and Maryankski (1993)].

An enormous amount of experimental work into the cytological, physiological and biochemical effects of heavy metals on animals has been conducted, aimed at revealing, through extrapolation, their effects on human health. Since the effects of heavy metals and metalloids on human and other animals are broadly similar, the discussion of the human health effects of these substances (provided in chapter 7), is also considered to be broadly applicable for other vertebrates and for many invertebrates in the environment [Robertson (1994)].

8.3.2. Particulates

Airborne suspended atmospheric particulates, in the diameter range of 0.1 to 2.0μm, have the potential to absorb and scatter light, thereby leading to the impairment of visibility [QUARG (1993)]. Particulates with a diameter of less than 0.1μm do not contribute to such impairment. However, as was noted in section 7.1.1 of chapter 7, such particles are in the nucleation range. Therefore, they will tend to grow by coagulation and may eventually reach the size when they too become visibility impairing. In addition to visibility impairment, light scattering and adsorption may also cause the sun to become entirely obscured when it is low in the sky and the pathway through the atmosphere is long (i.e. at sunrise and sunset). Substances which have been demonstrated to contribute to these effects are ammonium sulphate and nitrate with respect to light scattering and particulates containing carbon and iron oxide with respect to light absorption [QUARG (1993)].

Another visible potential effect of airborne atmospheric particulates is fog formation. This can occur with water soluble particulates because they tend to deliquesce and consequently increase in size under certain environmental conditions. They may then act as nuclei upon which water vapour can condense thereby giving rise to fog.

Particulates also tend to deposit, by the processes of wet and dry deposition on materials and structures such as clothes, buildings, cars, windows, window sills, paint work etc. and thereby create nuisance dusts. Such depositions make materials and structures appear dirty. Also, where the particulates are acidic, they are known to contribute to the corrosion of metals and the deterioration of stone and paint on buildings.

According to the scheme illustrated in Table 5.1, all of these effects may be classified under the category of Landscape Degradation. In addition, the deposition of acidic particulates may also be classified as contributing to the category of Acidification.

8.3.3. Sulphur Dioxide and Related Compounds

Sulphur dioxide (SO_2) is the principle cause of the environmental effect of Acidification [McKinney and Schoch (1996)]. Acidification of the environment is caused primarily by acid rain, which is thought to arise principally from the contamination of atmospheric water by the acid H_2SO_4. (As was noted in section 4.2.3.2, this is formed from the oxidation of SO_2.) Depending on the concentration, this may cause the pH to be lowered appreciably. The effects of such acid rain are primarily upon bodies of water such as lakes and streams which rest on top of soil with a limited capacity to neutralise acidic compounds (i.e. they have limited buffering capacity). SO_2 may also contribute to episodic acidification (e.g. from rapid snow melts and heavy downpours). Changes in the pH of such bodies from both acid rain and episodic acidification cause significant losses to aquatic life, especially to fishes [ICI (1996)]. Acid rain may also damage plants, sensitive building materials (e.g. limestone) and soils directly. (Thus, it can also contribute to the effects of Physical Ecosystem Degradation and Landscape Degradation.) In addition, SO_2 contributes to the environmental effects of Photochemical Oxidant Formation.

Sulphates (SO_4^{2-}), as particulates, are emitted directly to the atmosphere by a number of the lead industry processes. For example, they are emitted from the BRM stack, at Northfleet, in Kent, UK, the MIM Pb smelter stack at Mt. Isa, in Queensland, Australia, and the BZL Zn/Pb smelter stack at Avonmouth, UK. Also, SO_4^{2-} ions in water, are emitted to water directly. For example, they are emitted by BRM, as a consented discharge from the lead-acid battery breaking and processing plant at Northfleet. Aqueous SO_4^{2-} are also formed , when atmospheric sulphates dissolve in water bodies following dry or wet precipitation. Aqueous SO_4^{2-} ions contribute not only to the category of Acidification, but also to Eutrophication.

8.3.4. Nitrogen Oxides and Related Compounds

As with SO_2, nitrogen oxides (NO_x) are oxidised in the atmosphere to form an acid, HNO_3, which is a significant contributor to the environmental effect of Acidification. NO_x also has a nutrifying effect (i.e. it contributes to the effect of Eutrophication). In water bodies, this can lead to increased growth by micro-organisms which, in turn, increases the biochemical oxygen demand (B.O.D.). Since the amount of dissolved oxygen in water bodies is normally limited, it results in a consequent decrease in the dissolved oxygen concentration. These changes may, in turn, cause large scale fish kills.

NO released by aeroplanes may contribute to depletion of the stratospheric O_3 layer (i.e. contribute to the category of Ozone Depletion), whilst ground level emissions (such as those from BRM, MIM Pb life-cycle and other lead industry processes) contribute significantly to Photochemical Oxidant Formation. NO_x makes a significant contribution to the latter, since atmospheric NO_2 reduction is one of the main routes for ground level O_3 production and O_3 is the most abundant photochemical oxidant in such photochemical smogs.

In addition, O_3 is known to contribute directly to the category of Global Warming (i.e. enhanced radiative forcing of the atmosphere). Since, O_3 production is dependent upon atmospheric NO_2 concentrations, global warming is, therefore, indirectly dependant upon the NO_x concentrations. O_3 is also a significant contributor to the category of Landscape Degradation, since it is thought to be responsible for about 90% of all air pollution damage to crops. For example, it may be responsible for a total crop loss of 6-7% of United States agricultural production [McKinney and Schoch (1996)].

8.3.5. Carbon Dioxide and Carbon Monoxide

Atmospheric carbon dioxide (CO_2) concentrations are often the main limiting factor for photosynthesis under daylight conditions. Therefore, increased ambient concentrations of CO_2 might, conceivably, give rise to increased rates of photosynthesis and hence increased biomass production.

With regard to the health effects of CO on vertebrates, they are expected to be similar to those discussed for humans in chapter 7. In addition, both carbon monoxide (CO) and CO_2 contribute to the category of Global warming, with CO having a potential which is about 3 times that of CO_2 [IPCC (1990)]. CO contributes to the category to Photochemical Oxidant Creation.

8.3.6. Other Significant Emissions

It needs to be noted that environmental effects may also be attributable to periodic fugitive and point source emissions of miscellaneous other organic and inorganic compounds to the air, water and land. The author has obtained insufficient data to predict such effects, either for the lead industry in general, or for BRM and MIM Pb life-cycle processes in particular. However, for the latter, all of the foreground activities are confined to Australia and the UK (apart from the ship transport of intermediate materials between the two countries). In both countries, the operations concerned are closely regulated to ensure the direct environmental effects are within limits deemed currently to be acceptable within these countries.

8.3.7. Combined effects

Both NO_2 and O_3 have been demonstrated to affect tree growth [QUARG (1993)]. Also, several studies have indicated that urban air quality (which will contain quantities of particulates, NO_2 and O_3) negatively influences the growth of herbaceous plants. However, apparently, it has been difficult to single out any particular pollutant as being the cause of such effects in the mixture [QUARG (1993)].

NO_x, SO_2 and acidic particulates have been assumed, in currently accepted assessment methodologies (such LCA), to contribute additively to the environmental effect of Acidification. However, synergistic or even antagonistic effects might occur.

The synergistic antagonistic interactions between various metals, discussed in section 7.3.3 of chapter 3 for human health effects, are also considered likely to occur, to varying extents, within exposed animals in the environment. For plants, additive and synergistic interactions between Pb and Cd have been reported by Tyler et al. (1989). However, the interactions between metals in the soil and in plants appear mostly to be antagonistic [Robertson (1994)]. These effects are all to the category of Ecotoxicity.

However, as with the contributions of NO_x, SO_2 and acidic particulates to Acidification, currently accepted methodologies (such as LCA) assume the contributions of different metals to Ecotoxicity to be additive. The implications of this assumption are discussed further in the Overall Conclusions and Recommendations (chapter 11).

9. BRM Case Study

9.1. Purpose and Frame of Reference

This study provides quantitative inventories and potential human health and environmental effect assessments, plus additional qualitative analyses, attributed to:

1. the primary and secondary Pb refining processes at BRM combined (i.e. at the *overall site scale*), and
2. the various throughputs through the BRM site (i.e. at the *individual throughputs through the site scale*).

Hence, this case study addresses the 'subnational (local): BRM Limited' scale identified in Figure 4.1 of chapter 4, as well as other smaller scales.

Demonstrations are provided of how

- the assessments can be used to compare the relative inventories and potential human health and environmental effects of different throughputs through the site,
- the assessments may be applied practically for environmental performance indicator setting at different decision scales, and
- such information aids decision makers, both within BRM and those charged with regulating it, in identifying 'best' environmental practice.

The uncertainties underlying all of the above assessments are also examined. To help guide decision making deliberations based on these and other similar assessments, consideration is given, for each assessment, of how such uncertainties should influence the decision making of all relevant parties. These considerations resolve around two suppositions, which are that:

- decisions have to be made even though those making the decision may be faced with considerable uncertainty, and
- decision makers would want to make the 'best' decision(s) possible (leading to the adoption of the 'best' environmental practice), given available evidence.

The overall aim of this case study is to provide BRM with a series of inventories, assessments and approaches, to address existing and likely future environmental challenges in the most effective manner. The approaches developed herein, are intended to act as site scale and individual throughput through the site scale elements within a new overarching strategy developed by the author to help MIM and its subsidiary companies, to meet these challenges.[32] Therefore, whilst the assessments and approaches may be applied solely at these two scales, they in are fact posited within this strategy. This differs from existing strategies for dealing with environmental concerns in being fundamentally holistic and in providing a more multi-dimensional way of thinking about and dealing with environmental matters. Hence, for them to demonstrate their full capabilities, they should not be applied in isolation of the wider picture.

The approaches developed in this study are also intended to be applicable generically for the rest of the lead industry and for other industry sectors facing comparable challenges. In addition, the intention is for the assessments and approaches developed in this case study to become adopted as part of the normal environmental management practices within BRM and MIM. They are, therefore, viewed as providing a starting point to be progressively articulated and improved upon with time.

Note that wherever possible, the data were collected by the author from site personnel. Where other sources of data have been used, this is identified explicitly. In both instances, however, the reliability of the data and their influence on the usefulness of conclusions based upon such data are assessed. Note also, that although this research has been sponsored by BRM, the arguments and conclusions, which have been drawn, are those of the author. They may not, therefore, necessarily coincide with those of the company.

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[32] Preliminary work to develop an emissions source scale assessment approach, which would be applicable for BRM, has also been conducted by the author. Due to time restrictions in the preparation of this thesis, however, it has not been possible to include this as a case study. The need for approaches at this scale is discussed in the Overall Conclusions and Recommendations (chapter 11).

At the end of the case study, the main elements of the arguments are summarised and specific conclusions and recommendations are made.

9.2. Nature of the Business and Site History

The current site layout is indicated in Figure 9.1.

BRM is currently the largest capacity Pb refinery in the world and Europe's largest primary Pb and silver (Ag) refinery (i.e. refiner of these metals from virgin materials). The plant occupies approximately 10 Ha. It is situated on reclaimed marshland along the south bank of the River Thames at Northfleet in Kent, England (UK National Grid Reference: TQ 614 756). Refining operations have taken place here continuously since 1931. A lead-acid battery breaking facility also operates at the site and provides the raw materials for the secondary Pb production operations.

The original plant was set up on the site of the Britannia Cement Company works to refine the Pb and Ag output from Mount Isa in Queensland, Australia. It had a capacity of about 70,000 tonnes of Pb per annum. In 1968, this was increased to 150,000 tonnes per annum. Then, in 1976 a second refinery, with a capacity of about 30,000 tonnes per annum, was built to process the Pb bullion by-product output from the Zn Imperial Smelting Furnace (ISF) at Avonmouth, near Bristol, England.[33] These older and the newer parts of the plant are known as the No. 1 and No. 2 Refinery respectively.

In 1997, the kettle floor of the No. 2 Refinery was extended to accommodate the processing of additional lead bullion from MHD in Duisberg, Germany and to incorporate decopperising operations, which formerly had been carried out at the Avonmouth Zn refinery on the Pb bullion destined for BRM. Production figures for the calendar year 1998 are: 145,260 tonnes of refined Pb and Pb alloys per annum from the No. 1 Refinery, and 49,295 tonnes of Pb and Pb alloys per annum from the No. 2 Refinery.[34]

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[33] This refinery is operated by Britannia Zinc Limited (BZL), and is part of the MIM group of companies. Its operations, and the potential human health and environmental effects associated with them, are considered in some detail in the MIM Case Study.

In 1991, a CX Battery Breaking Plant (which replaced an earlier MA41 breaking plant) and a dedicated effluent treatment plant were also commissioned. Currently, the CX Plant, in conjunction with an Isasmelt furnace, treats approximately 56,000 tonnes per annum of scrap lead-acid batteries.

In 1997, a dedicated dross handling plant was also commissioned. This treats all of the skims and drosses produced by the BRM and Isa floors.

9.3. Current Status of the Plant

There are two distinct groups of operations at the site. These are:

1) Primary operations, which take place in the No. 1 and No. 2 Refineries.
2) Secondary operations, which take place in the CX Battery Breaking Plant and the Secondary Treatment Plant.

The location of these facilities, as well as the various chimney stacks associated with these operations are indicated in Figure 9.1.

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[34] Data from BRM production records.

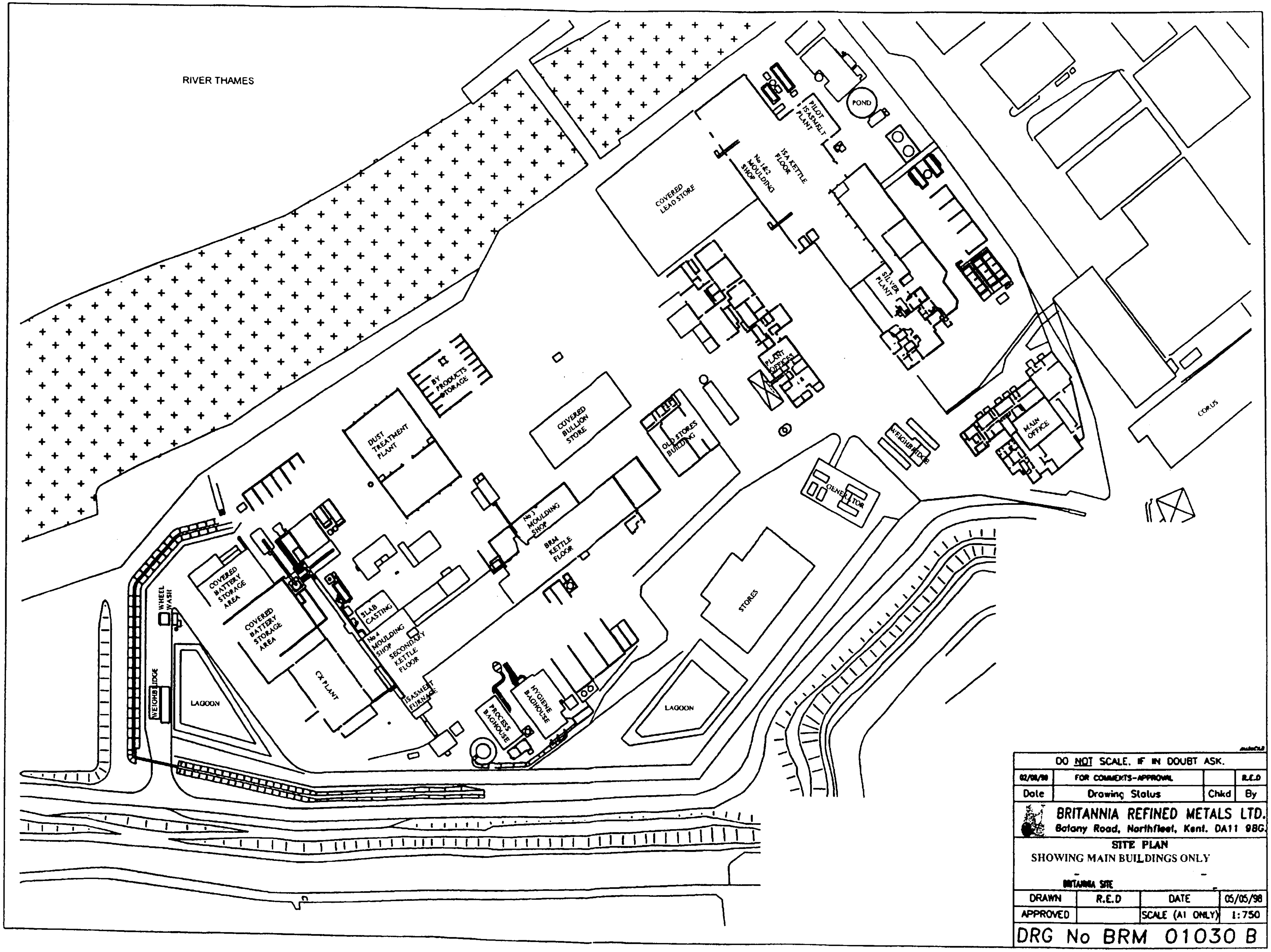


Figure 9.1 Plan of BRM site

9.3.1. Primary operations

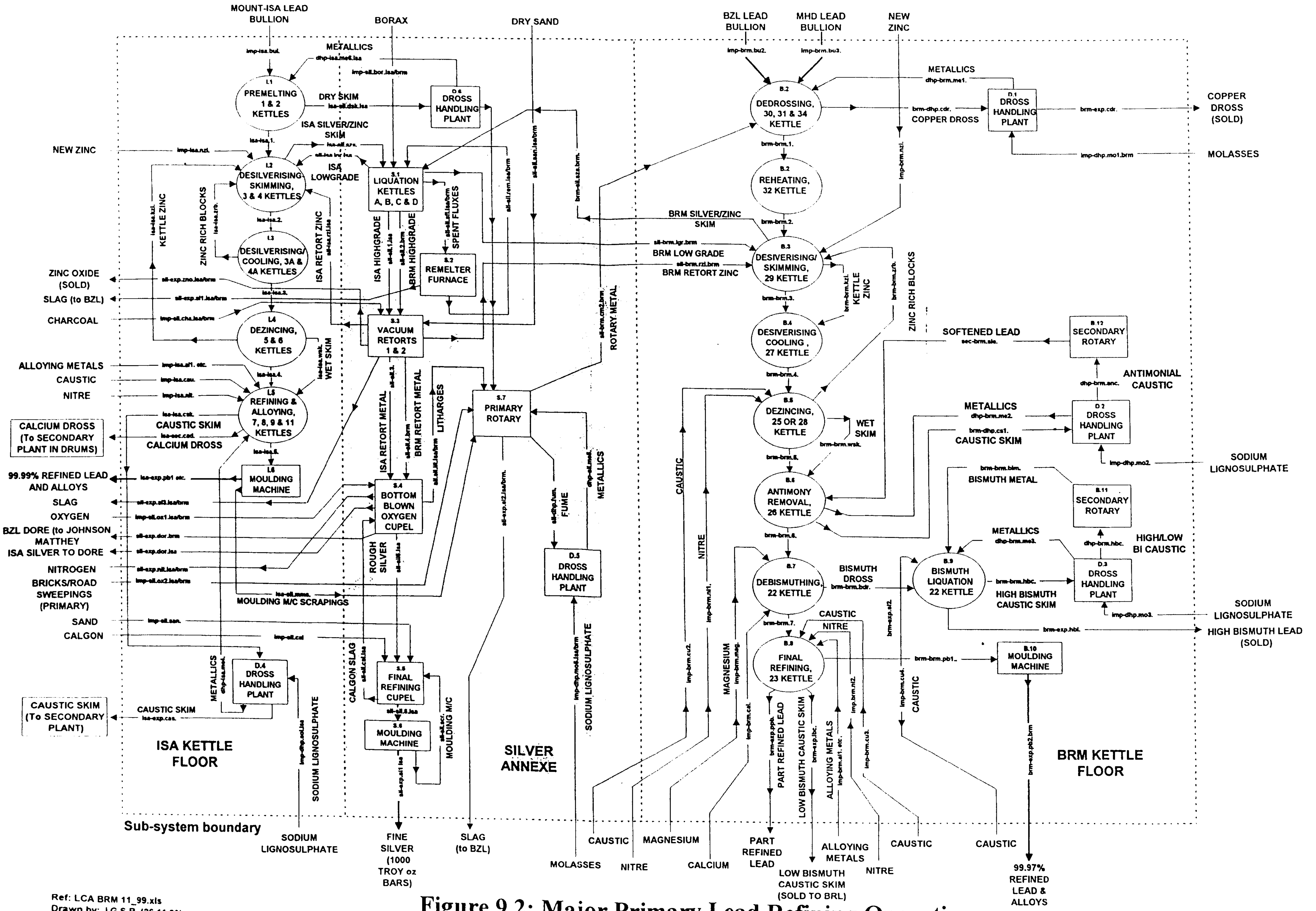
A flow sheet detailing the current set up for primary operations is provided (Figure 9.2).

The No.1 Refinery is composed of two linked parts; the Isa Kettle Floor and the Silver Annexe. Currently, the No. 1 Refinery produces refined Pb, Pb alloys and Ag; all of which are primary products. The No. 2 Refinery, on the other hand, produces both primary refined Pb and Pb alloys and secondary Pb alloys. This floor was known until 1997, as the BZL Kettle Floor, as its Pb crude bullion feedstock came entirely from the Zn ISF plant of Britannia Zinc Limited (BZL), at Avonmouth. However, it is now known as the BRM Kettle Floor, to reflect the fact that MHD Pb bullion is also now processed there. The No. 1 and No. 2 Refineries tend to operate independently of one another. However, Ag containing by-product metal is transferred from the No. 2 to the No. 1 Kettle Floor (indicated in Figure 9.2 by the flow labelled with the code: brm-isa.bpm.). Also, the throughputs from both the No. 1 and No. 2 Refineries, are processed in the Silver Annexe.

A rotary furnace is adjacent to the BRM Kettle Floor. This treats various Pb bearing and copper (Cu) bearing materials such as Cu dross, bismuth (Bi) dross, antimony (Sb)/Bi caustic skim (i.e. brm-brm.cd2., brm-brm.bdr. and brm-brm.abc. in Figure 9.2), and road sweepings, to produce various materials. These are then recycled back into the process.

The processes of refining both the Isa crude Pb bullion, and the BZL and MHD crude Pb bullions are essentially pyrometallurgical. In both refineries, refining is conducted in a series of gas heated kettles by a modified Parkes process. In essence, this involves the pumping of molten Pb, of various grades of purity, from one kettle to the next. As indicated in Figure 9.2, skimming operations are carried out in most kettles, whilst desilverising, dezincing and final alloy production takes place only in certain kettles. The desilverising and dezincing process, involves Zn being added, and subsequently removed as a skim, to extract the Ag from the molten mass. The Ag/Zn skims produced (i.e. isa-sil.szs. and brm-sil.szs.) are then sent to the Silver Annexe for further processing.

Annexe facilities are shared between the Isa and BRM floors **Flowsheet for period: 01.01.1998 - 31.12.1998 inclusive**



Ref: LCA BRM 11_99.xls
Drawn by: J.G.S.R. (26.11.99)
Modified from BRM charts: 00317 and 00322

Figure 9.2: Major Primary Lead Refining Operations

On the Isa Kettle Floor, the refining process is started by charging the Premelter kettles with crude Isa bullion blocks (kettles 1 and 2). On the BRM Kettle Floor, it is started by charging the dedrossing kettles with MHD and/or BZL Pb crude bullion blocks (kettles 30, 31 and 34). The impurities in the Isa crude bullion (which are fewer than in the MHD and BZL crude bullions), are removed in the dry skim taken from the surface of the Premelter kettles, and by the addition of caustic (NaOH) and nitre ($NaNO_3$) to the alloying and refining kettles (kettles 7, 8 9 and 11). The dry skim (i.e. isa-sil.dsk.) is then sent to the by-products area for treatment in the Silver Annexe short rotary furnace. The caustic skim (i.e. isa-exp.csk.), produced by the addition of the caustic and nitre in the alloying and refining kettles, is removed and sold on to the Britannia Recycling Limited (BRL) plant, at Wakefield, In Yorkshire, UK for further processing.[35]

The BZL and MHD crude bullions, however, contain significant quantities of Cu, Bi and Sb, which require removal. Therefore, in addition to the desilversing and dezincing processes, which are similar to those conducted on the Isa floor, further and separate Cu dedrossing, Sb removal and debisthmuthing steps are necessary. This results in the production of additional Cu dross and Bi dross (i.e. brm-dhp-cdr. and brm-dhp.bdr.), and an additional Sb caustic skim (i.e. brm-dhp.cs1.). The Bi dross is treated using caustic in a liquation kettle to produce a high Bi caustic skim and a high Bi Pb. The latter is then sold.

The various skims and drosses produced on both kettle floors, are processed by the dross handling plant. The overall scheme of operations conducted on these materials is as indicated in Figure 9.4. The functions of the plant are two fold. Firstly, it separates the large metallics from the smaller 'middlings' and fines, so they may be fed back into the processes. Secondly, calcium lignosulphonate is added to the fines, which acts as a binding agent and, as a consequence, helps reduce the problem of fugitive dust formation during material handling.

The refining of the Ag/Zn skims from the BRM and Isa floors in the Silver Annexe is conducted in separate batches but they use the same equipment. Whilst the refining processes are essentially similar, the end products differ since Ag doré (i.e. sil-exp.dor.brm.) is produced from the BRM Ag/Zn skim, whilst fine Ag (i.e. sil- exp.sil.isa.) is produced from the Isa Ag/Zn skim.

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[35] BRL is also a member of the MIM group of companies.

Ag refining in the Annexe is essentially a three stage process for the BRM floor throughput, and a four stage process for the Isa floor throughput. The former involves liquation, vacuum retorting and bottom blown oxygen cupellation only and produces an Ag doré. However, the latter also includes a final fine cupelling stage, to produce fine Ag.

A rotary furnace (the primary rotary) is also housed within the Silver Annexe.

The process of Ag refining is indicated in Figure 9.2. The first step involves the charging of a liquation kettle with Isa or BRM Ag/Zn skim (i.e. isa-sil.szs.isa or isa-sil.szs.brm), and heating it under a layer of borax (which acts as a flux) to about 715^{0}C. At this temperature, the melt separates into an Isa or BRM high grade Ag layer (i.e. sil-sil.1.isa or sil-sil.2.brm) and an Isa or BRM low grade layer (i.e. sil-sil.lgr. isa and sil-sil. lgr.brm). The high grade layer is then further treated by vacuum retorting, whilst the low grade layer is returned to the Isa or BRM floor respectively.

Vacuum retorting involves holding the Isa or BRM high grade under a vacuum, which causes the Zn to vaporise and condense in a condenser. It is then tapped, and returned to the Isa or BRM floor for re-use. The remaining metals in the retort (the Isa or BRM retort metal), now contain about 70% by mass of Ag, with some Zn and Cu impurities, and about 30% Pb. This retort metal (i.e. sil-sil.3.isa and sil-sil.4.brm) is then placed in the bottom blown oxygen cupel (BBOC) and heated to about 1000^{0}C. Oxygen (O_2) is injected in through the base, which causes the Pb and other impurities to oxidise and float to the surface. They are then decanted off as litharge.

The BRM throughput, is then moulded into blocks and sold. However, the Isa throughput is subjected to a final cupellation stage. Initially, this involves, air being blown through the melt in a cupel, to remove any final impurities, and then nitrogen (N_2), to remove the dissolved O_2, The refined Isa Ag is then moulded into 1000 Troy ounce bars, which are sold.

A small rotary furnace (the primary rotary) is used to reclaim metal from various drosses, scrapings and litharges.

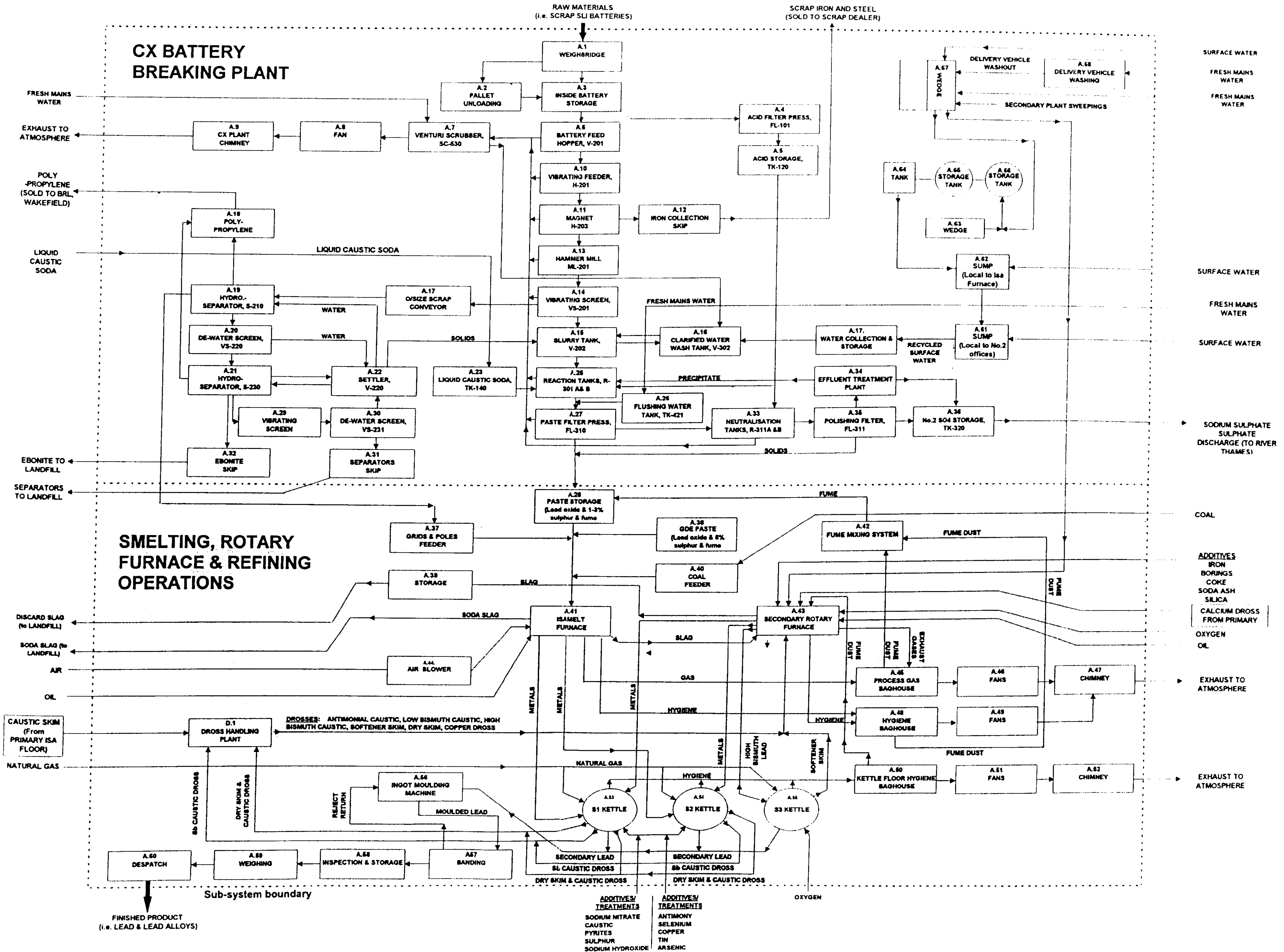
9.3.2. Secondary operations

A flow sheet detailing the current set up for primary operations is provided (Figure 9.3).

The Isasmelt furnace is at the north-western corner of the No. 2 Refinery building and two alloying kettles (S1 and S2 in Figure 9.1), plus a moulding plant, are adjacent to it. The CX Plant is close to this end of the No.2 Refinery. It crushes lead-acid batteries in a hammer mill and separates, by a hydrodynamic process, the non-Pb containing materials (polypropylene, separators, ebonite and sulphuric acid (H_2SO_4) electrolyte) from the Pb containing materials (Pb plates, grids and poles and Pb paste). The H_2SO_4 is neutralised using sodium hydroxide (NaOH) to form a sodium sulphate (Na_2SO_4) solution. The Pb paste, consisting of lead oxide (PbO) and lead sulphate ($PbSO_4$), is also partially desulphurised using NaOH, thereby creating additional Na_2SO_4 solution. Currently, after further on-site treatment, the waste Na_2SO_4 is released into the River Thames as a discharge consented by the UK Environment Agency (EA). However, this practice may cease within the foreseeable future and a useable Na_2SO_4 product be produced instead. Currently also, the ebonite and the separators from the crushed batteries are sent to a local landfill site and are classified as 'special wastes' under the Environmental Protection Act 1990. The crushed polypropylene from the battery cases, however, is transported by road to BRL, where it is further processed into polypropylene granules. These are then sold on as secondary polypropylene.

The Pb containing materials are fed into the Isasmelt furnace. After firing, the molten Pb, which is tapped from the furnace base, is added to kettles S1 and S2. If necessary, additional alloying metals are added and the skimming off of impurities from the surface takes place prior to moulding into 25kg pigs. The Pb, is then banded, stacked into piles, stored under cover in the adjacent product yard, and sold on to various customers.

Flowsheet for period: 07.07.1997 - Current



Ref: LCA BRM-99.xls (sheet2)
Drawn by J.G.S.R. (13.11.1996, updated 26.11.1999)
Modified from BRM charts O0317

Figure 9.3 Major Secondary Lead Production Operations

9.4. Environmental emissions sources, concerns and abatement procedures

Emissions to the three main environmental media (i.e. air, water and land) may occur through both controlled means as well as through uncontrolled (i.e. fugitive) escapes. Emissions to the atmosphere are mainly Pb, various other heavy metals and gases such as CO, CO_2, and SO_2. Losses of fine particulate by-products, feeds and various process materials due to handling, may also occur from time to time.

BRM has a licence to deposit the ferrosilicate slags, which are produced by the primary and secondary refinery rotary, at an on-site disposal facility. A soda slag from the Isasmelt furnace is also produced, which is landfilled at a 'special' waste disposal site. (In addition, the remelter furnace in the Silver Annexe produces a slag, which is sold to BZL for further processing.) There are no other direct emissions to land, though some fugitive emissions are inevitable. The only controlled emissions to water from the site processes are from the consented discharge from the CX Plant Effluent Treatment Plant. This contains small quantities of heavy metals similar to those emitted to the air as well as the Na_2SO_4 discharge, discussed previously. All of these discharges to water are consented by the UK EA. Waterborne emissions to land are thought to be negligible. However, some on-site emissions to land might occur through the occasional run off of contaminated water to surrounding areas during heavy storms.

Pollution abatement equipment is installed and is in operation, where it has been considered necessary. Thus, the CX Plant effluent treatment system is backed up by a filtration plant to provide a final polish to the discharge. In addition, devices that create significant quantities of metal laden dusts and fumes, such as the furnaces during normal operation and kettles when dedrossing operations are being carried out, have extraction systems connected to baghouses. The latter, filter the air and gases prior to their discharge via one of a number of stacks. Moreover, current work practices are designed to minimise emissions to air, land and water. However with the types of processes and technology involved, some controlled and fugitive emissions are inevitable.

Current practices at the plant, show that BRM adopts a responsible attitude and strives to improve its environmental performance. However awareness of the human health and environmental effects of Pb, of the various other heavy metals and of the miscellaneous other emissions to the environment, associated with the lead industry, was far less in the past. For some pollutants it was non-existent. Regular testing of the working environment and personnel within the industry did not commence, therefore, until the late 1970s. Thus records of the environmental impact of the operations at Northfleet (in the form of environmental monitoring data for the period leading up to the 1970s) are practically non-existent and even more recent records are incomplete.

However, in recognition of current knowledge of the human health and environmental effects of Pb the other substances associated with its activities and due to its desire to be a responsible organisation, BRM carries out extensive monitoring of emissions that goes beyond legal requirements. This work, carried out by BRM's own Environmental Department and analytical laboratory, is ongoing and aims to be both reactive to events that occur as well as predictive of potential problems. The author considers the sponsoring of this research project exemplifies the proactive and forward looking attitude of BRM to environmental concerns.

9.5. The process of regulation and the contributions of this case study

BRM is authorised by the UK EA to operate its primary and secondary Pb refining operations at Northfleet, Kent, UK, under the regime of Integrated Pollution Control (IPC), under Section 6 of the Environmental Protection Act (EPA, 1990). As defined in EPA (1990), and its subsidiary regulations, BRM operates a Part A Prescribed Process, under the category of "non-ferrous metals", best described by Section 2.2 of SI No. 472 Environmental Protection (Prescribed Processes and Substances) Regulations 1991 [BRM (1995)]. BRM is also licensed by the UK EA to dispose of ferrosilicate slags in on-site facility, plus rotary and Isasmelt furnace slags at a licensed landfill site. The latter are defined under the EPA (1990) as 'special wastes', and the landfill facility is one specifically for the disposal of such wastes.

The current approaches used by the UK EA in discharging its regulatory duties, are governed by its charter which requires it to, *inter alia*:

1. adopt an integrated approach which considers the impacts of substances on all environmental media and substances across all of its functions, to achieve environmental protection and to enhance the environment;
2. work with all regulated organisations, including industry, to develop approaches which deliver such protection and enhancement without imposing excessive costs on those organisations in relation to the benefits gained; and
3. develop a close and responsive relationship with its various affected parties (i.e. the 'public', local authorities, representatives of local communities and regulated organisations) (Environment Act, 1995: S4) [HMSO (1995)].

Thus, the UK EA has regard both to the costs and benefits of different options to both the regulated organisations and to the 'general public', when carrying out its pollution control responsibilities.

The regulatory approach of the UK EA is enshrined in two key principles, namely:

1. its requirement that BATNEEC be adopted for all Prescribed Processes (as defined in the EPA (1990) and
2. that processes under IPC demonstrate that the Best Practicable Environmental Option (BPEO) has been adopted.

With respect the emissions to the atmosphere, local authorities, in their regulatory role, are subject to similar requirements.

The requirement to weigh up the relative costs and benefits of options means that it tends to be pragmatic when carrying out its regulatory functions. Thus, it will often rank different options on the basis of what it considers to be the relative environmental, human health and economic costs and benefits of different options. Approaches such as BATNEEC and BPEO, help the regulators to identify which technologies and site-specific treatment options would, in their opinion, be most appropriate for achieving their goals of optimal environmental and human health protection without imposing excessive costs on the regulated organisations. Thus, BATNEEC and BPEO help to guide the regulators in this pragmatic ordering process.

The author considers the approach of weighing up the relative costs and benefits of different options and of developing a close and responsive relationship with its affected parties to be a potential strength, since regulation is more likely to be effective if it receives the support and active participation of those being regulated. Providing that the approaches (and the decisions which follow from them) are empirically verifiable, that their underlying reasoning is transparent and accepted by all affected parties and that they reflect the values, preferences and needs of these parties, then 'best environmental practice' would be also achieved. Consequently, any option(s) selected through such an approach would also represent the 'better environmental option(s)'.

The approaches of the regulators, being normative (i.e. reflecting and imposing the norms of the time) should, therefore, foster good decision making. However, it has been suggested that such approaches could, potentially at least, compromise the ability of the regulators to regulate effectively [Faversham House Group Limited (1995)]. The author has been mindful of this danger when writing this case study. Thus, where comparisons have been made, they have been conducted solely on the environmental and human health arguments, and have not included any other considerations.

Finally, it should be noted that an EU Directive of Integrated Pollution Prevention and Control (IPPC) came into force in October 1996, to be adopted by member states by October 1999 (Council Directive 91/61/EC). At the time of writing, however, (in autumn 2000), the UK Government is still at the consultation stage, in terms of implementation. Nevertheless, when it is implemented, it will lead to IPC being superseded by a regime of IPPC. IPPC emphasises prevention rather than end-of-pipe solutions for the control of the environmental and human health effects from industries. To meet the possible requirements imposed by this shift, regulated industries will need to have at their disposal a variety of approaches, which can be operated at different scales, to address the demands such forms of regulation may impose. The approaches developed in this case study, and in the MIM case study (chapter 10), provide crucial data and assessments, aimed at facilitating the development of such approaches.

9.6. BRM lead refining processes: Inventories, assessments of potential human health and environmental effects, and comparisons

9.6.1. Background and derivation of the modelling and assessment approaches

The overriding purpose of this study is to provide BRM with a whole site scale, and individual throughputs through the site scale approaches, which will act as a practical foundation for the company for dealing with current and future environmental challenges. Though applicable on their own, they are part of a wider strategy designed to function multi-dimensionally i.e. at different scales. The structure of these approaches have, therefore, been designed to be pre-adapted for full integration into other approaches, which will be developed to operate at other scales. Indeed, the means by which such integration is to be achieved is demonstrated by the way in which appropriate elements of it have been extracted and used within the MIM case study. This study, which is a cradle-to-gate scale assessment, developed by the author for MIM, is included within this thesis, in chapter 10.

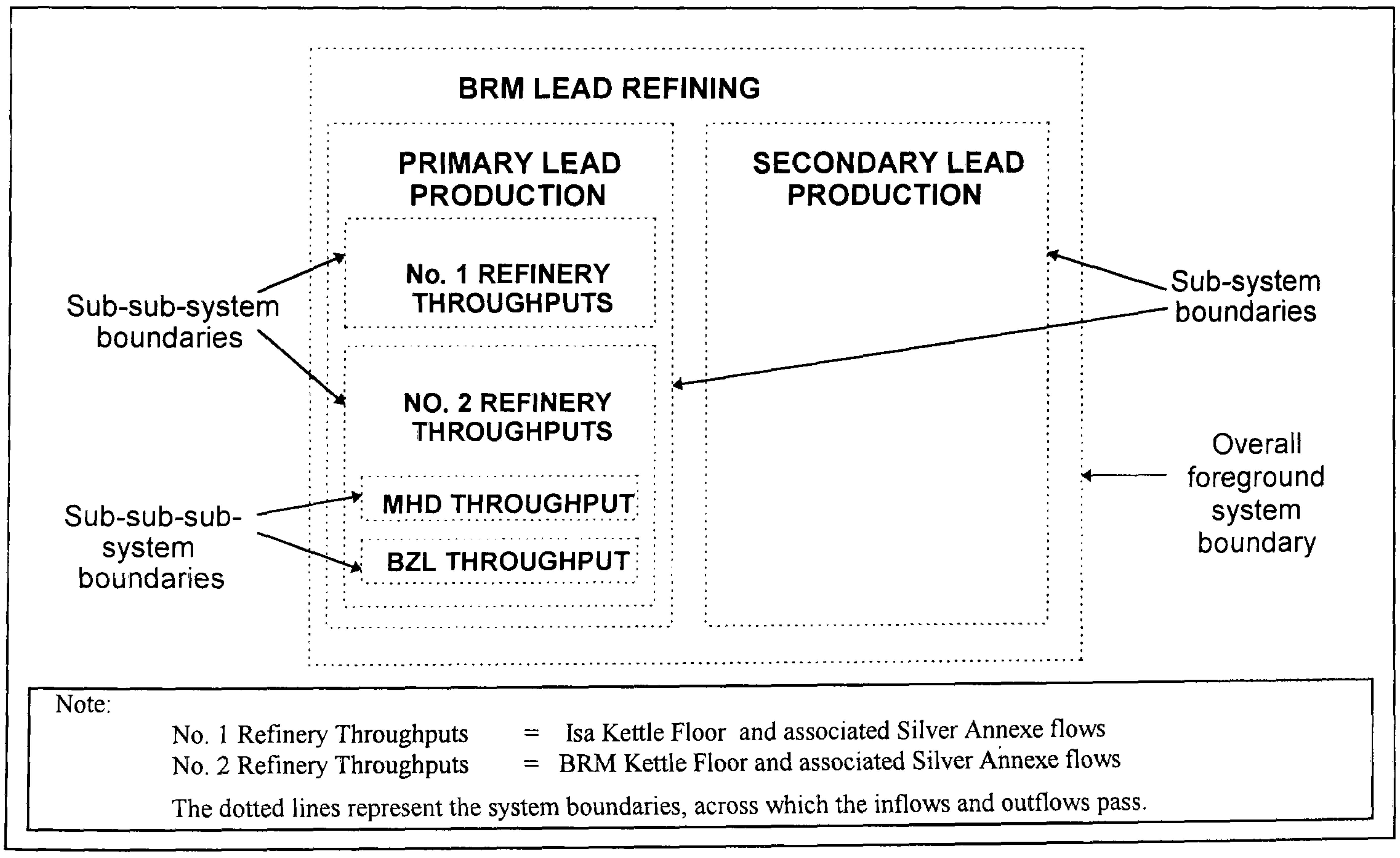
For the approaches in this case study to be fully commensurate with other scale approaches, similar framing assumptions and compatible system boundaries are required. However, the approaches also still need to be able need to take full account of the fact that the emphasis for environmental and human health effects assessments tends to differ significantly when they are required at different scales. Thus, whilst BRM is part of the MIM group of companies, it tends to have its focus of interests on its site, its throughputs, its specific emission sources, since they may be directly attributed to its own activities. MIM, on the other hand, is more concerned with the strategic planning and management of environmental and human health effects across its group of companies. Therefore, whilst it would still have some interest in individual site concerns, the primary focus of its interests tends to be directed towards more holistic (i.e. life-cycle type) concerns such as:

- comparing the environmental performance of different processes and stages in the life-cycle of a specific product,
- the environmental performance between products and
- competing systems providing an equivalent service, either as a single snapshot or as trends over over time.

Thus, approaches developed to operate at these different scales need to be able to reflect these differences, whilst, at the same time, appropriate elements of them also need to be readily extractable and able to be incorporated into approaches developed for other specific scales of focus.

These requirements are met by the use of a nested node type of structure, where individual nodes are derived from selected data modules and appropriately adjusted to fit the purpose to which they are put within the tool. Thus, the system boundaries become defined by the number and type of modules, which have been derived and linked together. The fundamental framing assumptions also tend to be similar, as regardless of the scale at which the approach has been developed, the approach is based on aggregated nodes derived from similarly constructed modules and in some cases some of the modules selected for use in different approaches may actually be the same.

Figure 9.4, which indicates the overall system structure for the BRM site scale assessment.



Note:

No. 1 Refinery Throughputs = Isa Kettle Floor and associated Silver Annexe flows
No. 2 Refinery Throughputs = BRM Kettle Floor and associated Silver Annexe flows

The dotted lines represent the system boundaries, across which the inflows and outflows pass.

Figure 9.4 The nested structure of the BRM site, and individual throughputs through the site scales of the inventories and assessments

Figure 9.4 indicates that the MHD, BZL and No. 1 Refinery Throughputs and the Secondary Lead Production are atomic nodes (i.e. they are not further subdivided) whilst Primary Lead Production, the No. 2 Refinery Throughputs and the BRM Lead Refining Operations are system nodes (i.e. they are composed of aggregates of lower order nodes). The atomic nodes are derived from modules, which have been given the same name and these contain the data collected from the BRM site. Atomic nodes and modules differ from each other, in that the latter contain the actual site data, normalised to the functional output for that process only, whilst the former contain data normalised appropriately to the functional unit of output for the whole system being modelled. The quantity and nature of the functional units of output used in this case study are identified and discussed in section 9.6.2.2.

The assessments are also directly compatible with the cradle-to-gate scale assessment developed by the author in the MIM case study, since two of the same modules; the No. 1 Refinery Throughputs and the BZL Throughput modules, have also been used to derive atomic nodes in the MIM case study assessments.

This type of approach has been possible through the adaptation of the TEAM™ LCA modelling software to a site-specific mode. The advantages and disadvantages of the use of the TEAM™ software for this purpose are discussed in section 9.6.4 of this case study, and in the Overall Conclusions and Recommendations (in chapter 11).

9.6.2. Scopes of the Inventories and Assessments

From the foregoing discussion, it is apparent the approaches developed in this case study are fully compatible with life-cycle based approaches, such as the MIM cradle-to-gate LCAs provided in the MIM case study (chapter 10). For LCA, there is now an international standard, ISO 14040:1997, which considers the general framework of LCAs and which defines their "key features". There is also an ISO standard, ISO 14041:1998, defining the characteristics of the LCA phases of goal and scope definition, and inventory analysis. Whilst the assessments in this case study are not LCAs themselves, their methodologies need to contain most of the key features identified in the ISO standard, if they are to achieve such full compatibility with LCA approaches. Hence, those ISO key features for LCA, which are applicable for these inventories and assessments, are reviewed below.

9.6.2.1. System boundaries

The foreground system boundary has been placed around what are considered, currently, to be the major concerns of the BRM site. These are the environmental and human health effects of its primary and secondary Pb and Ag refining activities. Background activities (i.e. ancillary material productions and transports, fuel production and transports, and waste disposal operations and transports) have also been considered, but have been characterised using databases which model the average or typical conditions. Thus, the foreground part of the system has been modelled using site specific data, whilst the background part has been modelled using generalised data. This approach, has been adopted, since it is peoples' concerns regarding the human health and environmental effects of its metal refining operations (i.e. its foreground system operations) which threaten, most directly, the future of BRM and the Pb industry. Therefore, detailed knowledge of the site specific effects, is necessary to provide the material basis for making decisions at this crucial scale. For the background activities, however, the environmental challenges tend to be similar to those for the rest of industry in general. Hence, it is considered appropriate, currently, to model these using such generalised data. However, since the framework is inherently flexible, the foreground system boundary could be easily modified, in the future, to incorporate one or more of these background processes within the foreground, as and when it might be deemed necessary.

Transports flows, of raw material and fuel inputs and waste material outputs, have been treated as crossing between the foreground and background systems.

The foreground inflows and outflows involved in the primary Pb and secondary Pb production sub-systems, illustrated in Figure 9.4, are detailed in Figures 9.2 and 9.3 respectively. Figure 9.3 also identifies separately the flows involved in the No.1 (i.e. the Isa stream) and No.2 (i.e. the BRM stream) refinery throughputs.

9.6.2.2. Functional Units of Output

ISO 14040:1997, notes that the scope of an LCA study should state clearly the functions of the system being studied, and that the functional unit should be a measure of the functional outputs of this system. Hence, it is arguing that the functional output should be based on the units of service which the product(s) provide. This definition for the functional unit is necessary to ensure a common basis for comparison of results between LCAs, which have different material outputs but have similar functional outputs.

However, the assessments conducted in this case study all have functional outputs based on mass (a material output). This is considered to be acceptable, for these assessments even though the approach is not fully compliant with the ISO standard, for the following reasons:

1. The assessments are not themselves LCAs.
2. All comparisons which are made, are between outputs similar both materially and functionally. Therefore, the use of functional outputs based on units of service would make no effective difference to any of these comparisons.
3. The assessments have been designed to meet, most effectively, the needs of BRM site personnel, who normally think in terms of mass rather than functionality.
4. Future iterations of the modelling may be adapted, with relative ease, to have a functional output based on units of service.

In each assessment, the functional unit is 1000 kg of the material output in question.

9.6.2.3. Allocated and Unallocated Systems

Primary refined Pb and Pb alloy production gives rise to several co-products and by products, and secondary refined Pb and Pb alloy production give rise to a polypropylene by-product. The modelling of such multiple outputs may be achieved in two possible ways. Either, the contributions to the inflows and outflows in the inventories associated with the co-products and/or by-products may be excluded, with the result that the inventories represent only that fraction of the total which is associated with the Pb and Pb alloys. Or, the contributions of the co-products and by-products are included. In the former case, the product outputs are 1000 kg of refined Pb and Pb alloys only. In the latter case, the products are 1000 kg of the refined Pb and Pb alloys, plus the appropriate quantities of the other co-products and/or by-products. In the latter case, data normalised to the functional unit of output, but not otherwise manipulated, are used. However, in the latter case, some form of allocation (i.e. manipulation) is required.

Both approaches are valid as means of modelling systems, certainly at the life-cycle scale. However for BRM, the local environmental challenges it faces are thought more likely to require it to assess the total impacts of its activities. As a result, unallocated approaches are expected be of greater use helping BRM, since they account for such total effects. It is for this reason, that in this case study, the unallocated system only has been considered. This differs from the MIM case study (chapter 10), where both allocated and unallocated systems have been modelled, However allocation in TEAM^{TM}, is achieved simply by altering partitioning factors in the modelling. Therefore, should an allocated model be required in the future, this may be produced relatively easily by modifying the unallocated models.

9.6.2.4. General Characteristics of the Data Used

The foreground system data, used to develop the input/output data sheets (supplied in the Portfolio), have been produced from data provided by BRM site personnel and cover one calendar year. Therefore, they are actual production and consumption data. A data coverage of one calendar year, has been selected, to try to ensure that short term fluctuations, and those associated with seasonal variations, are taken into account. In some cases, it has been necessary to partition whole site data to each of the throughputs through the BRM site. Where this has been necessary, the means by which it has been achieved is indicated clearly in the notes at the bottom of the relevant input/output sheet. In addition, its potential influence upon the overall findings of the assessments is assessed by sensitivity analysis, and in the discussion and conclusions to the case study (section 9.6.4). The likely precision, completeness and representativeness of the data are also characterised on the input/output sheets by the use of a scoring system for each characteristic considered. A description and a key to this scoring system prefaces the BRM case study input/output data sheet set in volume 3 of the Portfolio.

Transport data (which cross between the foreground and background systems) have been provided by site personnel, though DEAM^{TM} transport data modules have been used to derive the transport inventory data sheets, which are also enclosed in volume 3 of the Portfolio. Data characterisation columns, similar to those for the other material and fuel input/output data sheets, have been provided on these sheets.

Since it is not practical to model every single input or output, however trivial, only those which are considered 'significant' should be modelled. The criteria which the author considers should be used for identifying whether an input or output is 'significant' or not, and hence is to be monitored, is possession of one or more of the following:

- it is already monitored by BRM,
- its consumption or production is a concern to the general public on environmental, human health or resource depletion grounds, or
- it is tracked in the MIM case study, and so should be tracked in the BRM inventories, if total compatibility with them is to be achieved.

The author has not been able to obtain all possible data pertaining to all of these categories. For example, no data have been obtained for lubricant consumption, even though it is a 'significant' concern on the grounds that its production and transport to the site of use consumes significant quantities of raw materials and fuels. However, since the expectation is that the inventories and assessments developed herein will continue to be developed, any such omissions will be overcome in future iterations

The uncertainty associated with every modelled input and output flow, is considered qualitatively in the TEAM^{TM} module and node flow information boxes, which may be accessed by pressing the <INFO> buttons.

Further discussion of the overall uncertainty associated with the findings of the assessments is provided in the overall discussion and conclusions to this chapter (in section 9.6.4).

9.6.3. Inventories, Assessments and Comparisons

9.6.3.1. Introduction

The data input/output sheets provided in the Portfolio, show two types of data, namely:

1. The actual annual input and output data for the process in question. (Most of these data, are as supplied by BRM site personnel. However, as noted in section 9.6.2.4, in some cases it has been necessary to estimate certain flows using partitioning parameters.)
2. Annual input and output data, normalised to the functional unit of output for the process being modelled. (These data have been calculated directly from the actual annual input/output data.)

The input/output data are of two types: terminal exchanges and non-terminal exchanges. Terminal exchanges, are those which either input into the process in question directly from the environment, without additional processing, or output directly into it, and undergo no further processing. Non-terminal exchanges, are those which input into the process in question within the foreground system, from other processes within the background system (such as ancillary material manufacturing operations), or output from the process in question within the foreground system, to other processes within the background system (such as waste disposal operations).

Wherever possible, the normalised non-terminal input/output exchanges have been linked to background system DEAM^{TM} data modules using the TEAM^{TM} software. The DEAM^{TM} modules provide cradle-to-gate life cycle inventories (LCI) for each the raw material and fuel inputs, and gate-to-grave inventories for each of the outputs. The TEAM^{TM} software, adjusts the values in these database inventories so they are representative for the quantity of the material or energy they are being used to model (i.e. the functional output).

The inventories associated with each of the DEAM^{TM} data modules, plus any terminal inputs and outputs for the process in question, are compiled into overall inventories, which include both the background and foreground parts of the system, for the process in question. (These inventories are provided in volume 4 of the Portfolio.) In a some instances, however, suitable DEAM™ or other LCI data to model certain types non-terminal exchanges are not currently available. In most cases it has been possible to model the LCI data for the exchange using a close equivalent. For example, steel banding has been modelled using steel coil. In a very few cases, however, no suitable data are available. Where this has occurred, it has not been possible to account for the full inventories associated with these exchanges. Where such approximations or omissions have been necessary for a process, this is clearly recorded, both within the TEAM™ information box for the flow, and in the notes in the input/output data sheets in the Portfolio for the flow. They are also summarised in section 9.6.4, where their potential influences on the overall findings of the case study are assessed.

Environmental impact assessments have been generated using the TEAM™ software from these inventories. In accordance with common practice in LCAs, the categories used for these impacts are similar to those in Table 5.1. Hence, these environmental impact assessments also assess human health impacts as they are accounted for in the impact category of Human Toxicity. This approach, called the 'Problem Oriented Approach', was developed by the Centre of Environmental Science (CML), of the University of Leiden, in The Netherlands. In the CML approach, environmental impacts are calculated by multiplying individual types of terminal exchanges (i.e. inventory flows) by a weighting factor. This weighting factor is assigned relative to a reference burden. Thus, for example, for the impact category of Global Warming, carbon dioxide (CO_2) is assigned a weighting factor of 1. Other emissions are assigned values greater and lesser than 1 depending on their own global warming potentials (GWP) relative to CO_2. The individual contributions from each of the exchanges are then summed to produce the total contribution for that impact category.[36]

A complete set of inventories are provided in Volume 3 for the systems which have been studied. This has been conducted, as they are intended to form the basis of a database for BRM and other interested parties, which, it is hoped, will be updated on a regular basis (e.g. annually). However, the inventory graphs provided at the end of this chapter in section

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[36] Explanations of the process of impact assessment within an LCA context, of the Problem Oriented and other approaches, and of the exact method of calculation of impacts, are provided in the elective EngD module assignment, which was enclosed in volume 2 of the Portfolio (as Document 15). This document has not been supplied with this copy of the thesis.

9.7, are merely a selection of the more 'significant' inflows and outflows. (The criteria for deciding which flows are deemed 'significant', are provided in section 9.6.3.3.2.) Similarly, the impact assessment graphs and the graphs comparing the inventories and impacts of different systems (also in section 9.7), show only the most 'significant' inventories and impacts. They are not intended to provide an exhaustive analysis. However, should BRM, or any other parties, need to examine any other of the inventory flows or impacts in the future, this would be relatively easy task using either the inventory tables in Portfolio alone, or in conjunction with the TEAM^{TM} LCA software.

Finally, it should also be noted, the environmental impacts calculated in the impact assessments are in fact potential impacts i.e. the types and quantities of the flows in the inventories for the systems concerned, have the potential to give rise to impacts in each of the categories to the extent indicated. Whether or not they occur, in reality, depends on the actual fate in the environment of the inventory flows concerned. This question is not addressed in these case studies. To address it, site impact pathway assessment (IPA) approaches, as described in section 5.3 of chapter 5 would be required. To give a complete picture, these IPA approaches would also need to be conducted for both the foreground and background systems.[37]

9.6.3.2. Inventories

9.6.3.2.1. Introduction

Primary operations data, sourced from records kept by BRM personnel, were used to calculate the total annual inputs and outputs not only for the modelling in this case study but also for the BRM primary operations data modules in the LCA of MIM's primary lead products, which is included in the MIM case study (chapter 10). The year which these data cover is the calendar year 1998. However, for secondary operations, summary data only were obtained. These cover the BRM company years of 1997 to 1998 and 1998 to 1999. (The BRM company year runs from July to June inclusive.) Hence, the start and finish dates of the two sets of data differ, though they do overlap.

To account for this difference in coverage, the whole site scale modelling using TEAM™, has been duplicated for two completely separate systems, namely:

- 'BRM: Whole site (a)', which contain primary operations data for the calendar year 1998 and secondary operations data for the company year 1997 to 1998, and
- 'BRM: Whole site (b)', which contain primary operations data for the calendar year 1998 and secondary operations data for the company year 1998 to 1999.

This has been necessary due to the time constraints imposed in the preparation of this thesis. These have meant it has not been possible to remodel either the primary or the secondary data, so they both have the same start and finish dates. This should not necessarily be viewed as a major shortcoming in the modelling, for the following the following two reasons.

- Firstly, using a complete year of data ensures that average values over that year are used in the modelling.
- Secondly, average values are likely to remain similar from one year to the next, unless significant changes occur to the system being modelled. This is because operators normally try to maintain as near a steady state with the process as they can.

Therefore, even though the start and end dates differ, the modelling still provides a useful estimation of the typical inventories associated with the functional unit of output.

Nevertheless, since industrial systems do, inevitably, show some variation, separate modelling has been conducted with both sets of secondary data. Table 9.1 indicates the complete set of Excel file based inventory tables which have been collated for the 'BRM: Whole site (a)' TEAM model, and which have been also enclosed as sheets in volume 4 of the Portfolio. A similar set of Excel files (and sheets in volume 4 of the Portfolio), have also been created for the 'BRM: Whole site (b)' model.

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[37] This represents an area for further research.

Excel file name	Sheet name (where applicable)	Comprising nodes	Level	Code
ecobalBRMwholesite(a)unalloc2(1)		BRM: Lead Material Processing Operations at Northfleet, Kent, UK	1	(a)/-
		BRM: Primary Refined Lead Production Operations	2	(a)/X
		BRM: Refined Lead Production Operations (BRM stream throughput)	3	(a)/X1
		BRM: Refined Lead production Operations (Isa stream throughput)	3	(a)/X2
		BRM: Secondary Refined Lead Production Operations	2	(a)/Z
ecobalBRMwholesite(a)unalloc2(2)	Section 1 of 2	BRM: Refined Lead Production Operations (BRM stream throughput)	3	(a)/X1
		BRM: Lead Refining Operations (BRM floor throughput)	4	(a)/X1,1
		BRM: On-site Electricity Generation Operations (BRM floor throughput)	4	(a)/X1,2
		BRM: Overheads to BZL Input to BRM Floor Throughput Operations	4	(a)/X1,3
		BRM: Overheads to MHD Input to BRM Floor Throughput Operations	4	(a)/X1,4
ecobalBRMwholesite(a)unalloc2(2)	Section 2 of 2	BRM: Refined Lead production Operations (Isa stream throughput)	3	(a)/X2
		BRM: Lead Refining Operations (Isa stream)	4	(a)/X2,1
		BRM: On-site Electricity Generation Operations (Isa floor throughput)	4	(a)/X2,2
		BRM: Overheads (Isa stream operations)	4	(a)/X2,3
ecobalBRMwholesite(a)unalloc2(3)		BRM: Secondary Refined Lead Production Operations	2	(a)/Z
		BRM (secondary): CX Plant Operations	3	(a)/Z1
		BRM (secondary): Overheads of Secondary Processing Operations	3	(a)/Z2
		BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations	3	(a)/Z3

Table 9.1: Complete list of BRM nodes for which inventories have been developed in this case study, using the 'BRM: Whole site (a)' TEAM^{TM} model, and collated in the Portfolio

Each of the files listed in Table 9.1 contains inventories for the nodes indicated. The levels and codes identify their position according to the modelling structure shown in the sheets entitled: 'BRM Whole Site: TEAM^{TM} Model Structure' (which preface the collated inventory sheets in the Portfolio, and which have been enclosed as Document A1 in the

Appendix). The collated inventory sheets have been designed, so the data in the lower levels (i.e. higher level numbers) are subsumed into the higher levels. For example, in Excel file: 'ecobalBRMwholesite(a)unalloc2(3)', the level 3 data atoms are subsumed within the higher level (level 2) data atom which is: 'BRM: Secondary Refined Lead Production Operations'.

The collated inventories (in the Portfolio) are all of systems nodes, even at the lowest levels. For example, the model structure information sheets (in volume 3 of the Portfolio) indicate that, even data atom: 'BRM: Overheads (Isa stream operations)' code: (a)/X2,3, which is at the lowest level at which inventory data have been collated, is a system node. This approach has been adopted, since atomic nodes contain input and output data only in the categories originally entered by the author. Hence, they have been collated, and are included in volume 4 of the Portfolio, as a separate set of sheets.

Separate modelling has also been conducted at the secondary operations scale, for the BRM company years 1997 - 1998 and 1998 - 1999. The modelling and the data from these models have also been subsumed directly within the BRM Whole site (a) and (b) models. Inventories for the secondary operations models have not been included separately in volume 3, since they are incorporated within the sheets for Excel files 'ecobalBRMwholesite(a)' and 'ecobalBRMwhole site(b)'.

The categories of the inventory articles in the collated inventory sheets, are as indicated in Table 9.2. Each set of sheets contains over 350 different data articles.

Inflow/ Outflow	**Inventory category**	**Notes**
Inflow	*Resource consumptions*	Raw materials from the environment
Inflow	Ancillary materials (secondary)	Miscellaneous inflows associated with DEAM^{TM} LCI data modules which have been used in the modelling
Inflow	Ancillary materials (primary)	Miscellaneous inflows for which there were LCI data available to model them. i.e. they represent data gaps.
Inflow	Energy (secondary)	Energy consumptions associated with DEAM^{TM} LCI data modules which have been used in the modelling
Outflow	*Emissions to atmosphere*	Emissions to the environment
Outflow	*Emissions to atmosphere (radioactive)*	Emissions to the environment
Outflow	*Emissions to ground*	Emissions to the environment
Outflow	*Emissions to water*	Emissions to the environment
Outflow	*Emissions to water (radioactive)*	Emissions to the environment
Outflow	*Ancillary material outputs (wastes)*	Wastes deposited into the environment
Outflow	Material outputs (products, co-products and by-products)	Products, plus co-products and by-products (if any)
Outflow	Energy Reminders	Intermediate energy consumptions in various categories.
Note: Those categories which are terminal with the environment have been identified using italic script.		

Table 9.2: Data categories shown in inventory sheets

The principal aim of the research has been to create a database which BRM, MIM and industry in general may use for their own comparisons and assessments. The sheets enclosed in the Portfolio provide this database. However, as an indication of its usefulness for helping to develop environmental performance indicators and, hence, for aiding environmental decision making, some simple comparisons of some of the more 'significant' inventory articles have been conducted and enclosed in this thesis. Any of the other inventory inflows and outflows may also be compared relatively easily, at a later date, should someone wish to do so.[38]

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[38] This is an area for further research.

9.6.3.2.2. Identifying 'significant' flows in the inventories

The criteria for identifying inventory flows as being 'significant', are to be included in one or more of the following categories.

1. To be in the TEAM Plus^{TM} list of automatic charts, which are part of the software package provided by Ecobilan, for the analysis of inventory data generated using their TEAM^{TM} modelling tool. [See Ecobilan (1995).]
2. Specific flows considered significant by BRM, its regulators and MIM. For example emissions reported by BRM to the UK EA, or monitored by MIM in its annual environmental reports. [See BRM (1995)and MIM Holdings Ltd. (1998) respectively.]
3. Flows which are closely associated with the flows selected under any of the other criteria. For example, emissions of '(a) Hydrocarbons (except methane)' are significant because they are reported on an annual basis to the UK EA. Closely associated with these are emissions of '(a) Hydrocarbons (unspecified)'. Hence, these also need to be considered.
4. For the quantity of the flow at the highest system level, in either of the modelled years, when normalised to the functional unit of 1000 kg of final lead product, to be:
    a) ≥ 5.00E+00 kg for resource consumptions,
    b) ≥ 1.00E+03 mg, kg or litre for ancillary materials (secondary),
    c) ≥ 1.00E+03 g for air, water and ground emissions, and
    d) ≥ 5.00E+00 kg for wastes.

Criteria 1 and 2 are normative rather than arbitrary, since these flows would be expected to reflect, broadly speaking, current concerns. Hence, the flows thus selected as 'significant', reflect societal concerns. However, it has also been necessary to adopt two additional non-normative criteria. The aim of the first (criterion 3) is to ensure that flows allied to the identified criteria are also tracked. The aim of the second (criterion 4), is to try to ensure other terminal exchanges are identified, which occur in relatively large quantities but which are not currently identified by any of the other criteria. In future iterations of the modelling, the two latter criteria should be replaced by more normatively based approaches.[39]

The flows in Table 9.3 have been identified as 'significant' using the four criteria.

Flow	Units
Resource consumptions	
(r) Coal (in ground)	kg
(r) Iron (Fe, ore)	kg
(r) Lead (Pb, ore)	kg
(r) Limestone (CaCO3, in ground)	kg
(r) Natural Gas (in ground)	kg
(r) Oil (in ground)	kg
(r) Sand (in ground)	kg
(r) Uranium (U, ore)	kg
Ancillary materials (secondary)	
Raw materials (unspecified)	kg
Water Used (total)	kg
Reminders	
E Total Primary Energy	MJ
Electricity	MJ elec
E Fuel Energy	MJ
E Feedstock Energy	MJ
E Renewable Energy	MJ
E Non Renewable Energy	MJ
Emissions to air	
(a) Carbon Dioxide (CO2, fossil)	g
(a) Carbon Dioxide (CO2, biomass)	g
(a) Carbon Monoxide (CO)	g
(a) Hydrocarbons (except methane)	g
(a) Hydrocarbons (unspecified)	g
(a) Methane (CH4)	g
(a) Nitrogen Oxides (NOx as NO2)	g
(a) Nitrous Oxide (N2O)	g
(a) Sulphur Oxides (SOx as SO2)	g
(a) Sulphur Dioxide (SO2)	g
(a) Hydrogen Chloride (HCl)	g
(a) Particulates (unspecified)	g
(a) Arsenic (As)	g
(a) Cadmium (Cd)	g
(a) Lead (Pb)	g
(a) Zinc (Zn)	g
(a) Copper (Cu)	g
(a) Tin (Sn)	g
(a) Antimony (Sb)	g
(a) Tellurium (Te)	g
(a) Selenium (Se)	g
(a) Metals (unspecified)	g

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[39] This is an area for further research.

Emissions to water	
(w) Suspended Mattter (unspecified)	g
(w) COD (Chemical Oxygen Demand)	g
(w) BOD5 (Biochemical Oxygen Demand)	g
(w) Arsenic (As3+, As5+)	g
(w) Cadmium (Cd++)	g
(w) Lead (Pb++, Pb4+)	g
(w) Zinc (Zn++)	g
(w) Copper (Cu++)	g
(w) Antimony (Sb++)	g
(w) Metals (unspecified)	g
(w) Ammonia (NH4+,Nh3, as N)	g
(w) Fluorides (F-)	g
(w) Chlorides (Cl-)	g
(w) Nitrates (NO3-)	g
(w) Sulphate (SO4--)	g
(w) Sulphates (SO4--)	g
(w) Phosphates (PO4 3-, HPO4--, H2PO4-, H3PO4, as P)	g
(w) Suspended Matter (unspecified)	g
Ancillary material outputs (wastes)	
Waste (total)	kg
Waste (hazardous)	kg
Waste: Non Mineral (inert)	kg
Waste (municipal and industrial)	kg
Waste: Mineral (inert)	kg
Waste (industrial, miscellaneous)	kg
Waste: Mining	kg
Waste: Slags and Ash (unspecified)	kg
Water Used (total)	litre

Table 9.3: 'Significant' inventory flows

Note, 'significant' flows are distinct from 'dominant' flows, identified by dominance analysis in section 9.6.4.1. This is because, 'significant' flows are simply those which are considered likely to be of the most direct interest for BRM and other stakeholders. The 'dominant' flows, however, are used to assess the influence of variability and other uncertainties in the input/output data (entered by the author), on the findings from inventory analyses and effect assessments (section 9.6.4.1). Whilst there may be considerable overlap between the two, many of the 'dominant' flows may result from specific processes not under the direct control of BRM or other stakeholders. Hence, they would not be directly 'significant' for them. Conversely, some 'significant' flows may not be dominating, in terms of their ability to give rise to the specific impacts which have been considered in this case study.

9.6.3.2.3. Graphs of selected flows of interest

It is impractical within the confines of this thesis, to plot graphs for all of the flows identified in Table 9.3, either at the whole system level or to examine how the nodes listed in Table 9.1 contribute to these total flows.[40] Indeed, since the aim of the research has been create a database, to be used into the future as an aid to decision making, these are tasks which should be conducted only as and when the need arises. However, as an illustration of the capabilities of the modelling approach, graphs for some of the more important current concerns are provided at the end of the chapter (section 9.8).

The 'significant' inventory flows selected for graphing are:

1. '(r) Natural gas (in ground)',
2. '(r) Oil (in ground)',
3. '(a) Carbon Dioxide (CO2, fossil)',
4. '(a) Sulphur Oxides (SOx as SO2)',
5. '(a) Lead (Pb)',
6. '(a) Cadmium (Cd)',
7. '(w) Lead (Pb++, Pb4+)' and
8. '(w) Cadmium (Cd++)'.

These graphs cover the modelling year considered in the 'BRM: Whole site (a)' TEAM^{TM} model only. However, data for the BRM company year 1998 - 9 is compared with that for the 1997 - 8 company year in section 9.6.3.4. The data for this modelling is the same as that which is also used in the 'BRM: Whole site (a)' and 'BRM: Whole site (b)' TEAM^{TM} models.

For each selected inventory flow, four graphs have been produced. Each graph shows the overall total for the flow, at the highest node level being examined in that graph, on the left hand side with the disaggregated flows to the right. The manner in which the nodes have been disaggregated follows the sequence in Table 9.1. The data used to compile these graphs have been enclosed in volume 5 of the Portfolio (in Document 1).

The graphs are enclosed at the end of the chapter (section 9.8), and are discussed in section 9.6.4.5.1.

9.6.3.3. Assessments

9.6.3.3.1. Introduction

The inventory data from the 'BRM Whole site (a)' and 'BRM Whole site (b)' TEAM™ models may also be used to conduct a variety of different assessments. It was noted in section 9.6.2, however, that one of the aims in this case study is to provide assessments fully compatible with the LCA based modelling and assessment approach developed for the MIM case study (chapter 10). Hence, in compliance with general current practice, the assessments in the MIM case study, follow the guidelines for the impact assessment phase of LCA, devised by the Society for Environmental Toxicology and Chemistry (SETAC), and incorporated in the SETAC Code of Practice (Consoli et al. 1993). To be fully compatible with the MIM case study LCA, the assessments which have been devised in this case study also follow this set of guidelines.[41]

Consoli et al. (1993) and Lindfors et al. (1995), state that the following categories of impacts should be assessed if comprehensive assessments are to be conducted:

1. resource depletion,
2. human health effects, and
3. ecological effects.

However, it is not the aim in this thesis to provide a fully comprehensive assessment. As with the inventory analysis, conducted in section 9.6.3.2, the aim is to provide a database, which may then be used as a tool by interested parties to conduct their own assessments, as and when the need arises. The principal purpose of the assessments provided below, is, therefore, to demonstrate some of the capabilities of the approach as a decision support tool. Hence, the assessments of the potential impacts consider some of the impacts only, and only at some of the levels at which these assessments may be made.

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[40] This is an area for further work.

9.6.3.3.2. Graphs of selected potential impacts

The types of impacts which have been considered in the assessment graphs at the end of the chapter (section 9.8) are:

1. Depletion of non-renewable resources -CML [Heijungs et al. (1995)],
2. Air Acidification -CML [Heijungs et al. (1995)],
3. Aquatic Ecotoxicity -CML [Heijungs et al. (1995)],
4. Depletion of ozone layer (high) [WMO (1991)],
5. Depletion of ozone layer (low) [WMO (1991)],
6. Eutrophication -CML [Heijungs et al. (1995)],
7. Eutrophication (aquatic) -CML [Heijungs et al. (1995)],
8. Human toxicity -CML [Heijungs et al. (1995)]
9. Terrestrial ecotoxicity -CML [Heijungs et al. (1995)],
10. Greenhouse effect (direct, 500 years) [IPCC (1994)],
11. Greenhouse effect (direct, 100 years) [IPCC (1994)],
12. Greenhouse effect (direct, 20 years) [IPCC (1994)],
13. Photochemical oxidant formation (high) [WMO (1991)], and
14. Photochemical oxidant formation (low) [WMO (1991)].

These have been conducted using the TEAM^{TM} Plus assessment database version 2.5.

The inventory data which have been used to conduct the assessments are from Excel file: 'ecobalBRMwholesite(a)unalloc2(1)'. Hence, the 'BRM: Whole site (a)' TEAM^{TM} model only has been considered in this section.

The nodes which have been graphed are:

1. 'BRM: Lead Material Processing Operations at Northfleet, Kent, UK'
2. 'BRM: Primary Refined Lead Production Operations',

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[41] These are reviewed in the elective EngD module assignment, in volume 2 of the Portfolio (in Document 15). This document has not been supplied with this copy of the thesis.

3. 'BRM: Refined Lead Production Operations (BRM stream throughput)',
4. 'BRM: Refined Lead Production Operations (Isa stream throughput)', and
5. 'BRM: Secondary Refined Lead Production Operations'.

These graphs are discussed in section 9.6.4.5.2.

The data used to compile these graphs have been enclosed in volume 5 of the Portfolio (in Document 2)

9.6.3.4. Comparisons

Graphical comparisons have also been conducted which indicate the differences in the inventory flows and potential impacts, at the secondary operations scale, between the BRM company years 1998 - 1999 and 1997 - 1998.

The TEAM^{TM} models of secondary operations (i.e. 'BRM Sec. Refined Pb Production 1998-9' and 'BRM Sec. Refined Pb Production 1997-8') were used, .rather than simply extracting the data from the BRM Whole site (a) and (b) models. This is because, the normalised secondary refined lead outputs in the 'BRM Whole site (a) and (b) models are 137.439 and 125.812 kg respectively. In order to make direct comparisons between the two years, the data would need to be scaled to a similar output. (In compliance with the convention used in these case studies, this normalisation quantity is 1000 kg of output for each year.) Whilst the inventory may be estimated simply by multiply all values by an appropriate scaling factor, calculation of the potential impacts is more complicated. Hence the 'BRM Sec. Refined Pb Production' models were used, as they provided the simplest method available for conducting the calculation.

In both the inventory and effect comparisons, percentage differences with a negative sign, suggest an improvement in modelled year 1998 - 9 compared with 1997 - 8 and those with a positive sign a deterioration. The data used to compile these graphs have been enclosed in volume 5 of the Portfolio (in Document 3)

The graphs are enclosed at the end of the chapter (section 9.8), and are discussed in section 9.6.4.5.3.

9.6.4. Discussion and uncertainty analysis

9.6.4.1. The need for uncertainty analysis

As noted in section 9.6.2, this case study aims to achieve full compatibility with other LCA approaches. Efforts have been conducted to ensure this compatibility exists not only in the modelling, but also in the manner in which the data outputs are analysed. Hence, where aspects of the international standard (ISO14041 (1998)E) for the "goal and scope definition and inventory analysis" phases of LCA, are relevant to the BRM case study. they have been adopted.

All three types of uncertainty (i.e. epistemological, methodological and technical) are present, to varying degrees, in modelling and assessment approaches such as these. In section 4.4.2.1 of chapter 4, it was explained that, within an LCA context, epistemological uncertainty corresponds to ignorance of the system (i.e. lack of understanding of system behaviour), methodological uncertainty with bias from the choice of model and the system boundary, and technical uncertainty with inexactness [Lindfors et al. (1995)]. It was also pointed out, that whilst the scientific method aims to keep such uncertainties to a level where they are so small they are negligible (within defined parameters), for large scale problems (such as those dealing with the environment) the uncertainties are likely to be high, and hence significant. However, decisions still have to made, even at these large scales. In addition, as shown in Figure 4.2, the stakes of such decisions may be very high. For example, it was argued, in chapter 2, that there are potential threats to the future of the entire lead industry. Therefore, if the lead industry is to argue its environmental case effectively using data from such modelling, an effective assessment of the influence of uncertainties, on all findings is essential.

ISO14041 (1998)E addresses this question, when it states:

> "the results of the LCI [life-cycle inventory] shall include a data quality assessment and sensitivity analysis of inputs, outputs and methodological choices in order to understand the uncertainty of the results" (p. 13).

It notes also, that the interpretation of an inventory analysis should consider:

1. whether the delineations of the functional unit are appropriate,
2. whether the definitions of the system boundaries are appropriate, and
3. the limitations identified by the data quality assessment in the sensitivity analysis.

However, it does recognise that uncertainty analysis is a technique still in its infancy, but states, nevertheless, that:

> "it would help to characterise uncertainty in results using ranges and/or probability distributions".

Whilst this case study is not an LCA. It has been designed to integrate fully with the cradle-to-gate LCA in the MIM case study (chapter 10). Hence, it needs to meet the above requirements if such integration is to achieved. Also, complete transparency and verifiability in the modelling, assessment and analysis are vital pre-requisites if statements are to be made using such data. Therefore, trying to meet such requirements is a laudable aim in itself.

To meet the requirements of the standard, two forms of uncertainty analysis have been adopted for this case study, namely:

1. sensitivity analysis, involving simulations of the effects of manual changes to key variables (i.e. inputs and outputs), and
2. Monte Carlo simulations, an automatic sensitivity analysis approach, to derive uncertainty ranges for the flows.

The manner in which these have been conducted, and their findings are discussed sections 9.6.4.2 and 9.6.4.3 below. In the light of these analyses, the findings of the inventory, assessment and comparison graphs in sections 9.6.3.2 to 9.6.3.4 are then discussed in section 9.6.4.4.

9.6.4.2. Sources of uncertainty in the system models

For each of the three categories of uncertainty, relating to the BRM whole site modelling, the major general contributing sources are first reviewed, and various specific aspects are then discussed. However, all three categories are inter-linked and overlapping to varying degrees.

The likely influence of these uncertainties, on the findings of the inventory and potential environmental impact assessment graphs, is discussed and assessed in section 9.6.4.5. Additional uncertainties, associated with the calculation of potential environmental impacts from inventory flows, are also discussed and assessed in the section.

9.6.4.2.1 Epistemological uncertainties:

a) General sources:

1. It is not known whether or not all of the DEAM and other data modules, used to provide background inventories, really do provide industry representative data. This is partly a result of the lack of detailed source information for some modules, and partly due to the time constraints in the preparation of this thesis, which have meant that it has not been possible to cross check them all.[42]
2. The actual effects of the use of substitute modules, to provide approximate background inventories, where exact matches could not be obtained, are unknown. However, their relative contributions to the flows and potential environmental impacts, which have been graphed, are discussed in section 9.6.4.5. A list of the flows which have had to be thus treated is provided in part b), in Tables 9.4 and 9.5.

3. Insufficient data were available to model a few of the flows in the background part of the system. Whilst the overall effects on the modelling of these omissions are expected to be relatively minor (as only a very small number of flows have had to be treated thus), their actual effects are unknown. A list of the flows, where this has happened, is provided in part b), in Table 9.6.
4. In the foreground part of the system, only those flows considered 'significant' have been modelled. The exclusion of some other flows, implies they are not significant. This may not be the case. For example, significant flows may have been excluded from the modelling, due to ignorance of their existence.
5. Insufficient data were available to model one or two substances thought to warrant modelling. For example, there were insufficient data to model the consumption of lubricants for many processes. A list of inputs and outputs where this is known to have occurred is provided in part b), in Table 9.7.

b) Specific aspects:

The following are reviewed in Tables and discussed below:

1. Process material flows whose background inventories have had to be modelled using substitutes.
2. Process material flows for which it has not been possible to model the background inventories.
3. Process material flows where insufficient data were available to conduct modelling.

1. Process material flows whose background inventories have had to be modelled using substitutes:

Table 9.4 lists the material inflows and outflows (excluding water) which have had to be modelled using substitutes. Whilst Table 9.5 lists the inflows of 'mains water' and 'fresh water', which have had to be modelled using the substitute 'Water (Softened)'. In both cases, the aim has been to use substitutes which are materially are as close to the actual

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[42] This represents an area for further research.

material as possible. However, some materials are more representative than others. For this reason the likely representativeness of each substitute material is indicated in the Tables using a scoring from 1 to 4.

Flow ID code*	Inflow or Outflow	Flow name	Modelled substitute	Representativeness of approximation (scale of 1 to 5)***
BRM: Lead Refining (Isa stream) **Process code**: X2,1,1 Level: 5**				
AU3	Inflow	Nitre	Ammonium Nitrate (NH4NO3 as N)	4
AU16	Inflow	Calcium lignosulphonate	Pulp	4
AU22	Inflow	Calcium/aluminium	Magnesium (Mg)	4
AU23	Inflow	Calcium	Magnesium (Mg)	4
AU25	Inflow	Sodium	Aluminium (Al)	4
AU27	Inflow	Steel banding	Coil	2-3
BRM: Lead Refining (BRM floor throughput) **Process code**: X1,1,1 Level: 5**				
AU3 & 7	Inflow	Nitre	Ammonium Nitrate (NH4NO3 as N)	4
AU5	Inflow	Calcium	Magnesium (Mg)	4
AU25	Inflow	Steel banding	Coil	2-3
BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding **Process code**: Z2,1 Level: 4**				
AU1	Inflow	Iron borings	Starch	2-3
AU7	Inflow	Nitre	Ammonium Nitrate (NH4NO3 as N)	4

Notes:

* The Flow ID Codes are those cited in the input/output data sheets for each modelled process. (These sheets are provided for all processes in volume 3 of the Portfolio.)

** Process Codes have been produced for all of the modelled processes. The modelling structure and codes are indicated in the sheets entitled: "BRM Whole Site: TEAM^{TM} Model Structure" (enclosed in the Appendix as Document A1).

*** Scoring system for representativeness: 1 = exact match (i.e. not a substitution)
2 = good match
3 = fair match
4 = match quality uncertain

Table 9.4: Process inflows and outflows (other than water) where the background inventory data have had to be modelled using substitutes

Process name	Process code*	Level	Water Inflow	
			Flow ID**	Name
BRM: Lead refining (Isa stream)	X2,1,1	5	AU13	Towns water
BRM: Lead Refining (BRM floor throughput)	X1,1,1	4	AU16	Towns water
BRM (secondary):Smelting, Rotary Furnace, Refining and Moulding	Z2,1	5	AU20	Towns water
BRM: Overheads (Isa stream operations)	X2,3,1	5	AU1	Towns water
BRM: Overheads (BZL bullion processing)	X1,3,1	5	AU1	Towns water
BRM: Overheads (MHD bullion processing)	X1,4,1	5	AU1	Towns water
BRM (Secondary): Overheads	Z,3,1	4	AU1	Towns water
Notes: * For explanation, see the 'Notes' panel for Table 9.4. ** For explanation, see the 'Notes' panel for Table 9.4.				

Table 9.5: **Water inflows which have had to be modelled using the DEAM 'Water (Softened): Production' inventory data module**

Process	Process code*	Level	Input or output lacking a background inventory data		
			Flow ID**	Input/ Output	Name
BRM: Lead refining (Isa stream)	X2,1,1	5	AU9 AU19 AU26	Input Input Input	Calgon Antimony Tellurium
BRM: Lead Refining (BRM floor throughput)	X1,1,1	5	AU21	Input	Arsenic
BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding	Z2,1	4	AU8 AU11 AU12 AU16	Input Input Input Input	Pyrites Antimony Selenium Arsenic
Notes: * For explanation, see the 'Notes' panel for Table 9.4. ** For explanation, see the 'Notes' panel for Table 9.4. *** TTE water = Treated Trade Effluent					

Table 9.6: **Process inputs and outputs where suitable background data were not available**

Process name	**Process code***	**Level**	**Flow not modelled**		
			Flow ID**	**Input/ Output**	**Name**
BRM: Lead refining (Isa stream)	X2,1,1	5	AU13 AU14 WA3	Input Input Output	Cooling water Lubricants Lubricants
BRM: Lead Refining (BRM floor throughput)	X1,1,1	5	AU17 AU18 AU13 WA4	Input Input Output Output	Cooling water Lubricants Slag from secondary rotary, dross dross handling) Lubricants
BRM: Overheads (Isa stream operations)	X2,3,1	5	WA1	Output	Office waste
BRM: Overheads (BZL bullion processing)	X1,3,1	5	WA1	Output	Office waste
BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding	Z2,1	4	AU21/ WA3 AU18 AU19 WA4	Both Input Input Output	Office waste Surface water Cooling water General rubbish
BRM (secondary): CX Plant	Z1,1	4	AU5/ WA3 AU2 AU3 WA4	Both Input Input Output	Office waste Surface water Recycled water General rubbish

Notes:
* For explanation, see the 'Notes' panel for Table 9.4.
** For explanation, see the 'Notes' panel for Table 9.4.

Table 9.7: Process inputs and outputs where insufficient data were available to conduct modelling

Whilst the actual influence of the substitutions and omissions in Tables 9.4 to 9.7 , on any inventory and/or effect assessments, which may be conducted are unknown, some have a much greater potential to influence than others. For the inventory and assessment graphs, enclosed with this thesis, the dominant contributing sources to the overall findings have first been identified by dominance analysis, and their potential to influence these findings, has then been assessed using sensitivity analysis. The methodology of these analyses is indicated in section 9.6.

In the discussion of the inventory and assessment graphs (section 9.6.4.5), where key flows are shown to be associated with substances modelled using substitutes, their potential influence on any findings are considered and assessed. In the few cases where suitable LCI data for certain substances were not available, or where suitable data were not available to include certain substances associated with individual substances in the modelling (i.e. Tables 9.6 and 9.7 respectively), such assessment is not possible. However, the number of flows where this has occurred is relatively small and their mass contributions are relatively small. Whilst their individual influences are expected to be relatively minor, these omissions must be made explicit in any interpretations which may be made. This question is, therefore, considered further, in both the discussion of the inventory and effect graphs (section 9.6.4.5), and in the Overall Conclusions and Recommendations (chapter 11).[43]

9.6.4.2.2 Methodological uncertainties:

a) <u>General sources</u>:

1. Partitioning (i.e. allocation) approaches, where adopted, may significantly under or over estimate.
2. The assumption has been made that linear (i.e. incremental) changes in the demand for materials and/or energy from the background system and/or of wastes sent to the background, may be met by similar incremental changes in inventory values. This may not be the case in practice. For example, increased electricity demand may result in additional plant being brought into operation, rather than simply by an increase in the output from existing plant.
3. The input or output of a measured material or energy type may actually be significantly higher or lower than that indicated by the data. This bias may be due to the data not being fully representative of the full production cycle or to systematic errors in the measurement and/or subsequent handling and reporting chain.
4. The use of the arithmetic mean (the approach adopted in the modelling), assumes the distribution of the data spread is not significantly skewed. This may not be the case in practice. The data may, for example be log-normal. In such cases, the use of the geometric mean would give a less biased estimate of the ‘typical’ value.

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[43] It also represents a key area needing further research.

5. The use of the nearest truck capacity equivalents to estimate the inventories associated with transports, when an exact match is unavailable, may create systematic bias.

b) Various specific aspects:

The input/output annual data sheets for each modelled process (in volume 3 of the Portfolio), include a group of columns entitled 'Characterisation of Data'. The numbers entered into these columns characterise each input or output according to the scoring system indicated in Table 9.8.

Column number	Data descriptor
1	**Data source type**
2	**Representativeness of the data for characterising the unit 'process'**
3	**Representativeness of the unit 'process' for its purpose in modelling**
4	**Age of data**
5	**Reference**
6	**Strict confidentiality for specific data items**

Table 9.8: The criteria considered in the 'Characterisation of Data' columns in the input/output annual data sheets for each modelled process

The scoring system for the criteria in columns 1 to 3, has been modified from Wenzel et al. (1997), and assesses these characteristics against various criteria. For each flow, the number inserted in each column is intended to give a rough indication of the nature of the data with respect to each of the three descriptors. They provide, therefore, not only an indication of how representative the data which have been used are, but also an indication of the possibility of significant bias (i.e. methodological uncertainty). This is because, in general, the more representative the data are, the less likely they are to be biased. Therefore, the higher the numbers in columns 1 to 3, the greater the possibility of bias.

9.6.4.2.3 Technical uncertainties:

a) General sources:

1. Variability occurs in the inputs and outputs, relative to the functional output from the whole system.
2. Variability also occurs in the inputs and outputs, relative to the functional output from the processes with which they are directly associated.
3. Different flows may be independent variables. However, many are correlated either stepwise (i.e. between different processes in production sequences) and/or process wise (i.e. within specific processes).
4. Rounding up and rounding down errors occur inevitably when data from differing sources are aggregated in modelling such as this.

b) Various specific aspects:

The input/output annual data sheets for each modelled process (in volume 3 of the Portfolio), contain groups of columns entitled 'Variation of Data Per Period'.

For each input or output, these columns indicate the following:

- the minimum value per period,
- the maximum value per period,
- the arithmetic mean value per period,
- the coefficient of variation (i.e. standard deviation/arithmetic mean) for the data, and
- the statistical skew of the data.

Where insufficient data are available to calculate all of the statistics, the cells concerned are left blank. If one datum value only has been obtained, all of these cells are left blank. The 'period' which these data refer to, is the monthly company period. Since the data cover one year, the statistics contain at least 12 elements. However, in some cases that data set is composed of many more elements than this. To overcome possible bias which could be caused by there being differing numbers of days in the periods (because there are differing numbers of days in the months), the statistics have been calculated from normalised periods, all of which have a similar number of days.

To try to ensure the effects of rounding errors are minimised, data from original spreadsheet sources are used wherever possible, and are copied and pasted directly into calculations. In many cases, however, it has not been possible to obtain such direct data, and for some calculations, direct copying and pasting is impractical.

9.6.4.3. Uncertainty analysis using manual simulations

Two steps to uncertainty assessment in LCA may be identified. These are:

- dominance analysis, followed by
- sensitivity analysis.

Dominance analysis identifies
- firstly, the main impact categories contributing to the 'valuation' result,
- then, the most important contributing inventory flows giving rise to these impacts, and
- finally, the processes from which these burdens arise.

'Valuation' is identified by Consoli et al. (1993) as the final stage in impact assessment, and involves the relative importance of each of the different potential impacts being weighted against each other.[44] Valuation is normative in nature, since the weighting values are influenced by socio-economic rather than purely scientific considerations. It is a stage which will be applied, as and when the need arises, by those who choose to use the database. The aim of the assessments shown here is to illustrate the assessment capabilities of the database only. This stage has, therefore, not been included in this thesis. However, the need to develop effective and fair means of valuation, within an integrated strategy, is crucially import if the lead and associated industries are meet the environmental challenge

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[44] Valuation has been discussed further in the elective EngD module assignment, which was enclosed as Document 15, in volume 2 of the Portfolio. This document has not been supplied with this copy of the thesis.

with optimal effectiveness. This question is considered further in the Overall Conclusions and Recommendations (chapter 11).

Dominance analysis has been conducted for the inventories and effects, which have been graphed, for both the allocated and unallocated models. The results of these analyses are indicated in volume 5 of the Portfolio (in Document 7 for the inventories and in Document 8 for the potential effects). Sensitivity analyses have been conducted for flows, from all processes (whether they are in the foreground or background), which have been shown by the dominance analysis to contribute 5% or more to any of the overall system inventories or effects. The sensitivity analyses have been conducted by altering each of the flows in question separately, by plus one standard deviation from the actual value, and then calculating its effects on the inventories and effects considered in the graphs.

This method of analysis is possible, because all of the inputs and outputs associated with the foreground processes have been treated as variables in the TEAM^{TM} modelling. This means that any, or all, of them may be changed using externally driven control panels. The complete list of these variables was enclosed in volume 5 of the Portfolio (in Document 5). The list indicates, that for each variable there are three columns under the heading 'statistics' and two columns under the heading 'test values'. The columns under the former are labelled 'actual mean', coefficient of variation', and 'standard deviation', whilst under the heading 'test values' they are labelled 'minimum' and 'maximum'. The standard deviation and the 'minimum' and 'maximum' suggested test values, have been calculated from the mean and CoV values, in all cases. The 'minimum' and 'maximum' test values, are the mean minus and plus one standard deviation respectively. Where there have been sufficient data to calculate them, the 'coefficient of variation' (CoV) values are those cited in the 'Variation of Data per Period' columns in the input/output sheets for each process. However, where there are insufficient data for this, the CoV values have been estimated. Therefore, where insufficient data are available to derive the actual standard deviations, estimated standard deviations, using estimated CoV values, have been calculated instead.

The criteria for deriving the estimated CoV values are indicated in volume 5 of the Portfolio (in Document 17). Where estimated values have been used, the cells in the CoV column in the variables list (in Document 5 of volume 5 of the Portfolio) have been shaded.

It is the 'maximum' test values, which have been used in the sensitivity analysis for testing the key flow variables identified by dominance analysis. These key flows, and their influence on the inventories and potential effects, are indicated and assessed in the sensitivity analysis sheets for the inventories and for the effects, which are enclosed in volume 5 of the Portfolio (in Documents 7 and 8 respectively).

Interpretation of the inventory and potential effect graphs (in section 9.6.4.5), is conducted in conjunction with these sensitivity analyses, where appropriate.

9.6.4.4. Uncertainty analysis using Monte Carlo simulations

Monte Carlo simulations have been conducted to assess the relative ranges in variability of all of the inventory values. Whilst these simulations are needed for fully comprehensive assessments, the aim within this thesis is simply to illustrate some of the inventories and potential effects of interest. For the interpretation of such graphs, uncertainty analysis using Monte Carlo simulations is not necessary. Therefore, the findings from the Monte Carlo simulations, which have been conducted, have not been included within this thesis. However, they have been included in volume 5 the Portfolio (in Documents 19 and 20), and will be supplied to BRM, who have sponsored this research.

9.6.4.5. Discussion of the inventory and assessment graphs

The graphs, enclosed in section 9.7, have been categorised in sections 9.6.3.2.3 and 9.6.3.3.2.4 respectively. They identify two sets of graphs:

1. Inventory and effect graphs, which are for the whole BRM site and for the modelled year (a) only.
2. Inventory and effect comparison graphs, which compare BRM secondary operations, between the BRM company years 1998-9 and 1997-8.

Due to space restrictions on the graphs, the operations are identified by codes, in all cases. A legend at the bottom of each graph indicates which operations are depicted. The codes have been taken directly from the sheets entitled: 'BRM Whole Site: TEAM^{TM} Model Structure' in the Portfolio (which have been enclosed in this thesis as Document A1). The model is hierarchical, since the lower levels (indicated by the longer operations codes) are subsumed by the higher levels (the shorter codes). The codes also indicate the pathway by which they are subsumed. Hence, for example, operation: (a)/X1,2,1 is subsumed within operation: (a)/X1,2. This is itself subsumed within operation: (a)/X1, and, in turn, this is subsumed within operation: (a)/X. The highest level within the whole modelled system is operation: (a)/-, which is the system node: 'BRM: Lead Material Processing Operations at Northfleet, Kent, UK'. This represents all of the lead material processing operations (plus modelled associated operations) at the BRM site.

The examination of the graphs is not intended to be exhaustive. Instead, it aims to point out the salient features, as it is these which are expected to be of most use in decision making. Therefore, where individual graph bars are not discussed, it is assumed they are not significant. However, in the future, should BRM, or any other interested party, wish to examine them, sufficient data have been supplied in the Portfolio to facilitate this.

Finally, the following should be borne in mind by those reading the discussion which follows:

1. In all cases, the names used for the graphed inventories and effects are identical to those used in the modelling, which has been conducted using the TEAM^{TM} software. Where inventory flows and atoms are in the foreground, they have been named by the author, and follow the conventions recommended by the software suppliers. However, for atoms in the background, and for inventory flows associated with them, the names are entirely those provided with the software. In addition, the TEAM^{TM} software does not support the use of either superscripts or subscripts for the naming of flows and atoms. For example the inventory flow for carbon dioxide (CO_2) emissions to the atmosphere, arising from fossil fuel combustion, has been quoted as 'Carbon Dioxide (CO2, fossil)'. Also, many of the charges on aqueous ions are named in a non-standard manner in the DEAM^{TM} modules (which are the main source of background data). For example, emissions of aqueous cadmium (Cd) ions (i.e. Cd^{2+} ions), have been quoted as '(w)

Cadmium (Cd++)'. In consequence, for many chemical compounds, the names used to identify them, do not comply fully with the conventional methodology.

2. The TEAM^{TM} software calculates inventory and effect values to several decimal places. This is reflected in the discussion below. Therefore, the number of decimal places indicated in the discussion, are related to the output from the modelling, and do not indicate any particular level of accuracy.

9.6.4.5.1 Inventory graphs (Graphs 9.1 to 9.8)

For each inventory, the graphs have been divided into four linked series. In each the series, the bars on the left side of the plot area represent the higher levels. Differing shading has been used to group the various levels being shown.

a) <u>(r) Natural Gas (in ground)</u>: (Graph 9.1)

Graph 9.1 (Series 1) suggests, that primary refined lead production operations make a greater contribution to consumption of '(r) Natural Gas (in ground)' than secondary refined lead production operations. The Series 1 graph also suggests that for primary operations, the consumption by the BRM stream throughput and the Isa stream throughput are similar, whilst the Series 4 graph suggests that the principal contribution for secondary operations is from smelting, rotary furnace, refining and moulding operations, rather than from CX plant operations. Finally, the Series 2 and 3 graphs show, respectively, that the major consumption, for primary operations, in both the BRM stream throughput and the Isa stream throughput is by refining operations, rather than by any of the ancillary operations shown.

Dominance analysis indicates that consumption, at the whole system level, of '(r) Natural Gas (in ground)' may be attributed principally to the following atoms:

1. '403 Natural Gas: Combustion.1' which is within the system node: 'BRM: Lead Refining Operations (BRM floor throughput)'.

2. 403 Natural Gas: Combustion.2' which is within the system node: 'BRM: Lead Refining Operations (BRM floor throughput)'.
3. '403 Natural Gas: Combustion.1' which is within the system node: 'BRM: Lead Refining Operations (Isa stream)'.
4. '403 Natural Gas: Combustion.1' which is within the system node: 'BRM: On-site Electricity Generation Operations (Isa floor throughput)'.
5. '403 Natural Gas: Combustion.1' which is within the system node: 'BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations'.

Hence, at the whole system level, contributions to consumption of '(r) Natural Gas (in ground)' are attributable principally to natural gas combustion atoms. All other sources account for less than 5 % individually, of the total system consumption. In consequence, the overall system and individual consumptions of '(r) Natural Gas (in ground)' are expected to be relatively robust to potential inaccuracies and biases in any of the other individual atoms within the modelled system.

However, these atoms represent background processes. Since the flows in and out of the DEAM^{TM} modules (used to construct them) are not variables, flows into from them, cannot be modified on their own. They have been modified, therefore, by altering the linked flows in the foreground atomic nodes, to which they are connected. The linked foreground flows, and the paths by which they are linked, are indicated in volume 5 of the Portfolio (in Document 7).

For primary and secondary lead production operations, sensitivity analyses performed on the dominant contributors indicate changes of 2.7938 kg, 1.9921 kg, 2.5488 kg, 2.153 kg, and 0.8946 kg[45]. (Data from Document 7, of volume 5 of the Portfolio.) If it is assumed the likely potential deviation of the flow from its true value, is plus or minus one standard deviation, then the predicted potential variation ranges are: 2 x 2.7938 = 5.5876 kg, 2 x 1.9921 = 3.9842 kg, 2 x 2.5488 = 5.0976 kg, 2 x 2.153 = 4.306 kg, and 2 x 0.8946 = 1.7892 kg respectively. The difference in consumption between primary and secondary lead production operations is 73.29 - 16.61 = 56.69 kg. (Data from Document 1, of volume 5 of the Portfolio.) Since, this difference is greater than the predicted potential variation

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[45] Throughout the discussion, the number of decimal places reflects the number calculated by the TEAM^{TM} software. It does not suggest a particular level of accuracy.

ranges associated with the dominant contributors, when they are considered both individually and combined, it may be concluded, the assertion that there is a significant difference between the two operations is robust.

With regard to primary lead production operations, the difference between the BRM stream throughput and the Isa stream throughput is: 37.77 - 35.52 = 2.25 kg. The predicted potential variation ranges attributable to the dominant contributors are: 2 x 2.7938 = 5.5876 kg, 2 x 1.9921 = 3.9842 kg, 2 x 2.5488 = 5.0976 kg, and 2 x 2.153 = 4.306 kg. Therefore, the difference in consumption between the BRM and Isa streams is less than the predicted potential variation ranges associated with the dominant contributors, not only when they are combined, but is also less than all but one of the individual ranges. Hence, it may be concluded, the assertion that there is no significant difference between the two streams, seems most likely to be the case.

With regard to secondary lead production operations, the difference between CX plant operations and smelting, rotary furnace, refining and moulding operations is: 12.92 - 3.494 = 9.426 kg, whilst the predicted potential variation range attributable to the dominant contributor is: 2 x 0.8946 = 1.7892 kg. Since the difference between the operations is greater than the predicted potential variation range associated with the dominant contributor, it may be concluded, the assertion that there is a significant difference between the two operations is robust.

b) <u>(r) Oil (in ground)</u>: (Graph 9.2)

Graph 9.2 (Series 1) suggests, that secondary refined lead production operations make a greater contribution to consumption of '(r) Oil (in ground)' than secondary refined lead production operations. The Series 1 graph also suggests that for primary operations, the consumption by the BRM stream throughput is slightly less than that of the Isa stream throughput, whilst the Series 4 graph suggests the principal contribution for secondary operations is from smelting, rotary furnace, refining and moulding operations, rather than from CX plant operations. Finally, the Series 2 and 3 graphs show, respectively, that the major consumption, for primary operations, in both the BRM stream throughput and the Isa stream throughput, is by refining operations rather than by any of the ancillary operations shown.

Dominance analysis indicates that consumption, at the whole system level, of '(r) Oil (in ground)' may be attributed principally to the following atoms:

1. '403 Natural Gas: Combustion.1' which is within the system node: 'BRM: Lead Refining Operations (BRM floor throughput)'.
2. '403 Heavy Fuel Oil: Combustion.1' which connects to 'BRM: [Tr.] Heavy Fuel Oil'. Then, as 'Heavy Fuel Oil (used as fuel)' it flows into 'BRM: (secondary): Smelting, Rotary Furnace, Refining and Moulding'. Both are within the system node: 'BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations'.
3. '403 Heavy Fuel Oil: Combustion.1' which connects to 'BRM: [Tr.] Heavy Fuel Oil', then as 'Heavy Fuel Oil (used as fuel)1' it flows into 'BRM: (secondary): Smelting, Rotary Furnace, Refining and Moulding'. Both are within the system node: 'BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations'

Hence, at the whole system level, contributions to consumption of '(r) Natural Gas (in ground)' are attributable principally to natural gas and heavy fuel oil combustion atoms. All other sources account for less than 5 % individually, of the total system consumption. In consequence, the overall system and individual consumptions of '(r) Oil (in ground)' are expected to be relatively robust to potential inaccuracies and biases in any of the other individual atoms within the modelled system.

However, these atoms represent background processes. Since the flows in and out of the DEAM^{TM} modules (used to construct them) are not variables, flows into from them, cannot be modified on their own. They have been modified, therefore, by altering the linked flows in the foreground atomic nodes, to which they are connected. The linked foreground flows, and the paths by which they are linked, are indicated in Document 7 (of volume 5 of the Portfolio).

For primary and secondary lead production operations, sensitivity analyses performed on the dominant contributors, indicate changes of 0.01970 kg, 3.3324 kg, and 1.1114 kg[46]. (Data from Document 7, of volume 5 of the Portfolio.) If it is assumed the likely potential deviation of the flow from its true value, is plus or minus one standard deviation, then the predicted potential variation ranges are: 2 x 0.01970 = 0.0394 kg, 2 x 3.3324 = 6.6648 kg, and 2 x 1.1114 = 2.2228 kg respectively. However, the difference between the two operations is 24.56 - 5.657 = 18.903 kg. (Data from Document 1, of volume 5 of the Portfolio.) Since this difference is greater than the predicted potential variation ranges associated with the dominant contributors, when they are considered both individually and combined, it may be concluded, the assertion that there is a significant difference between the two operations is robust.

With regard to primary lead production operations, the difference between the BRM stream throughput and the Isa stream throughput is: 3.238 - 2.419 = 0.819 kg. The predicted potential variation range attributable to the dominant contributor is: 2 x 0.01970 = 0.0394 kg. Since the difference in consumption between the BRM and Isa streams is greater than the predicted potential variation range associated with the dominant contributor, it is concluded, that although the difference between the two streams appears to be relatively small, it is robust.

With regard to secondary lead production operations, the difference between CX plant operations and smelting, rotary furnace, refining and moulding operations is: 23.88 - 0.6664 = 23.214 kg, whilst the predicted potential variation ranges attributable to the dominant contributors are: 2 x 3.3324 = 6.6648 kg, and 2 x 1.1114 = 2.2228 kg. Since the difference between the two operations, both individually and combined, is greater than the predicted potential variation ranges associated with the dominant contributor, it may be concluded, the assertion that there is a significant difference between the two operations is robust.

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[46] Throughout the discussion, the number of decimal places reflects the number calculated by the TEAM™ software. It does not suggest a particular level of accuracy.

c) <u>(a) Cadmium (Cd)</u>: (Graph 9.3)

Graph 9.3 (Series 1) suggests, that secondary refined lead production operations make a greater contribution to emissions to the atmosphere of '(a) Cadmium (Cd)' than secondary refined lead production operations. The Series 1 graph also suggests that for primary operations, the consumption by the BRM stream throughput is slightly less than that of the Isa stream throughput, whilst the Series 4 graph suggests the principal contribution for secondary operations is from smelting, rotary furnace, refining and moulding operations, rather than from CX plant operations (whose contribution appears to be trivial). Finally, the Series 2 and 3 graphs show, respectively, that the major emission contribution, for primary operations, in both the BRM stream throughput and the Isa stream throughput, is by refining operations rather than by any of the ancillary operations shown.

Dominance analysis indicates that the dominating contributor to emissions to the atmosphere, at the whole system level, of '(a) Cadmium (Cd)' is the atom: '274 Zinc (Zn): Production.1' which is within the system node 'BRM: Lead Refining Operations (Isa stream)'. All other sources account for less than 5 % individually, of the total system emissions. In consequence, the overall system and individual emissions to the atmosphere of '(a) Cadmium (Cd)' are expected to be relatively robust to potential inaccuracies and biases in any of the other individual atoms within the modelled system.

The '274 Zinc (Zn): Production.1' atoms represents a background process. Since the flows in and out of the DEAM^{TM} module (used to construct it) are not variables, flows into from it, cannot be modified on their own. They have been modified, therefore, by altering the linked flow in the foreground atomic nodes, to which it is connected. The linked foreground flow, and the path by which it is linked, is indicated in Document 7 (of volume 5 of the Portfolio).

For primary and secondary lead production operations, sensitivity analysis performed on the dominant contributor, indicates a change of 0.02737 g[47]. (Data from Document 7, of volume 5 of the Portfolio.) If it is assumed the likely potential deviation of the flow from its true value is plus or minus one standard deviation, then the predicted potential variation

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[47] Throughout the discussion, the number of decimal places reflects the number calculated by the TEAM^{TM} software. It does not suggest a particular level of accuracy.

range is: 2 x 0.02737 = 0.05474 g. However, the difference between the two operations is: 0.5857 - 0.07774 = 0.50796 g. (Data from Document 1, of volume 5 of the Portfolio.) Since this difference is greater than the predicted potential variation range associated with the dominant contributor, it may be concluded, the assertion that there is a significant difference between the two operations is robust.

With regard to primary lead production operations, the difference between the BRM stream throughput and the Isa stream throughput is: 0.05577 - 0.02197 = 0.0338 g. The predicted potential variation range attributable to the dominant contributor is: 2 x 0.02737 = 0.05474 g. Since the difference in consumption between the BRM and Isa streams is less than the predicted potential variation range associated with the dominant contributor, it is concluded, that due to the uncertainty in the data, it is not possible to demonstrate a significant difference between the two streams.

With regard to secondary lead production operations, the difference between CX plant operations and smelting, rotary furnace, refining and moulding operations appears to be so great, it is assumed that the relative contribution of the latter is trivial, and that this assertion is robust.

d) <u>(a) Carbon Dioxide (CO2, fossil)</u>: (Graph 9.4)

Graph 9.4 (Series 1) suggests, that primary refined lead production operations make a greater contribution to emissions to the atmosphere of '(a) Carbon Dioxide (CO2, fossil)' than secondary refined lead production operations. The Series 1 graph also suggests that for primary operations, the consumption by the BRM stream throughput is slightly less than the Isa stream throughput, whilst the Series 4 graph suggests that the principal contribution for secondary operations is from smelting, rotary furnace, refining and moulding operations, rather than from the CX plant. Finally, the Series 2 and 3 graphs show, respectively, that the major emission contribution, for primary operations, in both the BRM stream throughput and the Isa stream throughput is by refining operations, rather than by any of the ancillary operations shown.

Dominance analysis indicates that emissions to the atmosphere, at the whole system level, of '(a) Carbon Dioxide (CO_2, fossil)' may be attributed principally to the following atoms:

1. '403 Heavy Fuel Oil: Combustion.1' which connects to 'BRM: [Tr.] Heavy Fuel Oil', then as 'Heavy Fuel Oil (used as fuel)1' it flows into 'BRM: (secondary): Smelting, Rotary Furnace, Refining and Moulding'. Both are within the system node: 'BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations'
2. '241 Water (Softened): Production.1' which is within the system node: 'BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations'.
3. '401 Electricity (United Kingdom, 1996): Production.1)' which is within the system node: 'BRM: Lead Refining Operations (Isa stream)'.

All other sources account for less than 5 % individually, of the total system consumption. In consequence, the overall system and individual emissions to the atmosphere of '(a) Carbon Dioxide (CO_2, fossil)' are expected to be relatively robust to potential inaccuracies and biases in any of the other individual atoms within the modelled system.

However, all three atoms represent background processes. Since the flows in and out of the DEAM^{TM} modules (used to construct them) are not variables, flows into from them, cannot be modified on their own. They have been modified, therefore, by altering the linked flows in the foreground atomic nodes, to which they are connected. The linked foreground flows, and the paths by which they are linked, are indicated in Document 7 (of volume 5 of the Portfolio).

For primary and secondary lead production operations, sensitivity analyses performed on the dominant contributors indicate changes of 10,954 g, 4,457 g, and 51,949 g. (Data from Document 7, of volume 5 of the Portfolio.) If it is assumed the likely potential deviation of the flow from its true value, is plus or minus one standard deviation, then the predicted potential variation ranges are: 2 x 10,954 = 21,908 g, 2 x 4,457 = 8,914 g, and 2 x 51,949 = 103,898 g respectively. The difference in consumption between primary and secondary lead production operations is 274,200 - 177,400 = 96,800 g. (Data from Document 1, of volume 5 of the Portfolio.) This difference in consumption between primary and secondary lead production operations is less than the predicted potential variation ranges associated with the dominant contributors, when they are combined, and is also less than the range attributable to one of the individual contributors. It is concluded, therefore, that due to the uncertainty in the data, it is not possible to ascertain whether or not a significant difference exists between the two operations.

With regard to primary lead production operations, the difference between the BRM stream throughput and the Isa stream throughput is: 149,200 - 125,000 = 24,200 g, whilst the predicted potential variation range attributable to the dominant contributor is: 2 x 51,949 = 103,898 g. Since, the difference in consumption between the BRM and Isa streams is also less than the predicted potential variation ranges associated with the dominant contributor, it may be concluded, the assertion that there is no significant difference between the two streams, seems most likely to be the case.

With regard to secondary lead production operations, the difference between CX plant operations and smelting, rotary furnace, refining and moulding operations is: 142,500 - 33,400 = 109,100 g, whilst the predicted potential variation ranges attributable to the dominant contributor are: 2 x 10,954 = 21,908 g, and 2 x 4,457 = 8,914 g. Since the difference between the operations is greater than the predicted potential variation ranges associated with the dominant contributors, when considered both individually and combined, it may be concluded, the assertion that there is a significant difference between the two operations is robust.

e) <u>(a) Lead (Pb)</u>: (Graph 9.5)

Graph 9.5 (Series 1) suggests, that primary and secondary refined lead production operations make a similar contribution to emissions to the atmosphere of '(a) Lead (Pb)'. The Series 1 graph also suggests that for primary operations, the consumption by the BRM stream throughput is less than the Isa stream throughput, whilst the Series 4 graph suggests the principal contribution for secondary operations is from smelting, rotary furnace, refining and moulding operations, rather than from CX plant operations. Finally, the Series 2 and 3 graphs show, respectively, that the major emission contribution, for primary operations, in both the BRM stream throughput and the Isa stream throughput is by refining operations, rather than by any of the ancillary operations shown (whose contributions appears to be trivial).

Dominance analysis indicates that emissions to the atmosphere, at the whole system level, of '(a) Lead (Pb)' may be attributed principally to the following atoms:

1. '274 Zinc (Zn): Production.1' which connects to 'BRM: [Tr.] Zinc', and then as 'Zinc (Zn)' it flows into 'BRM: Lead Refining (BRM floor throughput)'. All are within the system node: 'BRM: Lead Refining Operations (BRM floor throughput)'.
2. '274 Zinc (Zn): Production.1' which connects to 'BRM: [Tr.] Zinc', and then as 'Zinc (Zn)' it flows into 'BRM: Lead Refining (Isa stream)'. All are within the system node: 'BRM: Lead Refining Operations (Isa stream)'.

All other sources account for less than 5 % individually, of the total system consumption. In consequence, the overall system and individual emissions to the atmosphere of '(a) Lead (Pb)' are expected to be relatively robust to potential inaccuracies and biases in any of the other individual atoms within the modelled system.

However, all three atoms represent background processes. Since the flows in and out of the DEAM^{TM} modules (used to construct them) are not variables, flows into from them, cannot be modified on their own. They have been modified, therefore, by altering the linked flows in the foreground atomic nodes, to which they are connected. The linked foreground flows, and the paths by which they are linked, are indicated in Document 7 (of volume 5 of the Portfolio).

For primary and secondary lead production operations, sensitivity analyses performed on the dominant contributors indicate changes of 1.20240 g, and 1.01230 g[48]. (Data from Document 7, of volume 5 of the Portfolio.) If it is assumed the likely potential deviation of the flow from its true value, is plus or minus one standard deviation, then the predicted potential variation ranges are: 2 x 1.20240 = 2.40480 g, and 2 x 1.01230 = 2.0246 g respectively. The difference in consumption between primary and secondary lead production operations is 7.852 - 7.527 = 0.325 g. (Data from Document 1, of volume 5 of the Portfolio.) Since, this difference is less than the predicted potential variation ranges associated with the dominant contributors, when they are considered both individually and

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[48] Throughout the discussion, the number of decimal places reflects the number calculated by the TEAM^{TM} software. It does not suggest a particular level of accuracy.

combined, it may be concluded, the assertion that there is a significant difference between the two operations appears likely.

With regard to primary lead production operations, the difference between the BRM stream throughput and the Isa stream throughput is: 4.376 - 3.151 = 1.225 g, whilst the predicted potential variation ranges attributable to the dominant contributors are: 2 x 1.20240 = 2.40480 g, and 2 x 1.01230 = 2.0246 g. Therefore, the difference in consumption between the BRM and Isa streams is less than the predicted potential variation ranges associated with the dominant contributors, when considered both individually and combined. Hence, it may be concluded, that due to the uncertainty in the data, it is not possible to demonstrate a significant difference between the two streams.

With regard to secondary lead production operations, the difference between CX plant operations and smelting, rotary furnace, refining and moulding operations appears to be so great, it is assumed that the relative contribution of the latter is trivial, and that this assertion is robust.

f) (a) Sulphur Dioxide (SOx, as SO2)[49]: (Graph 9.6)

Graph 9.6 (Series 1) suggests, that emissions to the atmosphere of '(a) Sulphur Oxides (SOx, as SO2)' are dominated by secondary refined lead production operations. Primary refined lead production operations appear to make only a trivial contribution. The Series 4 graph suggests that the principal contribution for secondary operations is almost entirely from smelting, rotary furnace, refining and moulding operations. CX plant operations appear to make only a trivial contribution. Finally, the Series 2 and 3 graphs show, respectively, that contributions from primary operations, in both the BRM stream and the Isa stream throughput, are dominated by refining operations. The ancillary operations shown appear to make only a trivial contribution.

The difference between the dominant and the apparently trivial operations are so great, it is assumed that all of the above assertions are robust.

g) <u>(w) Cadmium (Cd++)</u>: (Graph 9.7)

Graph 9.7 (Series 1) suggests, that primary refined lead production operations make a smaller contribution to emissions to water of '(w) Cadmium (Cd++)' than secondary refined lead production operations. The Series 1 graph also suggests that for primary operations, the emission contribution by the BRM stream throughput is slightly less than the Isa stream throughput, whilst the Series 4 graph suggests that the principal contribution for secondary operations is from CX plant operations, rather than from smelting, rotary furnace, refining and moulding operations. Finally, the Series 2 and 3 graphs show, respectively, that the major emission contribution, for primary operations, in both the BRM stream throughput and the Isa stream throughput is by refining operations, rather than by any of the ancillary operations shown.

Dominance analysis indicates that emissions to the atmosphere, at the whole system level, of '(w) Cadmium (Cd++)' may be attributed principally to the following atoms:

1. '403 Heavy Fuel Oil: Combustion.1' which connects to 'BRM: [Tr.] Heavy Fuel Oil', then as 'Heavy Fuel Oil (used as fuel)' it flows into 'BRM: (secondary): Smelting, Rotary Furnace, Refining and Moulding'. Both are within the system node: 'BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations'.
2. Heavy Fuel Oil: Combustion.1' which connects to 'BRM: [Tr.] Heavy Fuel Oil', then as 'Heavy Fuel Oil (used as fuel)1' it flows into 'BRM: (secondary): Smelting, Rotary Furnace, Refining and Moulding'. Both are within the system node: 'BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations'.

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[49] Throughout the discussion, the names of the inventory flows, and atoms, are those supplied the TEAM™ software. In consequence, names for many chemical compounds do not comply fully with conventional methodology. For further information, see section 9.6.4.5 of this chapter.

All other sources account for less than 5 % individually, of the total system consumption. In consequence, the overall system and individual emissions to the atmosphere of '(w) Cadmium (Cd++)' are expected to be relatively robust to potential inaccuracies and biases in any of the other individual atoms within the modelled system.

However, all three atoms represent background processes. Since the flows in and out of the DEAM™ modules (used to construct them) are not variables, flows into from them, cannot be modified on their own. They have been modified, therefore, by altering the linked flows in the foreground atomic nodes, to which they are connected. The linked foreground flows, and the paths by which they are linked, are indicated in Document 7 (of volume 5 of the Portfolio).

For primary and secondary lead production operations, sensitivity analyses performed on the dominant contributors indicate changes of 0.00011374 g, and 0.00003794 g[50]. (Data from Document 7, of volume 5 of the Portfolio.) If it is assumed the likely potential deviation of the flow from its true value, is plus or minus one standard deviation, then the predicted potential variation ranges are: 2 x 0.00011374 = 0.0002275 g, and 2 x 0.00003794 = 0.00007588 g respectively. The difference in consumption between primary and secondary lead production operations is 0.2991 - 0.04305 = 0.25605 g. (Data from Document 1, of volume 5 of the Portfolio.) Since, this difference is greater than the predicted potential variation ranges associated with the dominant contributors, when they are considered both individually and combined, it may be concluded, the assertion that there is a significant difference between the two operations is robust.

With regard to primary lead production operations, the difference between the BRM stream throughput and the Isa stream throughput is 0.02651 - 0.01654 = 0.00997 g. Since there are no dominant contributors associated with the BRM and Isa stream, the difference between them is believed to be robust. However, since the difference is so small, further research is warranted to confirm whether or not this is actually the case.

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[50] Throughout the discussion, the number of decimal places reflects the number calculated by the TEAM™ software. It does not suggest a particular level of accuracy.

With regard to secondary lead production operations, the difference between CX plant operations and smelting, rotary furnace, refining and moulding operations is 0.0001701 - 0.0008603 = 0.0008407 g, whilst the predicted potential variation ranges attributable to the dominant contributors are: 2 x 0.00011374 = 0.0002275 g, and 2 x 0.00003794 = 0.00007588 g. Since the difference between the operations is greater than the predicted potential variation ranges associated with the dominant contributors, when considered both individually and combined, it may be concluded, the assertion that there is a significant difference between the two operations is robust.

h) <u>(w) Lead (Pb++, Pb4+)</u>[51]: (Graph 9.8)

Graph 9.8 (Series 1) suggests, that primary refined lead production operations make a smaller contribution to emissions to water of '(w) Lead (Pb++, Pb4+)' than secondary refined lead production operations. The Series 1 graph also suggests that for primary operations, the emission contribution by the BRM stream throughput is slightly less than the Isa stream throughput, whilst the Series 4 graph suggests that the principal contribution for secondary operations is from CX plant operations, rather than from smelting, rotary furnace, refining and moulding operations. Finally, the Series 2 and 3 graphs show, respectively, that the major emission contribution, for primary operations, in both the BRM stream throughput and the Isa stream throughput is by refining operations, rather than by any of the ancillary operations shown.

Since no individual contributors account for greater less than 5 % individually, of the total system emissions, the overall system and individual emissions water of '(w) Lead (Pb++, Pb4+)' are expected to be relatively robust to potential inaccuracies and biases in any of the individual atoms within the modelled system. Therefore, the above observed differences between the operations are considered to be robust. However, since the difference between BRM and Isa stream primary lead refining operations is small, further research is warranted to confirm whether or not this is actually the case.

9.6.4.5.2 Effect graphs (Graphs 9.9 to 9.22)

The bars on the left hand side of the plot area of each graph represent the higher levels. Differing shading has been used to group the various levels being shown.

a) <u>CML -Air Acidification</u>: (Graph 9.9)

Graph 9.9 suggests that, secondary refined lead production operations make a greater contribution to the effect than primary refined lead production operations. It also suggests that, for primary lead production operations, the BRM stream and Isa stream throughputs make approximately equal contributions.

Dominance analysis indicates that contributions to the effect, at the whole system level, may attributed principally to the atomic node 'BRM (Secondary): Smelting, Rotary Furnace, Refining and Moulding' which is within the system node: 'BRM (Secondary): Smelting, Rotary Furnace, Refining and Moulding Operations' (all of whom are within the system node: 'BRM: Secondary Lead Production Operations'). All other sources account for less than 5 %, individually, of the total system effect. In consequence, the overall system effect, and individual contributions to the effect, of 'CML -Air Acidification' are expected to be relatively robust to potential inaccuracies and biases in any of the other individual atoms within the modelled system.

Furthermore, dominance analysis indicates that 86.81 % of the effect overall may be attributed to emissions to the atmosphere of '(a) Sulphur Oxides (SOx, as SO2)'[52], and that emissions to the atmosphere of '(a) Sulphur Oxides (SOx, as SO2)' from the atomic node 'BRM (Secondary): Smelting, Rotary Furnace, Refining and Moulding' account for 77.90 % of the effect overall. Emissions to the atmosphere of '(a) Nitrogen Oxides (NOx, as NO2)' also account for 12.09 % of the effect overall, though no individual contributions from the emissions account for 5 % or more of the total effect.

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[51] Throughout the discussion, the names of the inventory flows, and atoms, are those supplied the TEAM™ software. In consequence, names for many chemical compounds do not comply fully with conventional methodology. For further information, see section 9.6.4.5 of this chapter.

[52] Throughout the discussion, the names of the inventory flows, and atoms, are those supplied the TEAM™ software. In consequence, names for many chemical compounds do not comply fully with conventional methodology. For further information, see section 9.6.4.5 of this chapter.

Individual contributions from atomic nodes, other than 'BRM (Secondary): Smelting, Rotary Furnace, Refining and Moulding', are all less than 5 %. In consequence, the overall system and individual contributions to the effect 'CML -Air Acidification' are expected to be relatively robust to potential inaccuracies and biases in any of these other individual atoms within the modelled system.

The atomic node 'BRM (Secondary): Smelting, Rotary Furnace, Refining and Moulding' is in the foreground. Emissions of '(a) Sulphur Oxides (SOx, as SO2)' from it are calculated from data from an on-line SO2 monitor on the BRM main stack. They are, therefore, expected to be reasonably accurate. Hence, sensitivity analysis has not been necessary to conclude that secondary refined lead production operations make a greater contribution to the effect than primary refined lead production operations. For primary operations, however, since the difference between BRM and Isa stream is so small, it is viewed, currently, that it is not possible to ascertain whether or not there is a significant difference. Further research would be needed to answer this question.

b) <u>CML -Aquatic Ecotoxicity</u>: (Graph 9.10)

Graph 9.10 suggests that, secondary refined lead production operations make a greater contribution to the effect than primary refined lead production operations. It also suggests that, for primary lead production operations, the BRM stream and Isa stream throughputs make approximately equal contributions.

Dominance analysis indicates that contributions to the effect, at the whole system level, may attributed principally to the atomic nodes:

1. '403 Natural Gas: Combustion.1' which is within the system node: 'BRM: Lead Refining Operations (BRM floor throughput)'.
2. 403 Natural Gas: Combustion.2' which is within the system node: 'BRM: Lead Refining Operations (BRM floor throughput)'.
3. '403 Natural Gas: Combustion.1' which is within the system node: 'BRM: Lead Refining Operations (BRM floor throughput)'.

4. ‘403 Heavy Fuel Oil: Combustion.1’ which connects to ‘BRM: [Tr.] Heavy Fuel Oil’. Then, as ‘Heavy Fuel Oil (used as fuel)’ it flows into ‘BRM: (secondary): Smelting, Rotary Furnace, Refining and Moulding’. Both are within the system node: ‘BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations’.

Sensitivity analyses have been conducted for these dominating atoms, as indicated in Document 8 (of volume 5 of the Portfolio).

All other sources account for less than 5 %, individually, of the total system effect. In consequence, the overall system effect, and individual contributions to the effect, of ‘CML -Aquatic Ecotoxicity’ are expected to be relatively robust to potential inaccuracies and biases in any of the other individual atoms within the modelled system.

All of the dominant atoms (listed above) represent background processes. Since the flows in and out of the DEAM™ modules (used to construct them) are not variables, flows from them which contribute to the effect, cannot be modified on their own. They have been modified, therefore, by altering the linked flows in the foreground atomic nodes, to which they are connected. The linked foreground flows, and the paths by which they are linked, are indicated in Document 8 (of volume 5 of the Portfolio).

Dominance analysis indicates, furthermore, that the effect overall may be attributed principally to the following emissions to the atmosphere:

‘(w) Cadmium (Cd++)’[53]	59.16 %
‘(w) Oils (unspecified)	32.51 %
‘(w) Chromium (Cr III)	6.01 %

For primary and secondary lead production operations, sensitivity analyses performed on the dominant contributors, indicate changes in effect of 7.3699 m^3, 6.72460 m^3, 35.29 m^3, and 11.77 m^3 [54]. (Data from Document 8, of volume 5 of the Portfolio.) If it is assumed the

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[53] Throughout the discussion, the names of the inventory flows, and atoms, are those supplied the TEAM™ software. In consequence, names for many chemical compounds do not comply fully with conventional methodology. For further information, see section 9.6.4.5 of this chapter.

[54] Throughout the discussion, the number of decimal places reflects the number calculated by the TEAM™ software. It does not suggest a particular level of accuracy.

likely potential deviation of the flow from its true value, is plus or minus one standard deviation, then the predicted potential variation ranges are: 2 x 7.3699 = 14.7398 m^3, 2 x 6.72460 = 13.4492 m^3, 2 x 35.29 = 70.58 m^3, and 2 x 11.77 = 23.54 m^3 respectively. However, the difference between the two operations is 654.6272 - 314.0794 = 340.5478 m^3. (Data from Document 2, of volume 5 of the Portfolio.) Since this difference is greater than the predicted potential variation ranges associated with the dominant contributors, when they are considered both individually and combined, it may be concluded, the assertion that there is a significant difference between the two operations is robust.

With regard to primary lead production operations, the difference between the BRM stream throughput and the Isa stream throughput is: 165.3786 - 148.7010 = 16.6776 m^3 . The predicted potential variation ranges attributable to the dominant contributors are: 2 x 7.3699 = 14.7398 m^3, and 2 x 6.72460 = 13.4492 m^3. Therefore, the difference in consumption between the BRM and Isa streams is less than the predicted potential variation ranges associated with the dominant contributors, when they are combined. Hence, it may be concluded, the assertion that there is no significant difference between the two streams, seems most likely to be the case.

c) CML -Depletion of non-renewable resources: (Graph 9.11)

Graph 9.11 suggests, that primary refined lead production operations make a greater contribution to the effect than primary refined lead production operations. It also suggests that, for primary lead production operations, the Isa stream throughput makes a substantially greater contribution. The contribution of the BRM stream operations appears to be relatively trivial.

Dominance analysis indicates that contributions to the effect, at the whole system level, may attributed principally to the atomic node: '274 Tin (Sn): Production.1' within the system node: 'BRM: Lead Refining Operations (Isa stream). A sensitivity analysis has been conducted for this dominating atom, as indicated in Document 8 (of volume 5 of the Portfolio). All other sources account for less than 5 %, individually, of the total system effect. In consequence, the overall system effect, and individual contributions to the effect are expected to be relatively robust to potential inaccuracies and biases in any of the other individual atoms within the modelled system.

The '274 Tin (Sn): Production.1' atom represents a background processes. Since the flows in and out of the DEAM™ module (used to construct it) are not variables, flows from it which contribute to the effect, cannot be modified on their own. They have been modified, therefore, by altering the linked flows in the foreground atomic nodes, to which they are connected. The linked foreground flows, and the path by which they are linked, are indicated in Document 8 (of volume 5 of the Portfolio).

Dominance analysis indicates, furthermore, that the effect overall may be attributed principally to the following consumptions:

'(r) Tin (Sn, ore)'	94.00 %
'(r) Zinc (Zn, ore)'	4.89 %

For primary and secondary lead production operations, sensitivity analyses performed on the dominant contributor, indicates a change in the effect of 2.4992E-11 frac. of reserve[55]. (Data from Document 8, of volume 5 of the Portfolio.) If it is assumed the likely potential deviation of the flow from its true value, is plus or minus one standard deviation, then the predicted potential variation range is: 2 x 2.4992E-11 = 4.9984E-11 frac. of reserve. However, the difference between the two operations is 9.09E-11 - 3.59E-12 = 8.731E-11 frac. of reserve. (Data from Document 2, of volume 5 of the Portfolio.) Since this difference is greater than the predicted potential variation ranges associated with the dominant contributor, it may be concluded, the assertion that there is a significant difference between the two operations is robust.

With regard to primary lead production operations, the difference between the BRM stream throughput and the Isa stream throughput is: 8.70E-11 - 3.85E-12 = 8.315E-11frac. of reserve. The predicted potential variation range attributable to the dominant contributor is: 2 x 2.4992E-11 = 4.9984E-11 frac. of reserve. Therefore, the difference in consumption between the BRM and Isa streams is greater than the predicted potential variation range

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[55] Throughout the discussion, the number of decimal places reflects the number calculated by the TEAM™ software. It does not suggest a particular level of accuracy.

associated with the dominant contributor. Hence, it may be concluded, the assertion that there is a significant difference between the two streams is robust

d) <u>CML -Eutrophication</u>: (Graph 9.12)

Graph 9.12 suggests that, primary refined lead production operations make a greater contribution to the effect than secondary refined lead production operations. It also suggests that, for primary lead production operations, the BRM stream and Isa stream throughputs make approximately equal contributions.

Dominance analysis indicates that contributions to the effect, at the whole system level, may attributed principally to the atomic nodes:

1. '403 Heavy Fuel Oil: Combustion.1' which connects to 'BRM: [Tr.] Heavy Fuel Oil'. Then, as 'Heavy Fuel Oil (used as fuel)' it flows into 'BRM: (secondary): Smelting, Rotary Furnace, Refining and Moulding'. Both are within the system node: 'BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations'.
2. 'BRM: Lead Refining (BRM floor throughput)' which is within the system node: 'BRM: Lead Refining Operations (BRM floor throughput)'.
3. 'BRM: Lead Refining (Isa stream)' which is within the system node: 'BRM: Lead Refining Operations (Isa stream)'.

Sensitivity analyses have been conducted for these dominating atoms, as indicated in Document 8 (of volume 5 of the Portfolio).

All other atoms account for less than 5 %, individually, of the total system effect. In consequence, the overall system effect, and individual contributions to the effect are expected to be relatively robust to potential inaccuracies and biases in any of the other individual atoms within the modelled system.

Dominance analysis indicates, furthermore, that 99.14 % of the effect overall may be attributed to emissions to the atmosphere of '(a) Nitrogen Oxides (NOx, as NO2)'[56].

The 'BRM: Lead Refining (BRM floor throughput)' node and 'BRM: Lead Refining (Isa stream)' node both represent foreground processes. In the sensitivity analyses for these atoms, the flow of '(a) Nitrogen Oxides (NOx, as NO2)', is manipulated in isolation of all of other flows. However, the other dominant atom (i.e. '403 Heavy Fuel Oil: Combustion.1') represents a background process. Since the flows in and out of the DEAM™ module (used to construct this background process) are not variables, flows from it which contribute to the effect, cannot be modified on their own. They have been modified, therefore, by altering the linked flows in the foreground atomic node, to which they are connected. The linked foreground flows, and the path by which they are linked, are indicated in Document 8 (of volume 5 of the Portfolio).

For primary and secondary lead production operations, sensitivity analyses performed on the dominant contributor, indicate changes in effect of 3.02640 g eq. PO_4, 163.1500 g eq. PO_4, and 162.4506 g eq. PO_4 [57]. (Data from Document 8, of volume 5 of the Portfolio). If it is assumed the likely potential deviation of the flow from its true value, is plus or minus one standard deviation, then the predicted potential variation ranges are: 2 x 3.02640 = 6.0528 g eq. PO_4, 2 x 163.1500 = 326.300 g eq. PO_4, and 2 x 162.4506 = 324.9012 g eq. PO_4 respectively. However, the difference between the two operations is 418.02544 - 43.9422 = 374.08324 g eq. PO_4. (Data from Document 2, of volume 5 of the Portfolio.) This difference is less than the predicted potential variation ranges associated with the dominant contributors when they are considered in combination. However, it is greater than them when they are considered individually. It is concluded, therefore, that whilst there may be a significant difference between the operations, the uncertainties in the data are such, that it has not been possible to demonstrate this unequivocally.

With regard to primary lead production operations, the difference between the BRM stream throughput and the Isa stream throughput is: 210.103 - 207.9224 = 2.1806 g eq. PO_4. The

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[56] Throughout the discussion, the names of the inventory flows, and atoms, are those supplied the TEAM™ software. In consequence, names for many chemical compounds do not comply fully with conventional methodology. For further information, see section 9.6.4.5 of this chapter.

[57] Throughout the discussion, the number of decimal places reflects the number calculated by the TEAM™ software. It does not suggest a particular level of accuracy.

predicted potential variation ranges attributable to the dominant contributors are: 2 x 163.1500 = 326.300 g eq. PO_4, and 2 x 162.4506 = 324.9012 g eq. PO_4. Therefore, the difference in consumption between the BRM and Isa streams is less than the predicted potential variation ranges associated with the dominant contributors, when they are considered both individually and in combination. Hence, it may be concluded, the assertion that there is no significant difference between the two streams is highly probable.

e) CML -Eutrophication (water): (Graph 9.13)

Graph 9.13 suggests that, primary and secondary refined lead production operations make a similar contributions to the effect. It also suggests that, for primary lead production operations, the Isa stream throughput makes a greater contribution to the effect than the BRM stream throughput.

Dominance analysis indicates that contributions to the effect, at the whole system level, may attributed principally to the atomic nodes:

1. '403 Heavy Fuel Oil: Combustion.1' which connects to 'BRM: [Tr.] Heavy Fuel Oil'. Then, as 'Heavy Fuel Oil (used as fuel)' it flows into 'BRM: (secondary): Smelting, Rotary Furnace, Refining and Moulding'. Both are within the system node: 'BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations'.
2. '274 Tin (Sn): Production.1' which connects to: 'BRM: Lead Refining Operations (Isa stream). Both are within the system node: 'BRM: Lead Refining Operations (BRM floor throughput).
3. '274 Tin (Sn): Production.1' which connects to: 'BRM: Lead Refining Operations (Isa stream). Both are within the system node: 'BRM: Lead Refining Operations (Isa stream).
4. '211 Pulp (Sulphate, Unbleached): Production.1' which connects to: 'BRM: Lead Refining Operations (Isa stream). Both are within the system node: 'BRM: Lead Refining Operations (Isa stream).

Sensitivity analyses have been conducted for these dominating atoms, as indicated in Document 8 (of volume 5 of the Portfolio).

All other sources account for less than 5 %, individually, of the total system effect. In consequence, the overall system effect, and individual contributions to the effect are expected to be relatively robust to potential inaccuracies and biases in any of the other individual atoms within the modelled system.

All of the dominant atoms (listed above) represent background processes. Since the flows in and out of the DEAM^{TM} modules (used to construct them) are not variables, flows from them which contribute to the effect, cannot be modified on their own. They have been modified, therefore, by altering the linked flows in the foreground atomic nodes, to which they are connected. The linked foreground flows, and the paths by which they are linked, are indicated in Document 8 (of volume 5 of the Portfolio).

Dominance analysis indicates, furthermore, that the effect overall may be attributed principally to the following emissions to the atmosphere:

'(w) Phosphates (PO4 3-, HPO4, as P)'[58]	31.15 %
'(w) Ammonia (NH4+, NH3, as N)'	29.65 %
'(w) Nitrogenous Matter (unspecified, as N)'	27.41 %
'(w) COD (Chemical Oxygen Demand)'	6.65 %

For primary and secondary lead production operations, sensitivity analyses performed on the dominant contributors, indicate changes in effect of 0.04587 g eq PO_4, 0.00001348 g eq. PO_4, 0.0003686 g eq. PO_4, and 0.1096 g eq. PO_4 [59]. (Data from Document 8, of volume 5 of the Portfolio.). If it is assumed the likely potential deviation of the effect from its true value, is plus or minus one standard deviation, then the predicted potential variation ranges are: 2 x 0.04587 = 0.09174 g eq. PO_4, 2 x 0.00001348 = 0.00002696 g eq. PO_4, 2 x 0.0003686 = 0.0007372 g eq. PO_4, and 2 x 0.1096 = 0.2192 g eq. PO_4 respectively. However, the difference between the two operations is 1.3647 - 1.1890 = 0.1757 g eq. PO_4. (Data from Document 2, of volume 5 of the Portfolio.) This difference is less than the

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[58] Throughout the discussion, the names of the inventory flows, and atoms, are those supplied the TEAM^{TM} software. In consequence, names for many chemical compounds do not comply fully with conventional methodology. For further information, see section 9.6.4.5 of this chapter.

[59] Throughout the discussion, the number of decimal places reflects the number calculated by the TEAM^{TM} software. It does not suggest a particular level of accuracy.

predicted potential variation ranges associated with the dominant contributors, when they are considered in combination and is also less than one of them individually. Hence, it may be concluded, no significant difference between the two operations has been demonstrated.

With regard to primary lead production operations, the difference between the BRM stream throughput and the Isa stream throughput is: 0.8678 - 0.4969 = 0.3709 g eq. PO_4. The predicted potential variation ranges attributable to the dominant contributors are: 2 x 0.00001348 = 0.00002696 g eq. PO_4, 2 x 0.0003686 = 0.0007372 g eq. PO_4, and 2 x 0.1096 = 0.2192 g eq. PO_4. Since this difference is greater than the predicted potential variation ranges associated with the dominant contributor, it may be concluded, the assertion that there is a significant difference between the two streams is robust.

f) <u>CML -Human Toxicity</u>: (Graph 9.14)

Graph 9.14 suggests that, secondary refined lead production operations make a greater contribution to the effect than primary refined lead production operations. It also suggests that, for primary lead production operations, the Isa stream throughput makes a slightly contribution to the effect than the BRM stream throughput.

Dominance analysis indicates that contributions to the effect, at the whole system level, may attributed principally to the atomic node: '403 Heavy Fuel Oil: Combustion.1' (which connects to 'BRM: [Tr.] Heavy Fuel Oil'. Then, as 'Heavy Fuel Oil (used as fuel)' it flows into 'BRM: (secondary): Smelting, Rotary Furnace, Refining and Moulding'), and is within the system node: 'BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations'. Sensitivity analyses have been conducted for this atom, as indicated Document 8 (of volume 5 of the Portfolio). All other sources account for less than 5 %, individually, of the total system effect. In consequence, the overall system effect, and individual contributions to the effect are expected to be relatively robust to potential inaccuracies and biases in any of the other individual atoms within the modelled system.

The '403 Heavy Fuel Oil: Combustion.1' atom represents a background processes. Since the flows in and out of the DEAM^{TM} module (used to construct it) are not variables, flows from it which contribute to the effect, cannot be modified on their own. They have been modified, therefore, by altering the linked flows in the foreground atomic node, to which it is connected. The linked foreground flow, and the path by which it is linked, is indicated in Document 8 (of volume 5 of the Portfolio).

Dominance analysis indicates, furthermore, that the effect overall may be attributed principally to the following emissions to the atmosphere:

'(a) Sulphur Oxides (SOx, as SO2)'[60]	74.59 %
'(a) Nitrogen Oxides (NOx, as NO2)'	9.70 %
'(a) Lead (Pb)'	8.69 %
'(a) Arsenic (As)'	3.30 %

For primary and secondary lead production operations, sensitivity analysis performed on the dominant contributors, indicates a change in the effect of 213.5289 g[61]. (Data from Document 8, of volume 5 of the Portfolio.) If it is assumed the likely potential deviation of the effect from its true value, is plus or minus one standard deviation, then the predicted potential variation range is: 2 x 213.5289 = 427.0578 g. However, the difference between the two operations is 23,204.4656 - 5,114.6488 = 18,089.8168 g. (Data from Document 2, of volume 5 of the Portfolio.) Since this difference is greater than the predicted potential variation range associated with the dominant contributor, it may be concluded, the assertion that there is a significant difference between the two operations is robust.

With regard to primary lead production operations, the difference between the BRM stream throughput and the Isa stream throughput is 2,841.2385 - 2,273.41523 = 567.8233 g. The predicted potential variation ranges attributable to the dominant contributor is: 2 x 213.5289 = 427.0578 g. Therefore, the difference in consumption between the BRM and Isa streams is greater than the predicted potential variation ranges associated with the dominant contributor. Hence, it may be concluded, that it is probable there is a small but significant difference between the two streams.

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[60] Throughout the discussion, the names of the inventory flows, and atoms, are those supplied the TEAM™ software. In consequence, names for many chemical compounds do not comply fully with conventional methodology. For further information, see section 9.6.4.5 of this chapter.

[61] Throughout the discussion, the number of decimal places reflects the number calculated by the TEAM™ software. It does not suggest a particular level of accuracy.

g) CML -Terrestrial Ecotoxicity: (Graph 9.15)

Graph 9.15 suggests that, primary refined lead production operations make a greater contribution to the effect than secondary refined lead production operations. It also suggests that, for primary lead production operations, the BRM stream and Isa stream throughputs make approximately equal contributions.

Dominance analysis indicates that contributions to the effect, at the whole system level, may attributed principally to the atomic nodes:

1. '403 Natural Gas: Combustion.1' which is within the system node: 'BRM: Lead Refining Operations (BRM floor throughput)'.
2. '403 Natural Gas: Combustion.1' which is within the system node: 'BRM: Lead Refining Operations (Isa stream)'.
3. 'Charcoal: Production' which is within the system node: 'BRM: Lead Refining Operations (BRM floor throughput)'.
4. 403 Natural Gas: Combustion.1' which is within the system node: 'BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations'.
5. '403 Natural Gas: Combustion.1' which is connected to: 'BRM: [Tr.] Natural Gas' and then 'BRM: On-site Electricity Generation (Isa floor throughput)'. All are within the system node: 'BRM: Lead Refining Operations (Isa stream)'.

Sensitivity analyses have been conducted for these dominating atoms, as indicated in Document 8 (of volume 5 of the Portfolio).

All other sources account for less than 5 %, individually, of the total system effect. In consequence, the overall system effect, and individual contributions to the effect are expected to be relatively robust to potential inaccuracies and biases in any of the other individual atoms within the modelled system.

All of the dominant atoms (listed above) represent background processes. Since the flows in and out of the DEAM™ and other modules (used to construct them) are not variables, flows from them which contribute to the effect, cannot be modified on their own. They have been modified, therefore, by altering the linked flows in the foreground atomic nodes, to which they are connected. The linked foreground flows, and the paths by which they are linked, are indicated in Document 8 (of volume 5 of the Portfolio).

Dominance analysis indicates, furthermore, that the effect overall may be attributed principally to the following emissions to the ground of:

'(s) Zinc (Zn)'	87.34 %
'(s) Chromium (Cr III, Cr VI)'	4.69 %
'(s) Arsenic (As)'	3.21 %

For primary and secondary lead production operations, sensitivity analyses performed on the dominant contributors, indicate changes in effect of 0.002463 t, 0.002247 t, 0.001758 t, 0.0007886 t, and 0.0018977 t[62]. (Data from Document 8, of volume 5 of the Portfolio.) If it is assumed the likely potential deviation of the effect from its true value, is plus or minus one standard deviation, then the predicted potential variation ranges are: 2 x 0.002463 = 0.004926 t, 2 x 0.002247 = 0.004494 t, 2 x 0.001758 = 0.003516 t, 2 x 0.0007886 = 0.0015772 t and 2 x 0.018977 =0.0037954 t respectively. However, the difference between the two operations is 0.06190 - 0.001172 = 0.05018 t. (Data from Document 2, of volume 5 of the Portfolio.) Since this difference is greater than the predicted potential variation ranges associated with the dominant contributors, when they are considered both individually and combined, it may be concluded, the assertion that there is a significant difference between the two operations is robust.

With regard to primary lead production operations, the difference between the BRM stream throughput and the Isa stream throughput is: 0.03132 - 0.03058 = 0.00074 t. The predicted potential variation ranges attributable to the dominant contributors are2 x 0.002463 = 0.004926 t, 2 x 0.002247 = 0.004494 t, 2 x 0.001758 = 0.003516 t, and 2 x 0.0018977 =0.0037954 t. Therefore, the difference in consumption between the BRM and Isa streams is less than the predicted potential variation ranges associated with the dominant contributors, when they are combined and also for most of them when considered

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[62] Throughout the discussion, the number of decimal places reflects the number calculated by the TEAM™ software. It does not suggest a particular level of accuracy.

individually. Hence, it may be concluded, it is highly probable that there is no significant difference between the two streams.

h) IPCC -Greenhouse effect (direct, 20 years): (Graph 9.16)

Graph 9.16 suggests that, primary refined lead production operations make a greater contribution to the effect than secondary refined lead production operations. It also suggests that, for primary lead production operations, the contribution to the effect from the Isa stream throughput is slightly greater than that from the BRM stream throughput.

Dominance analysis indicates that contributions to the effect, at the whole system level, may attributed principally to the atomic nodes:

1. '403 Natural Gas: Combustion.1' which is within the system node: 'BRM: Lead Refining Operations (BRM floor throughput)'.
2. '403 Natural Gas: Combustion.1' which is within the system node: 'BRM: Lead Refining Operations (Isa stream)'.
3. '403 Heavy Fuel Oil: Combustion.1' which connects to 'BRM: [Tr.] Heavy Fuel Oil'. Then, as 'Heavy Fuel Oil (used as fuel)' it flows into 'BRM: (secondary): Smelting, Rotary Furnace, Refining and Moulding'. Both are within the system node: 'BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations'.
4. '401 Electricity (United Kingdom, 1996): Production.1' which is within the system node: 'BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations'.
5. '241 Water (Softened): Production.1' which is within the system node: 'BRM: Lead Refining Operations (Isa stream)'.

Sensitivity analyses have been conducted for these dominating atoms, as indicated in Document 8 (of volume 5 of the Portfolio).

All other sources account for less than 5 %, individually, of the total system effect. In consequence, the overall system effect, and individual contributions to the effect are expected to be relatively robust to potential inaccuracies and biases in any of the other individual atoms within the modelled system.

All of the dominant atoms (listed above) represent background processes. Since the flows in and out of the DEAM^{TM} modules (used to construct them) are not variables, flows from them which contribute to the effect, cannot be modified on their own. They have been modified, therefore, by altering the linked flows in the foreground atomic nodes, to which they are connected. The linked foreground flows, and the paths by which they are linked, are indicated in Document 8 (of volume 5 of the Portfolio).

Dominance analysis indicates, furthermore, that the effect overall may be attributed principally to the following emissions to the atmosphere:

'(a) Carbon Dioxide (CO2, fossil)'[63]	84.02 %
'(a) Methane (CH4)'	15.70 %

For primary and secondary lead production operations, sensitivity analyses performed on the dominant contributors, indicate changes in effect of: 7,614.2400 g eq. CO_2, 6,947.2400 g eq. $CO_{2,}$, 17,201.9200 g eq. CO_2, 62,047.4800 g eq. CO_2, and 5,496.4400 g eq. CO_2 [64]. (Data from Document 8, of volume 5 og the Portfolio.) If it is assumed the likely potential deviation of the effect from its true value, is plus or minus one standard deviation, then the predicted potential variation ranges are: 2 x 7,614.2400 = 15,228.48 g eq. CO_2, 2 x 6,947.2400 = 13,894.48 g eq. CO_2, 2 x 17,201.9300 = 34,403.84 g eq. CO_2, 2 x 62,047.4800 = 124,094.96 g eq. CO_2, and 2 x 5,496.4400 = 10,992.88 g eq. CO_2 respectively. However, the difference between the two operations is 300,250.9579 - 237,328.6417 = 62,922.3162 g eq. CO_2. (Data from Document 2, of volume 5 of the Portfolio.) This difference is less than the predicted potential variation ranges associated with the dominant contributors, when they are considered in combination and is also less than one of them individually. Hence, it may be concluded, no significant difference between the two operations has been demonstrated. This does not mean, necessarily, there

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[63] Throughout the discussion, the names of the inventory flows, and atoms, are those supplied the TEAM^{TM} software. In consequence, names for many chemical compounds do not comply fully with conventional methodology. For further information, see section 9.6.4.5 of this chapter.

[64] Throughout the discussion, the number of decimal places reflects the number calculated by the TEAM^{TM} software. It does not suggest a particular level of accuracy.

is no significant difference. Rather, it is simply that due to the uncertainties in the modelling data, it has not been possible to demonstrate such a difference unequivocally.

With regard to primary lead production operations, the difference between the BRM stream throughput and the Isa stream throughput is: 162,465.8449 - 137,786.1144 = 24,679.7305 g eq. CO_2. The predicted potential variation ranges attributable to the dominant contributors are: 2 x 7,614.2400 = 15,228.48 g eq. CO_2, 2 x 6,947.2400 = 13,894.48 g eq. CO_2, and 2 x 5,496.4400 = 10,992.88 g eq. CO_2. This difference is less than the predicted potential variation ranges associated with the dominant contributors, when they are considered in combination. Hence, it may be concluded, no significant difference between the two operations has been demonstrated. This does not mean, necessarily, there is no significant difference. Rather, it is simply that due to the uncertainties in the modelling data, it has not been possible to demonstrate such a difference unequivocally.

Finally, it should be noted, the relative sizes of the bars, and findings from the sensitivity analyses, are similar to those for both the 'IPCC -Greenhouse effect (direct, 100 years)' and 'IPCC -Greenhouse effect (Direct, 500 years)'. Emissions to the atmosphere of '(a) Carbon Dioxide (CO2, fossil)' account for 84.02 % of this effect, 93.13 % of the 'IPCC - Greenhouse effect (Direct, 100 years)', and 97.68 % of the 'IPCC -Greenhouse effect (Direct, 500 years)'. Therefore, it is apparent, that the contributions of '(a) Methane (CH4)' (the principal contributor the remainder of the effect), make no significant difference to the findings.

i) <u>IPCC -Greenhouse effect (direct, 100 years)</u>: (Graph 9.17)

Graph 9.17 suggests that, primary refined lead production operations make a greater contribution to the effect than secondary refined lead production operations. It also suggests that, for primary lead production operations, the contribution to the effect from the Isa stream throughput is slightly greater than that from the BRM stream throughput.

Dominance analysis indicates that contributions to the effect, at the whole system level, may attributed principally to the atomic nodes:

1. '403 Natural Gas: Combustion.1' which is within the system node: 'BRM: Lead Refining Operations (BRM floor throughput)'.
2. '403 Natural Gas: Combustion.1' which is within the system node: 'BRM: Lead Refining Operations (Isa stream)'.
3. '403 Heavy Fuel Oil: Combustion.1' which connects to 'BRM: [Tr.] Heavy Fuel Oil'. Then, as 'Heavy Fuel Oil (used as fuel)' it flows into 'BRM: (secondary): Smelting, Rotary Furnace, Refining and Moulding'. Both are within the system node: 'BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations'.
4. '401 Electricity (United Kingdom, 1996): Production.1' which is within the system node: 'BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations'.
5. '241 Water (Softened): Production.1' which is within the system node: 'BRM: Lead Refining Operations (Isa stream)'.

Sensitivity analyses have been conducted for these dominating atoms, as indicated in Document 8 (of volume 5 of the Portfolio).

All other sources account for less than 5 %, individually, of the total system effect. In consequence, the overall system effect, and individual contributions to the effect are expected to be relatively robust to potential inaccuracies and biases in any of the other individual atoms within the modelled system.

All of the dominant atoms (listed above) represent background processes. Since the flows in and out of the DEAM^{TM} modules (used to construct them) are not variables, flows from them which contribute to the effect, cannot be modified on their own. They have been modified, therefore, by altering the linked flows in the foreground atomic nodes, to which they are connected. The linked foreground flows, and the paths by which they are linked, are indicated in Document 8 (of volume 5 of the Portfolio).

Dominance analysis indicates, furthermore, that the effect overall may be attributed principally to the following emissions to the atmosphere:

'(a) Carbon Dioxide (CO2, fossil)'[65] 93.13 %

'(a) Methane (CH4)' 6.53 %

For primary and secondary lead production operations, sensitivity analyses performed on the dominant contributors, indicate changes in effect of: 7,315.3400 g eq. CO_2, 6,674.5900 g eq. CO_2,, 13,296.9700 g eq. CO_2, 55,735.9300 g eq. CO_2, and 5,295.5400 g eq. CO_2 [66]. (Data from Document 8, of volume 5 of the Portfolio). If it is assumed the likely potential deviation of the effect from its true value, is plus or minus one standard deviation, then the predicted potential variation ranges are: 2 x 7,315.3400 = 14,630.68 g eq. CO_2, 2 x 6,674.5900 = 13,349.18 g eq. CO_2, 2 x 13,296.9700 = 26,593.94 g eq. CO_2, 2 x 55,735.9300 = 111,471.86 g eq. CO_2, and 2 x 5,295.5400 = 10,591.08 g eq. CO_2 respectively. However, the difference between the two operations is 284,440.9555 - 200,551.2334 = 83,889.7221 g eq. CO_2. (Data from Document 2, of volume 5 of the Portfolio.) This difference is less than the predicted potential variation ranges associated with the dominant contributors, when they are considered in combination and is also less than one of them individually. Hence, it may be concluded, no significant difference between the two operations has been demonstrated. This does not mean, necessarily, there is no significant difference. Rather, it is simply that due to the uncertainties in the modelling data, it has not been possible to demonstrate such a difference unequivocally.

With regard to primary lead production operations, the difference between the BRM stream throughput and the Isa stream throughput is: 154,421.6527 - 130,020.3045 = 24,401.3482 g eq. CO_2. The predicted potential variation ranges attributable to the dominant contributors are: 2 x 7,315.3400 = 14,630.68 g eq. CO_2, 2 x 6,674.5900 = 13,349.18 g eq. CO_2, and 2 x 5,295.5400 = 10,591.08 g eq. CO_2. This difference is less than the predicted potential variation ranges associated with the dominant contributors, when they are considered in combination. Hence, it may be concluded, no significant difference between the two operations has been demonstrated. This does not mean, necessarily, there is no significant

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[65] Throughout the discussion, the names of the inventory flows, and atoms, are those supplied the TEAM™ software. In consequence, names for many chemical compounds do not comply fully with conventional methodology. For further information, see section 9.6.4.5 of this chapter.

[66] Throughout the discussion, the number of decimal places reflects the number calculated by the TEAM™ software. It does not suggest a particular level of accuracy.

difference. Rather, it is simply that due to the uncertainties in the modelling data, it has not been possible to demonstrate such a difference unequivocally.

Finally, it should be noted, the relative sizes of the bars, and findings from the sensitivity analyses, are similar to those for both the 'IPCC -Greenhouse effect (direct, 100 years)' and 'IPCC -Greenhouse effect (Direct, 500 years)'. Emissions to the atmosphere of '(a) Carbon Dioxide (CO2, fossil)' account for 93.13 % of this effect, 84.02 % of the 'IPCC -Greenhouse effect (Direct, 20 years)', and 97.68 % of the 'IPCC -Greenhouse effect (Direct, 500 years)'. Therefore, it is apparent, that the contributions of '(a) Methane (CH4)' (the principal contributor the remainder of the effect), make no significant difference to the findings.

j) <u>IPCC -Greenhouse effect (direct, 500 years)</u>: (Graph 9.18)

Graph 9.18 suggests that, primary refined lead production operations make a greater contribution to the effect than secondary refined lead production operations. It also suggests that, for primary lead production operations, the contribution to the effect from the Isa stream throughput is slightly greater than that from the BRM stream throughput.

Dominance analysis indicates that contributions to the effect, at the whole system level, may attributed principally to the atomic nodes:

1. '403 Natural Gas: Combustion.1' which is within the system node: 'BRM: Lead Refining Operations (BRM floor throughput)'.
2. '403 Natural Gas: Combustion.1' which is within the system node: 'BRM: Lead Refining Operations (Isa stream)'.
3. '403 Heavy Fuel Oil: Combustion.1' which connects to 'BRM: [Tr.] Heavy Fuel Oil'. Then, as 'Heavy Fuel Oil (used as fuel)' it flows into 'BRM: (secondary): Smelting, Rotary Furnace, Refining and Moulding'. Both are within the system node: 'BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations'.
4. '401 Electricity (United Kingdom, 1996): Production.1' which is within the system node: 'BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations'.

5. '241 Water (Softened): Production.1' which is within the system node: 'BRM: Lead Refining Operations (Isa stream)'.

Sensitivity analyses have been conducted for these dominating atoms, as indicated in Document 8 (of volume 5 of the Portfolio).

All other sources account for less than 5 %, individually, of the total system effect. In consequence, the overall system effect, and individual contributions to the effect are expected to be relatively robust to potential inaccuracies and biases in any of the other individual atoms within the modelled system.

All of the dominant atoms (listed above) represent background processes. Since the flows in and out of the DEAM^{TM} modules (used to construct them) are not variables, flows from them which contribute to the effect, cannot be modified on their own. They have been modified, therefore, by altering the linked flows in the foreground atomic nodes, to which they are connected. The linked foreground flows, and the paths by which they are linked, are indicated in Document 8 (of volume 5 of the Portfolio).

Dominance analysis indicates, furthermore, that the effect overall may be attributed principally to the following emissions to the atmosphere:

'(a) Carbon Dioxide (CO2, fossil)'[67]	97.68 %
'(a) Methane (CH4)'	2.12 %

For primary and secondary lead production operations, sensitivity analyses performed on the dominant contributors, indicate changes in effect of: 7,191.5100 g eq. CO_2, 6,561.6350 g eq. CO_2,, 11,679.2050 g eq. CO_2, 53,121.1450 g eq. CO_2, and 5,212.3100 g eq. CO_2 [68]. (Data from Document 8, of volume 5 of the Portfolio.) If it is assumed the likely potential deviation of the effect from its true value, is plus or minus one standard deviation, then the

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[67] Throughout the discussion, the names of the inventory flows, and atoms, are those supplied the TEAM^{TM} software. In consequence, names for many chemical compounds do not comply fully with conventional methodology. For further information, see section 9.6.4.5 of this chapter.

[68] Throughout the discussion, the number of decimal places reflects the number calculated by the TEAM^{TM} software. It does not suggest a particular level of accuracy.

predicted potential variation ranges are: 2 x 7,191.5100 = 14,383.02 g eq. CO_2, 2 x 6,561.6350 = 13,123.27 g eq. CO_2, 2 x 11,679.2050 = 23,358.41 g eq. CO_2, 2 x 53,121.1450 = 106,242.29 g eq. CO_2, and 2 x 5,212.3100 = 10,424.62 g eq. CO_2 respectively. However, the difference between the two operations is 277,572.3126 - 184,824.4622 = 92,747.8504 g eq. CO_2. (Data from Document 2, of volume 5 of the Portfolio.) Since this difference is greater than the predicted potential variation ranges associated with the dominant contributors, when they are considered both individually and combined, it may be concluded, the assertion that there is a significant difference between the two operations is robust.

With regard to primary lead production operations, the difference between the BRM stream throughput and the Isa stream throughput is: 150,915.5255 - 126,657.788 = 24,257.7375 g eq. CO_2. The predicted potential variation ranges attributable to the dominant contributors are: 2 x 7,191.5100 = 14,383.02 g eq. CO_2, 2 x 6,561.6350 = 13,123.27 g eq. CO_2, and 2 x 5,212.3100 = 10,424.62 g eq. CO_2. Therefore, the difference in consumption between the BRM and Isa streams is less than the predicted potential variation ranges associated with the dominant contributors, when they are combined. Hence, it may be concluded, the assertion that there is no significant difference between the two streams, seems most likely to be the case.

Finally, it should be noted, the relative sizes of the bars, and findings from the sensitivity analyses, are similar to those for both the 'IPCC -Greenhouse effect (direct, 20 years)' and 'IPCC -Greenhouse effect (Direct, 100 years)'. Emissions to the atmosphere of '(a) Carbon Dioxide (CO2, fossil)' account for 97.68 % of this effect, 93.13 % of the 'IPCC - Greenhouse effect (Direct, 100 years)', and 84.02 % of the 'IPCC -Greenhouse effect (Direct, 20 years)'. Therefore, it is apparent, that the contributions of '(a) Methane (CH4)' (the principal contributor the remainder of the effect), make no significant difference to the findings.

k) <u>WMO -Depletion of Ozone Layer (high)</u>: (Graph 9.19)

Graph 9.19 suggests that, secondary refined lead production operations make a greater contribution to the effect than primary refined lead production operations. It also suggests that, for primary lead production operations, the BRM stream and Isa stream throughputs make approximately equal contributions.

Dominance analysis indicates that contributions to the effect, at the whole system level, may attributed principally to the atomic nodes:

1. '403 Heavy Fuel Oil: Combustion.1' which connects to 'BRM: [Tr.] Heavy Fuel Oil'. Then, as 'Heavy Fuel Oil (used as fuel)' it flows into 'BRM: (secondary): Smelting, Rotary Furnace, Refining and Moulding'. Both are within the system node: 'BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations'.
2. '403 Heavy Fuel Oil: Combustion.1' which connects to 'BRM: [Tr.] Heavy Fuel Oil'. Then, as 'Heavy Fuel Oil (used as fuel)1' it flows into 'BRM: (secondary): Smelting, Rotary Furnace, Refining and Moulding'. Both are within the system node: 'BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations'.

Sensitivity analyses have been conducted for these dominating atoms, as indicated in Document 8 (of volume 5 of the Portfolio).

All other sources account for less than 5 %, individually, of the total system effect. In consequence, the overall system effect, and individual contributions to the effect are expected to be relatively robust to potential inaccuracies and biases in any of the other individual atoms within the modelled system.

Both of the dominant atoms (listed above) represent background processes. Since the flows in and out of the DEAM^{TM} modules (used to construct them) are not variables, flows from them which contribute to the effect, cannot be modified on their own. They have been modified, therefore, by altering the linked flows in the foreground atomic nodes, to which they are connected. The linked foreground flows, and the paths by which they are linked, are indicated in Document 8 (of volume 5 of the Portfolio).

Dominance analysis indicates, furthermore, that the effect overall may be attributed entirely to emissions to the atmosphere of '(a) Halon 1301 (CF3Br)'[69].

For primary and secondary lead production operations, sensitivity analyses performed on the dominant contributors, indicate changes in effect of: 0.01167 g eq. CFC-11, and 0.003891 g eq. CFC-11[70]. (Data from Document 8, of volume 5 of the Portfolio.) If it is assumed the likely potential deviation of the effect from its true value, is plus or minus one standard deviation, then the predicted potential variation ranges are: 2 x 0.01197 = 0.02334 g eq. CFC-11, and 2 x 0.003891 = 0.007782 g eq. CFC-11 respectively. However, the difference between the two operations is 0.0844- 0.0155 = 0.0689 g eq. CFC-11. (Data from Document 2, of volume 5 of the Portfolio.) Since this difference is greater than the predicted potential variation ranges associated with the dominant contributors, when they are considered both individually and combined, it may be concluded, the assertion that there is a significant difference between the two operations is robust.

With regard to primary lead production operations, the difference between the BRM stream throughput and the Isa stream throughput is: 0.00820 - 0.00727 = 0.00093 g eq. CFC-11 . Since there are no dominating atoms for primary operations, the contributions from the Isa stream and BRM stream throughputs will be relatively robust to inaccuracies in their comprising data. It is considered likely, therefore, the finding that there is no significant difference between the two streams, really is the case.

l) <u>WMO -Depletion of Ozone Layer (low)</u>: (Graph 9.20)

Dominance analysis indicates, the effect is attributed entirely to emissions to the atmosphere of '(a) Halon 1301 (CF3Br)'[71]. This is also the case for the effect: 'WMO - Depletion of the Ozone Layer (high)'. Therefore, the relative sizes of the contribution bars

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[69] Throughout the discussion, the names of the inventory flows, and atoms, are those supplied the TEAM™ software. In consequence, names for many chemical compounds do not comply fully with conventional methodology. For further information, see section 9.6.4.5 of this chapter.

[70] Throughout the discussion, the number of decimal places reflects the number calculated by the TEAM™ software. It does not suggest a particular level of accuracy.

[71] Throughout the discussion, the names of the inventory flows, and atoms, are those supplied the TEAM™ software. In consequence, names for many chemical compounds do not comply fully with conventional methodology. For further information, see section 9.6.4.5 of this chapter.

in Graph 9.20 are identical to those in Graph 9.19 (which is for the latter effect). In consequence, all of the assertions and findings for that effect, are also applicable here.

m) WMO -Photochemical Oxidant Creation (high): (Graph 9.21)

Graph 9.21 suggests that, secondary refined lead production operations make a greater contribution to the effect than primary refined lead production operations. It also suggests that, for primary lead production operations, the Isa stream makes a greater contribution to the effect than the BRM floor throughput.

Dominance analysis indicates that contributions to the effect, at the whole system level, may attributed principally to the atomic nodes:

1. '403 Natural Gas: Combustion.1' which is within the system node: 'BRM: Lead Refining Operations (Isa stream)'.
2. '403 Heavy Fuel Oil: Combustion.1' which connects to 'BRM: [Tr.] Heavy Fuel Oil'. Then, as 'Heavy Fuel Oil (used as fuel)' it flows into 'BRM: (secondary): Smelting, Rotary Furnace, Refining and Moulding'. Both are within the system node: 'BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations'.
3. '403 Heavy Fuel Oil: Combustion.1' which connects to 'BRM: [Tr.] Heavy Fuel Oil'. Then, as 'Heavy Fuel Oil (used as fuel)1' it flows into 'BRM: (secondary): Smelting, Rotary Furnace, Refining and Moulding'. Both are within the system node: 'BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations'.
4. '241 Water (Softened): Production.1' which is within the system node: 'BRM: Lead Refining Operations (Isa stream)'.
5. '231 Coke: Production.1' which is within the system node: 'BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations'.

Sensitivity analyses have been conducted for these dominating atoms, as indicated in Document 8 (of volume 5 of the Portfolio).

All other sources account for less than 5 %, individually, of the total system effect. In consequence, the overall system effect, and individual contributions to the effect are expected to be relatively robust to potential inaccuracies and biases in any of the other individual atoms within the modelled system.

All of the dominant atoms (listed above) represent background processes. Since the flows in and out of the DEAM^{TM} modules (used to construct them) are not variables, flows from them which contribute to the effect, cannot be modified on their own. They have been modified, therefore, by altering the linked flows in the foreground atomic nodes, to which they are connected. The linked foreground flows, and the paths by which they are linked, are indicated in Document 8 (of volume 5 of the Portfolio).

Dominance analysis indicates, furthermore, that the effect overall may be attributed principally to the following emissions to the atmosphere:

'(a)Hydrocarbons (except methane)'[72]	69.26 %
'(a) Methane (CH4)'	8.43 %
'(a) Ethylene (C2H4)'	7.75 %
'(a) Hydrocarbons (unspecified)'	4.19 %
'(a) Propane (C3H8)'	2.84 %
'(a) Alkane (unspecified)'	2.57 %
'(a) Ethane (C2H6)'	2.55 %

For primary and secondary lead production operations, sensitivity analyses performed on the dominant contributors, indicate changes in effect of: 3.2260 g eq. ethylene, 24.3322 g eq. ethylene, 8.1165 g eq. ethylene, 7.05081 g eq. ethylene, and 16.7415 g eq. ethylene[73]. (Data from Document 8, of volume 5 of the Portfolio.) If it is assumed the likely potential deviation of the effect from its true value, is plus or minus one standard deviation, then the predicted potential variation ranges are: 2 x 3.2260 = 6.452 g eq. ethylene, 2 x 24.3322 = 48.6644 g eq. ethylene, 2 x 8.1165 = 16.233 g eq. ethylene, 2 x 7.05081 = 14.10162 g eq. ethylene, and 2 x 16.7415 = 33.483 g eq. ethylene respectively. However, the difference between the two operations is 338.4925- 197.6021 = 140.8904 g eq. ethylene. (Data from

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[72] Throughout the discussion, the names of the inventory flows, and atoms, are those supplied the TEAM^{TM} software. In consequence, names for many chemical compounds do not comply fully with conventional methodology. For further information, see section 9.6.4.5 of this chapter.

[73] Throughout the discussion, the number of decimal places reflects the number calculated by the TEAM^{TM} software. It does not suggest a particular level of accuracy.

Document 2, of volume 5 of the Portfolio.) Since this difference is greater than the predicted potential variation ranges associated with the dominant contributors, when they are considered both individually and combined, it may be concluded, the assertion that there is a significant difference between the two operations is robust.

With regard to primary lead production operations, the difference between the BRM stream throughput and the Isa stream throughput is 111.6256 - 85.9762 = 25.6494 g eq. ethylene. The predicted potential variation ranges attributable to the dominant contributors are: 2 x 3.2260 = 6.452 g eq. ethylene, and 2 x 7.05081 = 14.10162 g eq. ethylene. Since this difference is greater than the predicted potential variation ranges associated with the dominant contributors, when they are considered both individually and combined, it may be concluded, the assertion that there is a significant difference between the two operations is robust.

n) <u>WMO -Photochemical Oxidant Creation (low)</u>: (Graph 9.22)

Graph 9.22 suggests that, secondary refined lead production operations make a slightly greater contribution to the effect than primary refined lead production operations. It also suggests that, for primary lead production operations, the Isa stream throughput makes a slightly greater contribution than the BRM stream throughput.

Dominance analysis indicates that contributions to the effect, at the whole system level, may attributed principally to the atomic nodes:

1. '403 Natural Gas: Combustion.1' which is within the system node: 'BRM: Lead Refining Operations (BRM floor throughput)'.
2. '403 Natural Gas: Combustion.1' which is within the system node: 'BRM: Lead Refining Operations (Isa stream)'.
3. '403 Heavy Fuel Oil: Combustion.1' which connects to 'BRM: [Tr.] Heavy Fuel Oil'. Then, as 'Heavy Fuel Oil (used as fuel)' it flows into 'BRM: (secondary): Smelting, Rotary Furnace, Refining and Moulding'. Both are within the system node: 'BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations'.

4. '403 Heavy Fuel Oil: Combustion.1' which connects to 'BRM: [Tr.] Heavy Fuel Oil'. Then, as 'Heavy Fuel Oil (used as fuel)1' it flows into 'BRM: (secondary): Smelting, Rotary Furnace, Refining and Moulding'. Both are within the system node: 'BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations'.
5. '241 Water (Softened): Production.1' which is within the system node: 'BRM: Lead Refining Operations (Isa stream)'.
6. '231 Coke: Production.1' which is within the system node: 'BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations'.

Sensitivity analyses have been conducted for these dominating atoms, as indicated in Document 8 (of volume 5 of the Portfolio).

All other sources account for less than 5 %, individually, of the total system effect. In consequence, the overall system effect, and individual contributions to the effect are expected to be relatively robust to potential inaccuracies and biases in any of the other individual atoms within the modelled system.

All of the dominant atoms (listed above) represent background processes. Since the flows in and out of the DEAM^{TM} modules (used to construct them) are not variables, flows from them which contribute to the effect, cannot be modified on their own. They have been modified, therefore, by altering the linked flows in the foreground atomic nodes, to which they are connected. The linked foreground flows, and the paths by which they are linked, are indicated in Document 8 (of volume 5 of the Portfolio).

Dominance analysis indicates, furthermore, that the effect overall may be attributed principally to the following emissions to the atmosphere:

'(a)Hydrocarbons (except methane)'	62.77 %
'(a) Ethylene (C2H4)'[74]	28.76 %
'(a) Propane (C3H8)'	1.36 %
'(a) Alkane (unspecified)'	0.93 %
'(a) Butane (n-C4H10)'	0.70 %
'(a) Ethane (C2H6)'	0.63 %

For primary and secondary lead production operations, sensitivity analyses performed on the dominant contributors, indicate changes in effect of 1.7635 g eq. ethylene, 1.6088 g eq. ethylene, 4.8319 g eq. ethylene, 1.6118 g eq. ethylene, 1.6876 g eq. ethylene, and 4.08644 g eq. ethylene[75]. (Data from Document 8, of volume 5 of the Portfolio.) If it is assumed the likely potential deviation of the flow from its true value, is plus or minus one standard deviation, then the predicted potential variation ranges are: 2 x 1.7635 = 3.527 g eq. ethylene, 2 x 1.6088 = 3.2176 g eq. ethylene, 2 x 4.8319 = 9.6638 g eq. ethylene, 2 x 1.6118 = 3.2236 g eq. ethylene, 2 x 1.6876 = 3.3752 g eq. ethylene, and 2 x 4.08644 = 8.17288 g eq. ethylene respectively. However, the difference between the two operations is 77.0346 - 67.3372 = 9.6974 g eq. ethylene. (Data from Document 2, of volume 5 of the Portfolio.) This difference in contribution to the effect between primary and secondary lead production operations is less than the predicted potential variation ranges associated with the dominant contributors, when they are combined, and is also less than the range attributable to one of the individual contributors. It is concluded, therefore, that due to the uncertainty in the data, it is not possible to ascertain whether or not a significant difference exists between the two operations.

With regard to primary lead production operations, the difference between the BRM stream throughput and the Isa stream throughput is 36.7913 - 30.5459 = 6.2454 g eq. ethylene. The predicted potential variation ranges attributable to the dominant contributors are: 2 x 1.6088 = 3.2176 g eq. ethylene, and 2 x 1.6876 = 3.3752 g eq. ethylene. Therefore, the difference in consumption between the BRM and Isa streams is less than the predicted

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[74] Throughout the discussion, the names of the inventory flows, and atoms, are those supplied the TEAM™ software. In consequence, names for many chemical compounds do not comply fully with conventional methodology. For further information, see section 9.6.4.5 of this chapter.

[75] Throughout the discussion, the number of decimal places reflects the number calculated by the TEAM™ software. It does not suggest a particular level of accuracy.

potential variation ranges associated with the dominant contributors, when they are combined. Hence, it is concluded, that due to the uncertainty in the data, it is not possible to ascertain whether or not a significant difference exists between the two operations.

9.6.4.5.3 Comparison graphs (Graphs 9.23 to 9.44)

Relevant graphs: Graphs 10.23 to 10.30 -Inventory graphs
Graphs 10.31 to 10.44 -Effect graphs

The graphs show secondary operations only, and compare the BRM company year 1998 - 9 with 1997 - 8. Two levels of operations are shown. The highest level, level 2 (shown in black) is: 'BRM: Secondary Refined Lead Production Operations'. Subsumed within this are the three major level 3 operations, which are:

- BRM (secondary): CX Plant Operations,
- BRM: Overheads of Secondary Processing Operations, and
- BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations.

A positive value indicates a worsening, and a negative value an improvement in performance.

BRM data have been used to construct the foreground, whilst DEAM^{TM} and other data modules have been used for the background process modelling. For both modelled years, the BRM data have been measured and calculated, or estimated and calculated, in a similar manner. In addition, the background processes have been modelled using exactly the same DEAM and other data modules, for both modelled years. Therefore, any biases in both the foreground and background data are likely to be similar.

On the basis of this assumption, the inventories and effects have been compared and used directly as indicators of environmental performance, without the need for sensitivity analysis. However, for the category code: 'BRM: Overheads of Secondary Processing Operations', the inventories and effects are themselves dependant upon the relative throughputs of primary and secondary operations through the BRM site. This is because the overheads associated with secondary refined lead production have been apportioned from the total overheads. Since the ratio of secondary to primary refined lead production differs between the two company years, it is has led to a change in the relative overheads which appears to show a worsening of performance. (See code '<inv.1,1,2>' on the inventory graphs, and <eff.1,1,2>' on the effect graphs.) Therefore, for the overheads only, the differences in the graphs do not represent an actual improvement or deterioration in performance. They should, therefore, be ignored, and it is recommended that a more effective means of apportionment be devised.[76]

However, none of the other level 3 operations shown in the graphs are thus affected. In addition, Graphs 9.1 to 9.22 indicate that the relative contribution of the secondary overheads, is minor in all cases, compared with CX plant operations, and with smelting, rotary furnace, refining and moulding operations. Therefore, the overheads do not expected to exert a significant influence on the comparative 'BRM: Secondary Refined Lead Production Operations'.

The findings of the graphs are summarised in Table 9.9 for the inventories, and Table 9.10 for the effects. The data used to construct these graphs is provided in Document 3 (of volume 5 of the Portfolio). Overheads have been excluded from these tables.

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[76] This represents an area for further research.

Graph	Inventory	Operation and code:		
		BRM: Secondary Refined Lead Production Operations	**BRM (secondary): CX Plant Operations**	**BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations**
		<inv. 1,1>	**<inv. 1,1,1>**	**<inv. 1,1,3>**
9.23	**(r) Natural Gas (in ground)**		++	
9.24	**(r) Oil (in ground)**		+++	
9.25	**(a) Cadmium (Cd)**	-	+++	-
9.26	**(a) Carbon Dioxide (CO2, fossil)***	(+)	++	
9.27	**(a) Lead (Pb)**	-	-	-
9.28	**(a) Sulphur Dioxide (SOx, as SO2)***	+++	+++	+++
9.29	**(w) Cadmium (Cd++)***	(+)	+	
9.30	**(w) Lead (Pb++, Pb4+)***	-	(-)	++

Notes:

+++ = Improvement (30% and greater)
++ = Improvement (20% to 29.9%)
\+ = Improvement (10% to 19.9%)
(+) = Improvement (5% to 9.9%)
[Blank cell] = Change of 4.9% and less (Considered to be no significant change)
(-) = Worsening (5% to 9.9%)
\- = Worsening (10% to 19.9%)
-- = Worsening (20% to 29.9%)
--- = Worsening (30% and greater)

* Throughout the discussion, the names of the inventory flows, and atoms, are those supplied the TEAM™ software. In consequence, names for many chemical compounds do not comply fully with conventional methodology. For further information, see section 9.6.4.5 of this chapter.

Table 9.9: Findings for inventory comparison graphs

Graph	Effect	Operation and code:		
		BRM: Secondary Refined Lead Production Operations	**BRM (secondary) : CX Plant Operations**	**BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations**
		<inv. 1,1>	**<inv. 1,1,1>**	**<inv. 1,1,3>**
9.31	**CML -Air Acidification**	+++	+++	+++
9.32	**CML -Aquatic Ecotoxicity**		+	
9.33	**CML -Depletion of non-renewable resources**	+	+++	++
9.34	**CML -Eutrophication**	+	+++	(+)
9.35	**CML -Eutrophication (water)**		+++	
9.36	**CML -Human Toxicity**	+++	++	+++
9.37	**CML -Terrestrial Eco-toxicity**	(+)	+++	
9.38	**IPCC -Greenhouse effect (direct, 20 years)**	(+)	++	
9.39	**IPCC -Greenhouse effect (direct, 100 years)**	(+)	++	
9.40	**IPCC -greenhouse effect (direct, 500 years)**	(+)	++	
9.41	**WMO -Depletion of the Ozone Layer (high)**		+++	
9.42	**WMO -Depletion of the Ozone Layer (low)**		+++	
9.43	**WMO -Photochemical Oxidant Formation (high)**	(-)	+	(-)
9.44	**WMO -Photochemical Oxidant Formation (low)**	(-)	+	-

Notes:

+++ = Improvement (30% and greater)
++ = Improvement (20% to 29.9%)
+ = Improvement (10% to 19.9%)
(+) = Improvement (5% to 9.9%)
[Blank cell] = Change of 4.9% and less (Considered to be no significant change)
(-) = Worsening (5% to 9.9%)
- = Worsening (10% to 19.9%)
-- = Worsening (20% to 29.9%)
--- = Worsening (30% and greater)

Table 9.10: Findings for effect comparison graphs

9.7. Conclusions and recommendations

9.7.1. General

Graphs 9.1. to 9.22 provide quantitative indications of selected inventories and effects for BRM whole site operations (plus subsidiary operation scales i.e. 'levels'), whilst Graphs 9.23 to 9.44 provide quantitative comparisons of selected inventories and effects for BRM secondary refined lead production operations (plus subsidiary operation scales). The discussion in sections 9.6.4.5.1 and 9.6.4.5.2 provides qualitative analyses of the quantitative data indicated in Graphs 9.1 to 9.22, whilst the discussion in section 9.5.4.5.3 explains how comparisons, such as those in Graphs 9.23 to 9.44, may be used as environmental performance indicators at different scales.

Data from both graph sets, may also be used to aid identification of 'best' environmental practice. This may be achieved by comparing different modelled operations within the BRM case study, or by comparing modelled operations with various others. Comparisons may be made using various valuation criteria. One simple approach might be to assume that 'less is best'. However, numerous other valuation approaches could be adopted.[77] Whatever the approach, they should be only used as aids for identifying 'best' environmental practice. It should not be assumed they enable identification *per se*. This is because, the modelling and assessment procedures, which underlie comparisons, are based on implicit framing assumptions. As noted in section 4.4.2.2 (chapter 4), these need to be identified and challenged, and then accepted, rejected or modified as necessary, if the 'best' decisions are to be made. Since, this is a departure from much of common practice, it is discussed further in the Overall Conclusions and Recommendations (chapter 11).

Sections 9.6.4.1 to 9.6.4.4 explain the need to identify the types and sources of uncertainty in the 'BRM: Whole Site' and subsidiary 'BRM: Secondary Refined Lead Production' models, and also for assessments of their potential influence on inventory and effect findings. It is for this reason, that dominance and sensitivity analyses have been used to support all assertions made in the discussion of the inventory and effect graphs (sections

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[77] Valuation approaches are discussed further in the elective EngD module, which was enclosed as Document 15, in volume 2 of the Portfolio. This document has not been supplied with this copy of the thesis.

9.6.4.5.1 and 9.6.4.5.2). It is also apparent from the discussion that, whilst the modelling provides quantitative data, the assertions which have been made regarding the findings are all qualitative i.e. they simply assert whether one emission source is dominant and/or is significantly different from another. It is theoretically possible to identify the actual variability from the 'true' value for all foreground and background data. For foreground data, this has been assessed by the 'variability of data' columns which have been completed for all data. For background data, however, this is currently not possible, because the suppliers of the data do not provide sufficient information for this. Since the dominance analysis (discussed along with the graph findings in sections 9.6.4.5.1 and 9.6.4.5.2) reveals that most of the dominant contributors to the selected inventories and effects are associated with background atoms, the 'true' variability is not known. However, all of the background data are from recognised sources. This means that they are from the pool of data which is being used, currently, by other practitioners of modelling approaches such as these. Therefore, any errors, inaccuracies and omissions within these data will be duplicated by these other practitioners in their own modelling. It also means that when (or if) comparisons between the modelling conducted here, and other models are made, they will be commensurate, at least with regard to this aspect.

Taking into account these considerations does not mean, necessarily, that the quantitative data (provided in the inventories in volume 4 of the Portfolio) cannot be used to support claims which people may wish to make. Rather, it means that where they are, they must be accompanied by uncertainty assessments of the kind provided in sections 9.6.4.5.1 and 9.6.4.5.2 for each graph. This is because these discussions set the boundaries to the uncertainties underlying the numerical inventory and effect data.

However, this is not necessary for the comparative inventory and effect graphs (Graphs 9.23 to 9.44) as the uncertainties in the two models being compared, are assumed to be commensurate (for the reasons outlined in section 9.6.4.5.3). Hence, sensitivity analyses are not considered to be required, as an interpretation adjunct for these graphs.

With all modelling, findings can only be made on the basis of what is known. Whilst strenuous efforts have been made, through discussion with the data suppliers, to ensure all potentially significant inflows and outflows have been either incorporated in the modelling, or their absence noted and their possible influence assessed, it is possible that some significant flows may have been missed.

Finally, it should be noted that the modelling, data output and findings merely provide a snapshot. Nevertheless, the snapshot which the modelling provides is for a site which is currently the largest single producer of refined lead in the world. Therefore, it provides a bench mark. However, since processes are constantly evolving, the modelling and findings, are only able to provide an historical record. Nevertheless, providing any changes which have been made, since when the modelling was conducted, are not significant, it will still provide an accurate reflection of the current circumstance. Though, until modelling is updated, it cannot be known with certainty whether changes, which have occurred, are significant.

Therefore, if the full potential of the modelling is to be achieved, the modelling and assessments will need to be updated on a regular basis into the foreseeable future. Key uncertainties in this modelling, could also be addressed in such future iterations. These questions are considered further in the Overall Conclusions and Recommendations (chapter 11).

9.7.2. Specifics

The dominance analyses, indicate that, with two exceptions data atoms do not make a significant contribution overall to any of the graphed inventories and effects. The exceptions are the use of the data atoms:

- '241 Water (Softened): Production' which has been used as a substitute for 'towns water' (as indicated in Table 9.5), and
- '211 Pulp (Sulphate, Unbleached): Production.1' which has been used as a substitute for 'calcium lignosulphonate' in the system node: 'BRM: Lead Refining Operations (Isa stream)'.

Dominance analyses have shown, that '241 Water (Softened): Production' data atoms make a dominant contribution (i.e. greater than 5 % to the total) to the graphed inventory for '(a) Carbon Dioxide (CO2, fossil)', and to the following graphed effects:

- 'IPCC -Greenhouse Effect (direct, 100, 20 and 500 years)', and
- 'WMO -Photochemical Oxidant Formation (high and low)'.

It is thought likely that the use of the '241 Water (Softened): Production' atom exaggerates the actual LCI inventory for the type of water being considered. This means that inventories and effects which are significantly influenced by the use of this atom represent 'worst case scenarios', a approach which is in full compliance with the Precautionary Principle (which is that, in the absence of actual data one should always assume a worst case). However, without actual modelling data, this cannot be known with certainty.

The atom '211 Pulp (Sulphate, Unbleached): Production.1' makes a significant contribution to the inventory '(w) Arsenic (As3+, As5+)'[78] only. It is used as a substitute for 'calcium lignosulphonate'. However, not known how close an approximation it is for this substance.

It is recommended, therefore, that should assertions be made to the public, regarding any of the inventories and effects identified above, they must also include a comment about the influence of these substitute atoms on the findings. It is also recommended, that in future iterations of the modelling, actual LCI data be collected for 'calcium lignosulphonate', and for all of the flows listed in Table 9.5. These data should then replace the use of the substitute atoms for modelling these materials.

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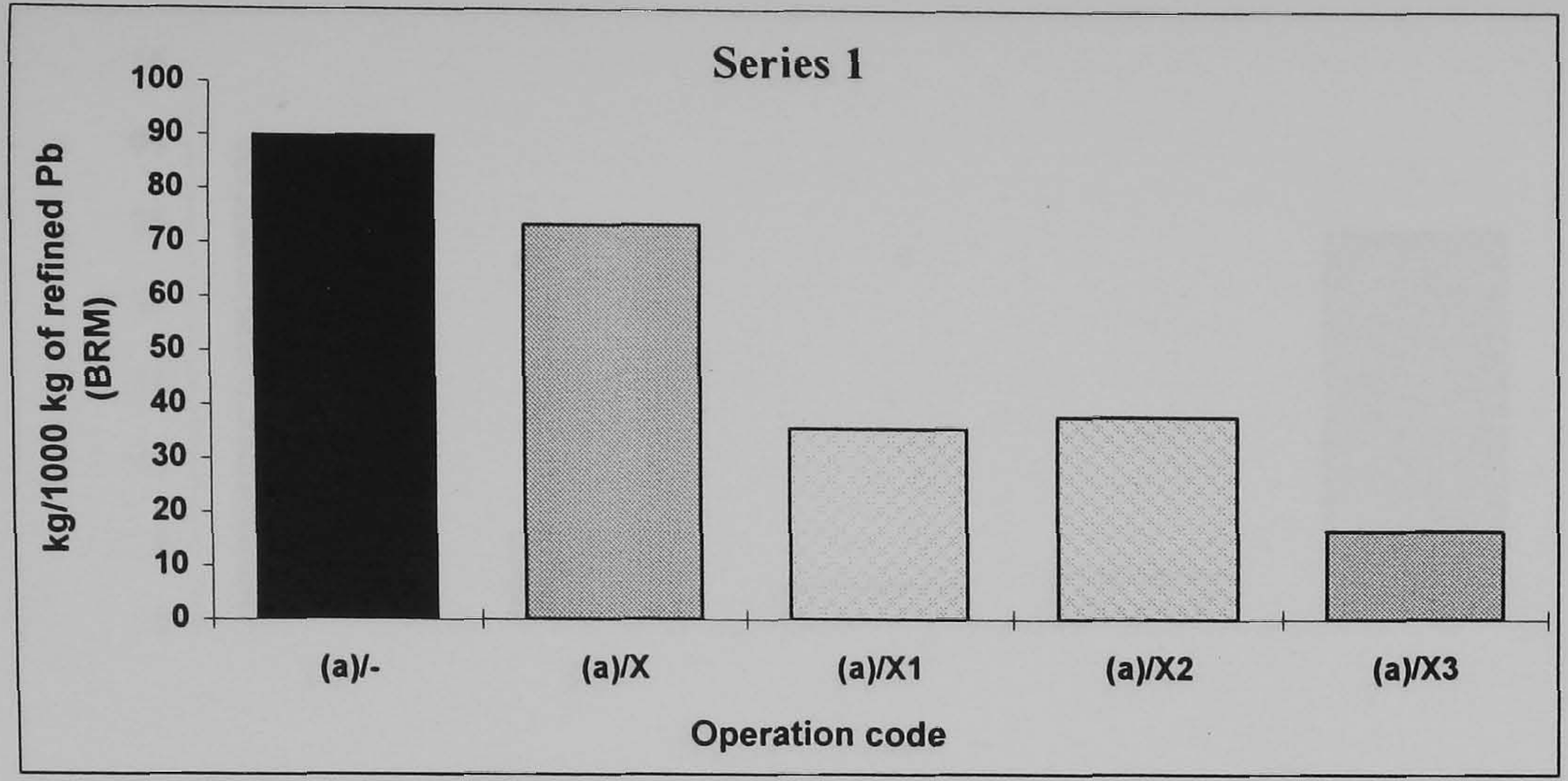
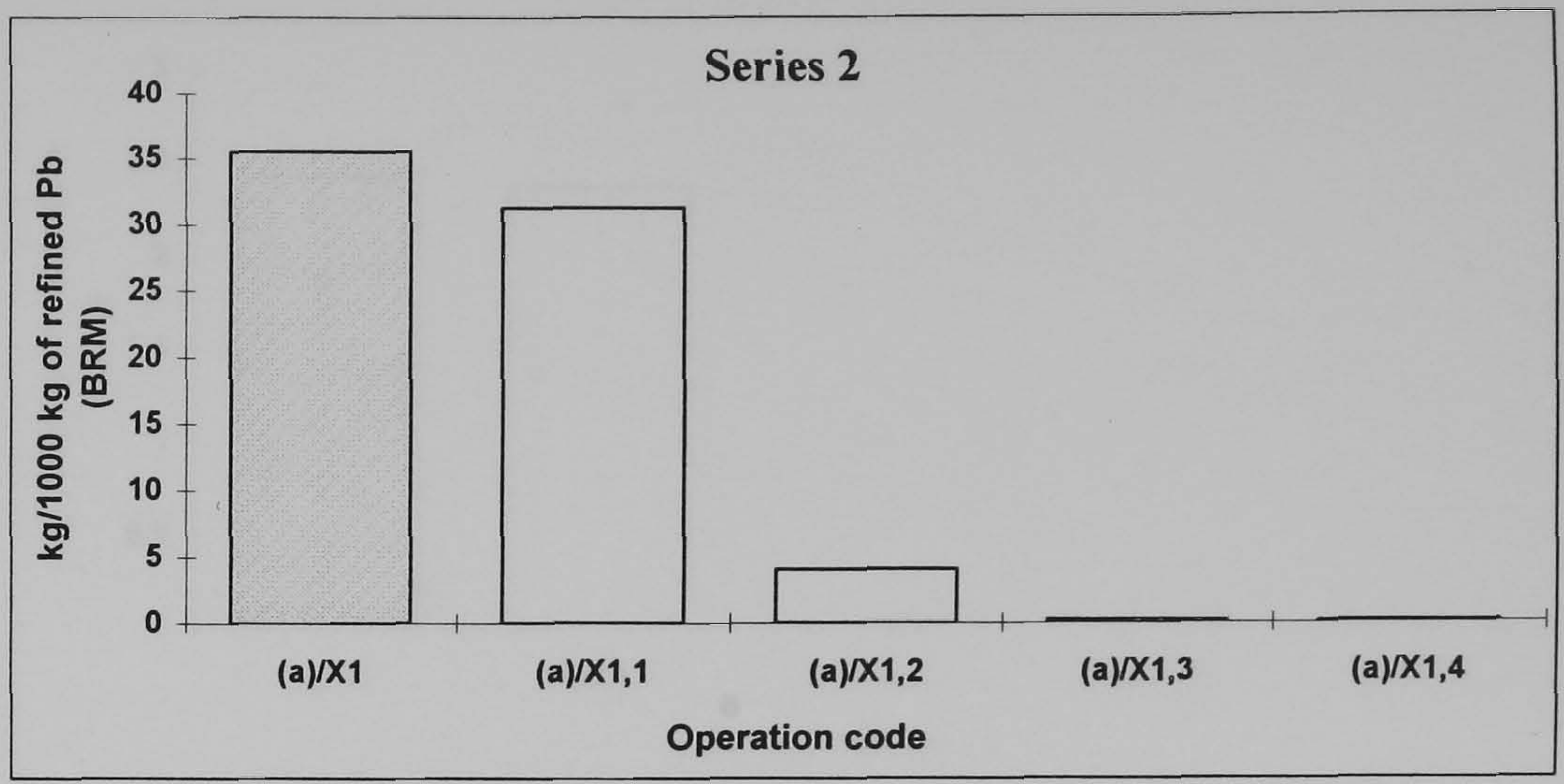
[78] Throughout the discussion, the names of the inventory flows, and atoms, are those supplied the TEAM™ software. In consequence, names for many chemical compounds do not comply fully with conventional methodology. For further information, see section 9.6.4.5 of this chapter.

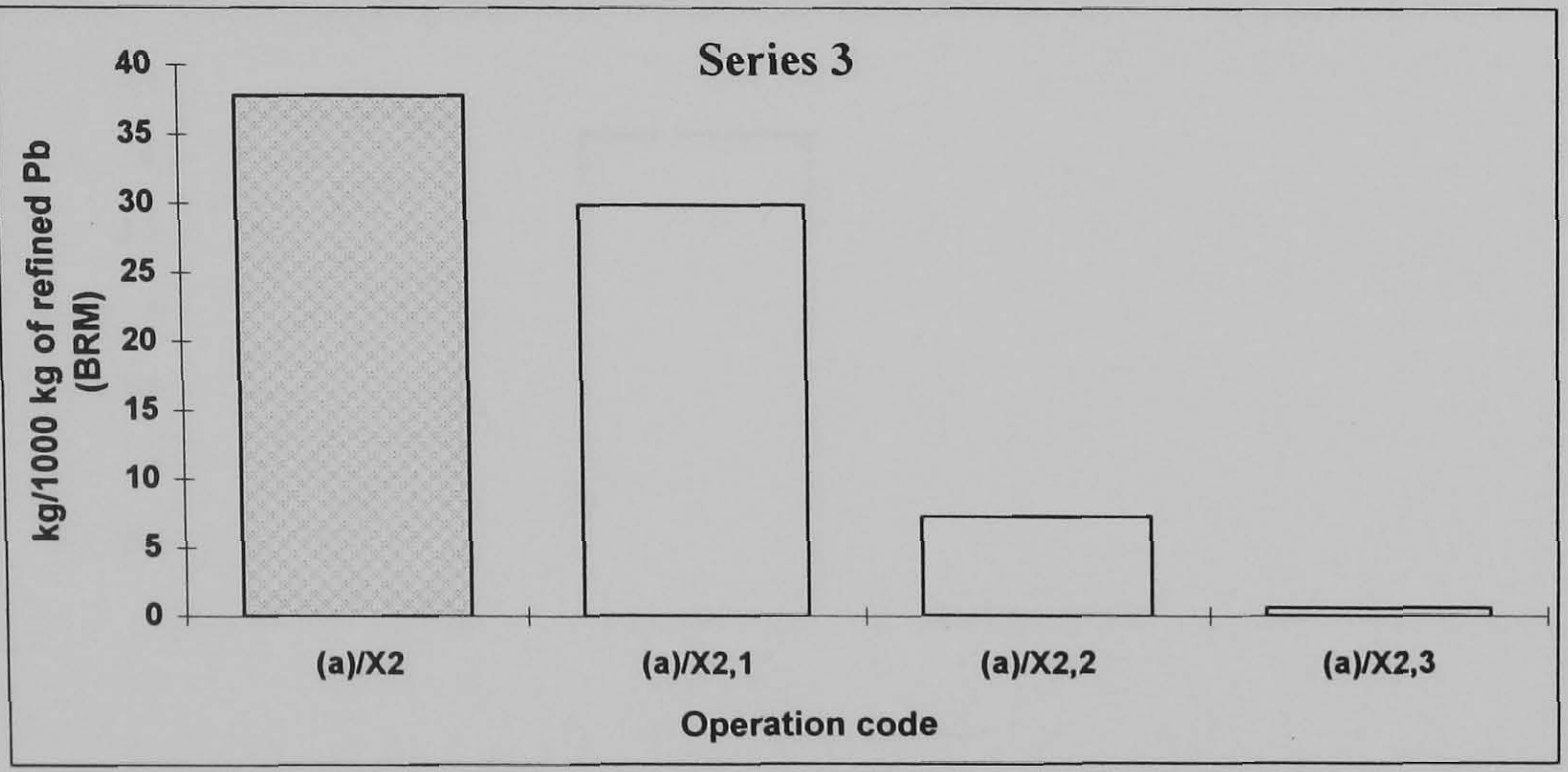
9.8. Graphs of systems modelled and compared



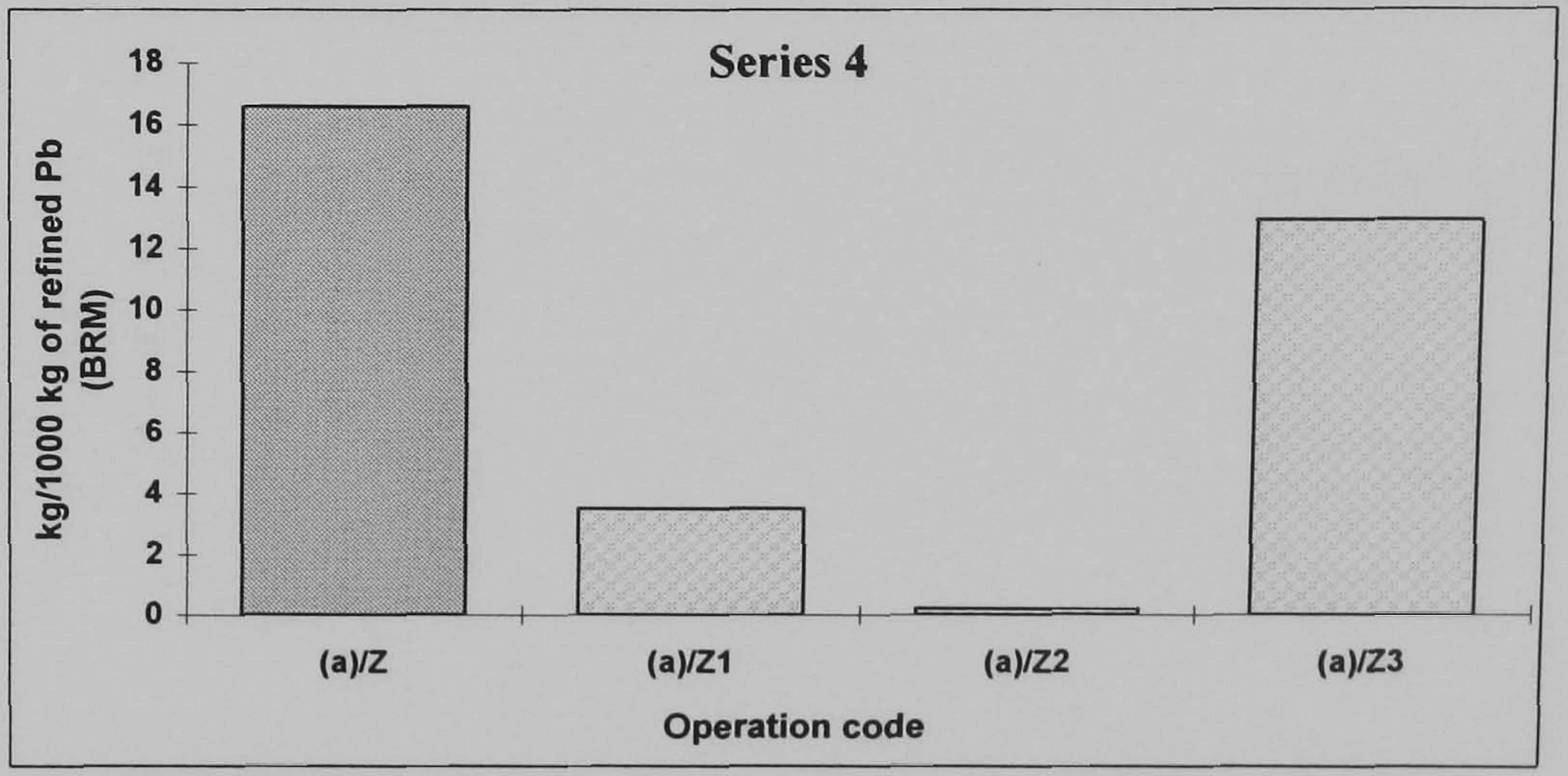
Code:	Operation:
(a)/-	BRM: Lead Material Processing Operations at Northfleet, Kent, UK
(a)/X	BRM: Primary Refined Lead Production Operations
(a)/X1	BRM: Refined Lead Production Operations (BRM stream throughput)
(a)/X2	BRM: Refined Lead production Operations (Isa stream throughput)
(a)/X3	BRM: Secondary Refined Lead Production Operations



Code:	Operation:
(a)/X1	BRM: Refined Lead Production Operations (BRM stream throughput)
(a)/X1,1	BRM: Lead Refining Operations (BRM floor throughput)
(a)/X1,2	BRM: On-site Electricity Generation Operations (BRM floor throughput)
(a)/X1,3	BRM: Overheads to BZL Input to BRM Floor Throughput Operations
(a)/X1,4	BRM: Overheads to MHD Input to BRM Floor Throughput Operations

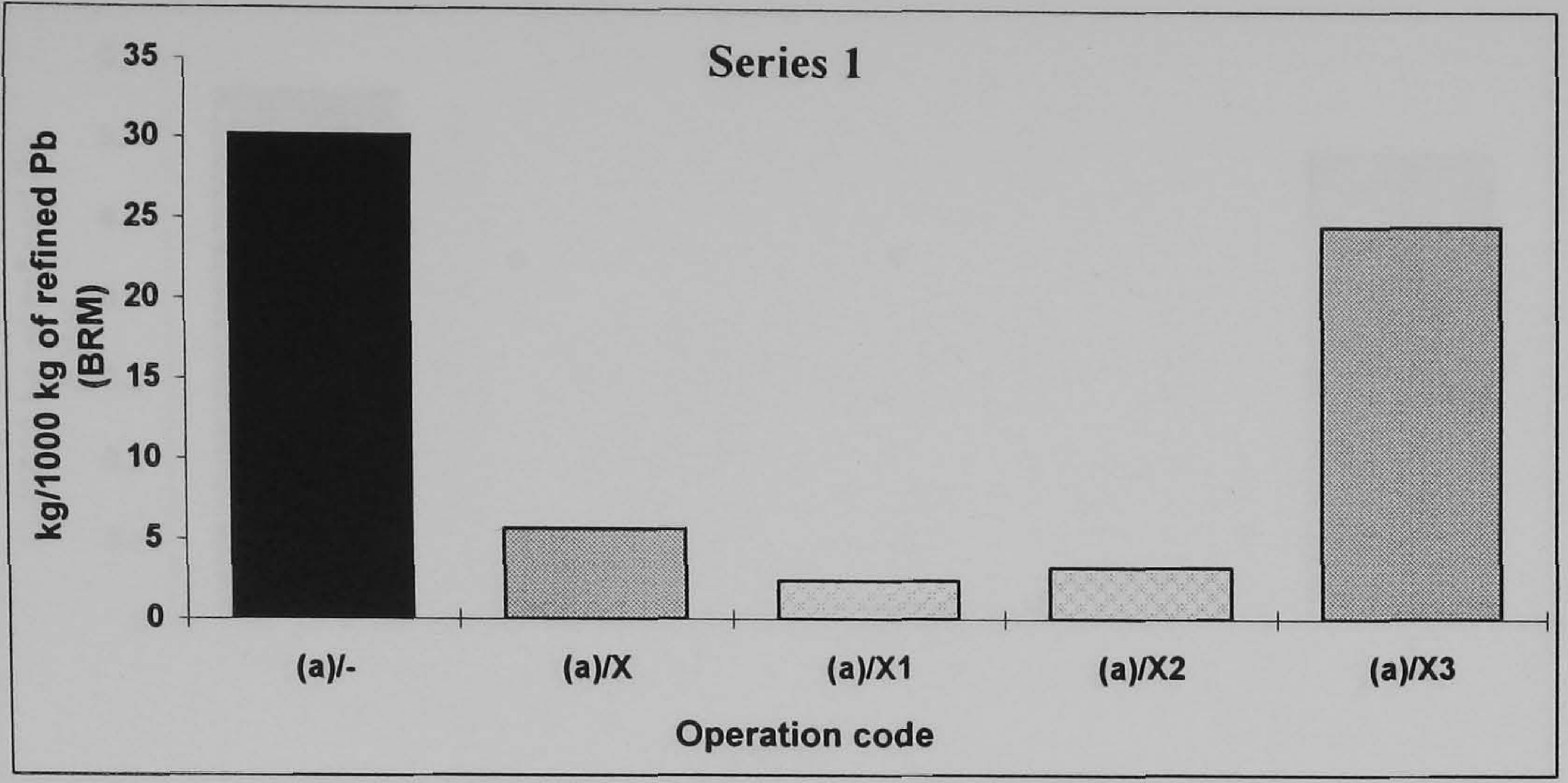


Code:	Operation:
(a)/X2	BRM: Refined Lead production Operations (Isa stream throughput)
(a)/X2,1	BRM: Lead Refining Operations (Isa stream)
(a)/X2,2	BRM: On-site Electricity Generation Operations (Isa floor throughput)
(a)/X2,3	BRM: Overheads (Isa stream operations)

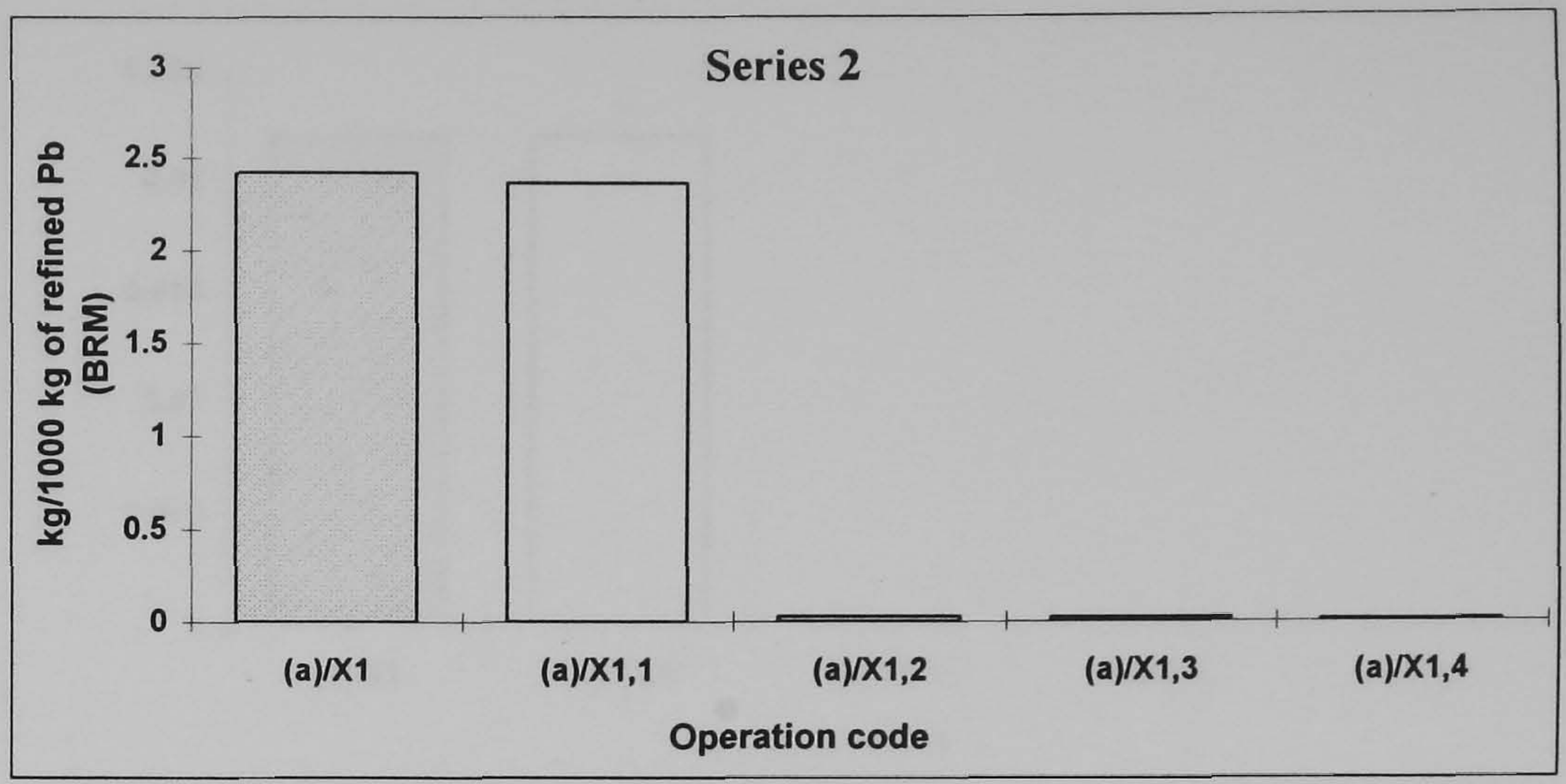


Code:	Operation:
(a)/Z	BRM: Secondary Refined Lead Production Operations
(a)/Z1	BRM (secondary): CX Plant Operations
(a)/Z2	BRM (secondary): Overheads of Secondary Processing Operations
(a)/Z3	BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations

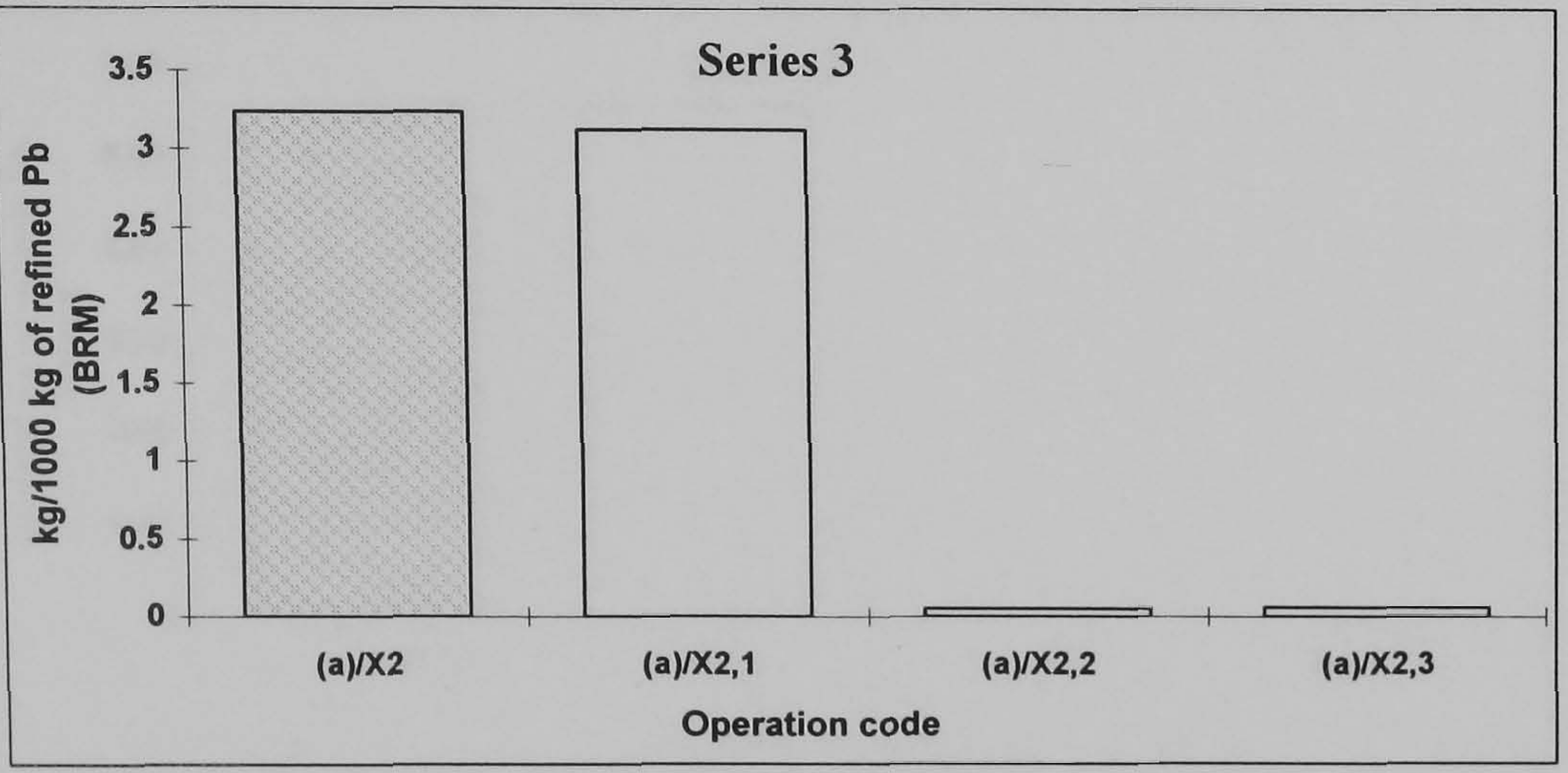
Graph 9.1: (r) Natural Gas (in ground) -Modelled year (a). Series 1 to 4



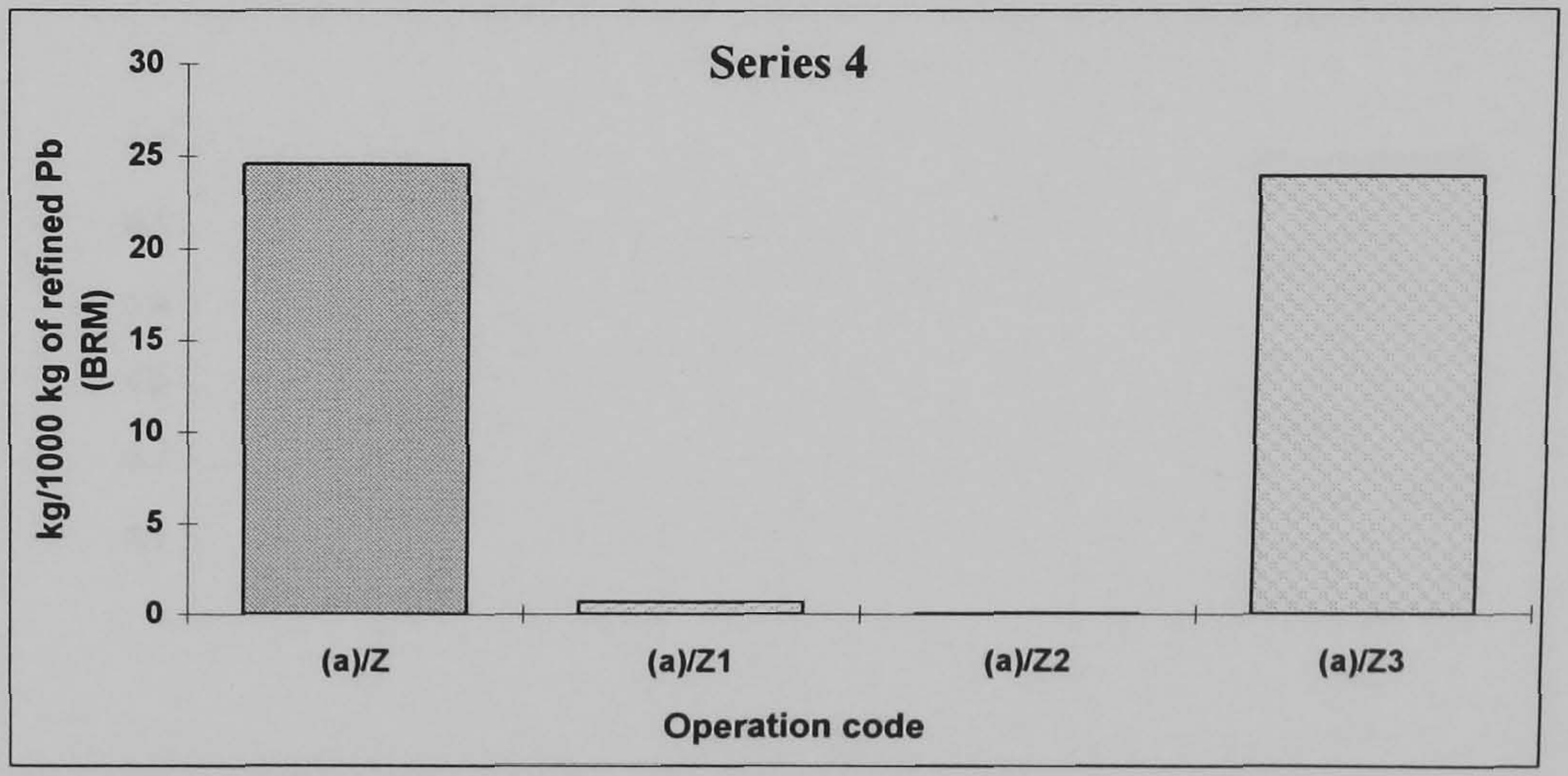
Code:	Operation:
(a)/-	BRM: Lead Material Processing Operations at Northfleet, Kent, UK
(a)/X	BRM: Primary Refined Lead Production Operations
(a)/X1	BRM: Refined Lead Production Operations (BRM stream throughput)
(a)/X2	BRM: Refined Lead production Operations (Isa stream throughput)
(a)/X3	BRM: Secondary Refined Lead Production Operations



Code:	Operation:
(a)/X1	BRM: Refined Lead Production Operations (BRM stream throughput)
(a)/X1,1	BRM: Lead Refining Operations (BRM floor throughput)
(a)/X1,2	BRM: On-site Electricity Generation Operations (BRM floor throughput)
(a)/X1,3	BRM: Overheads to BZL Input to BRM Floor Throughput Operations
(a)/X1,4	BRM: Overheads to MHD Input to BRM Floor Throughput Operations

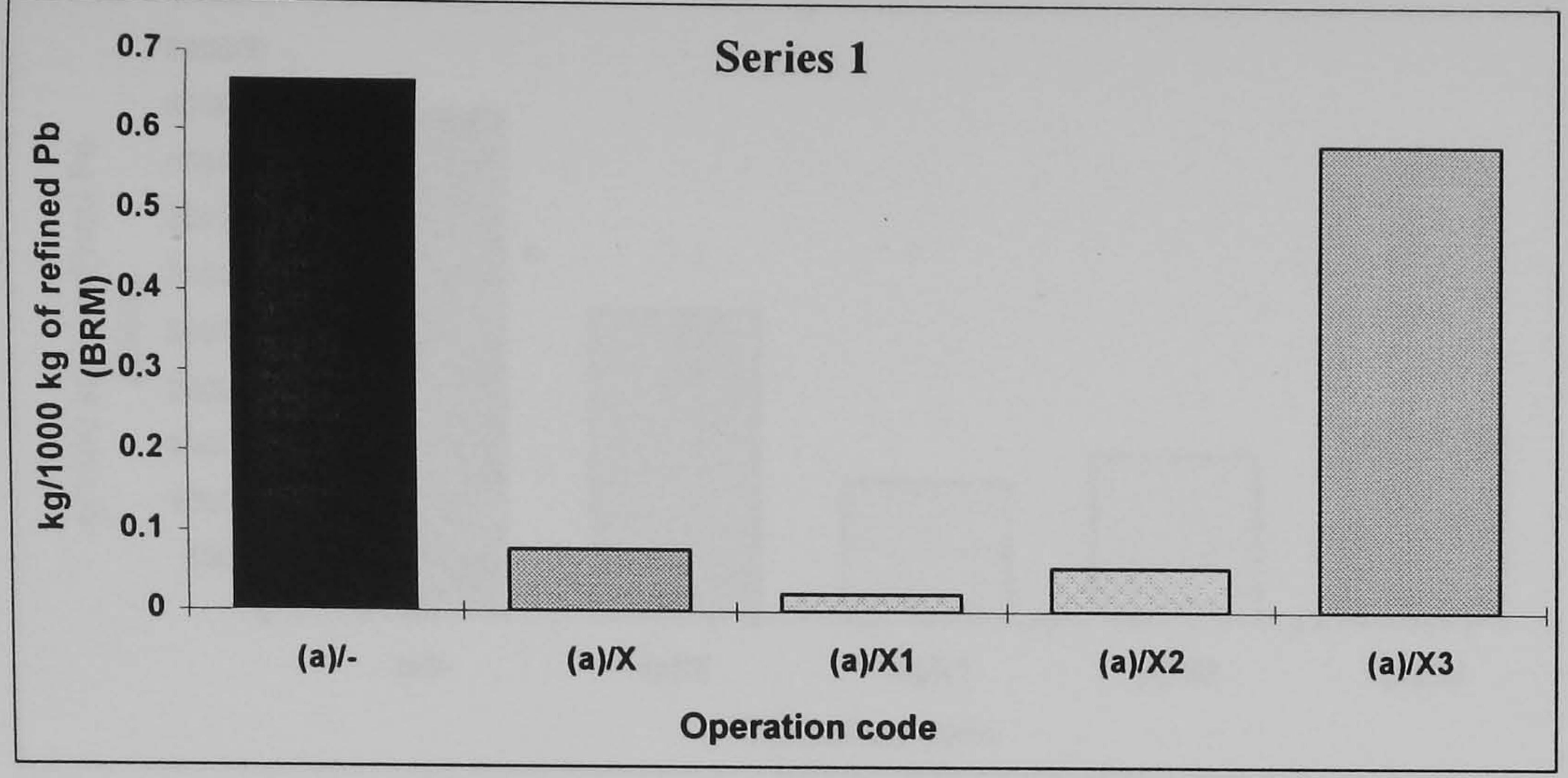


Code:	Operation:
(a)/X2	BRM: Refined Lead production Operations (Isa stream throughput)
(a)/X2,1	BRM: Lead Refining Operations (Isa stream)
(a)/X2,2	BRM: On-site Electricity Generation Operations (Isa floor throughput)
(a)/X2,3	BRM: Overheads (Isa stream operations)

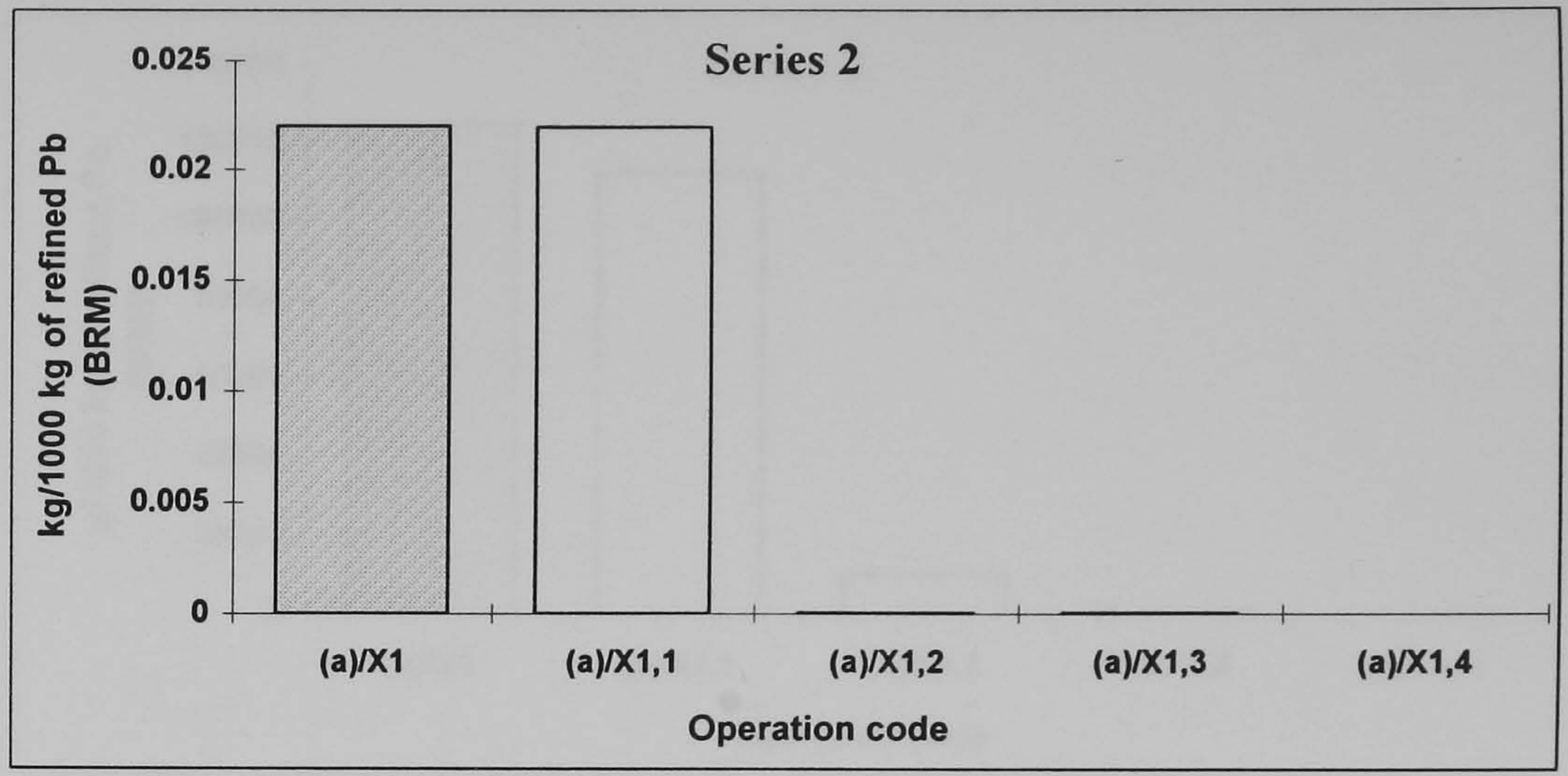


Code:	Operation:
(a)/Z	BRM: Secondary Refined Lead Production Operations
(a)/Z1	BRM (secondary): CX Plant Operations
(a)/Z2	BRM (secondary): Overheads of Secondary Processing Operations
(a)/Z3	BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations

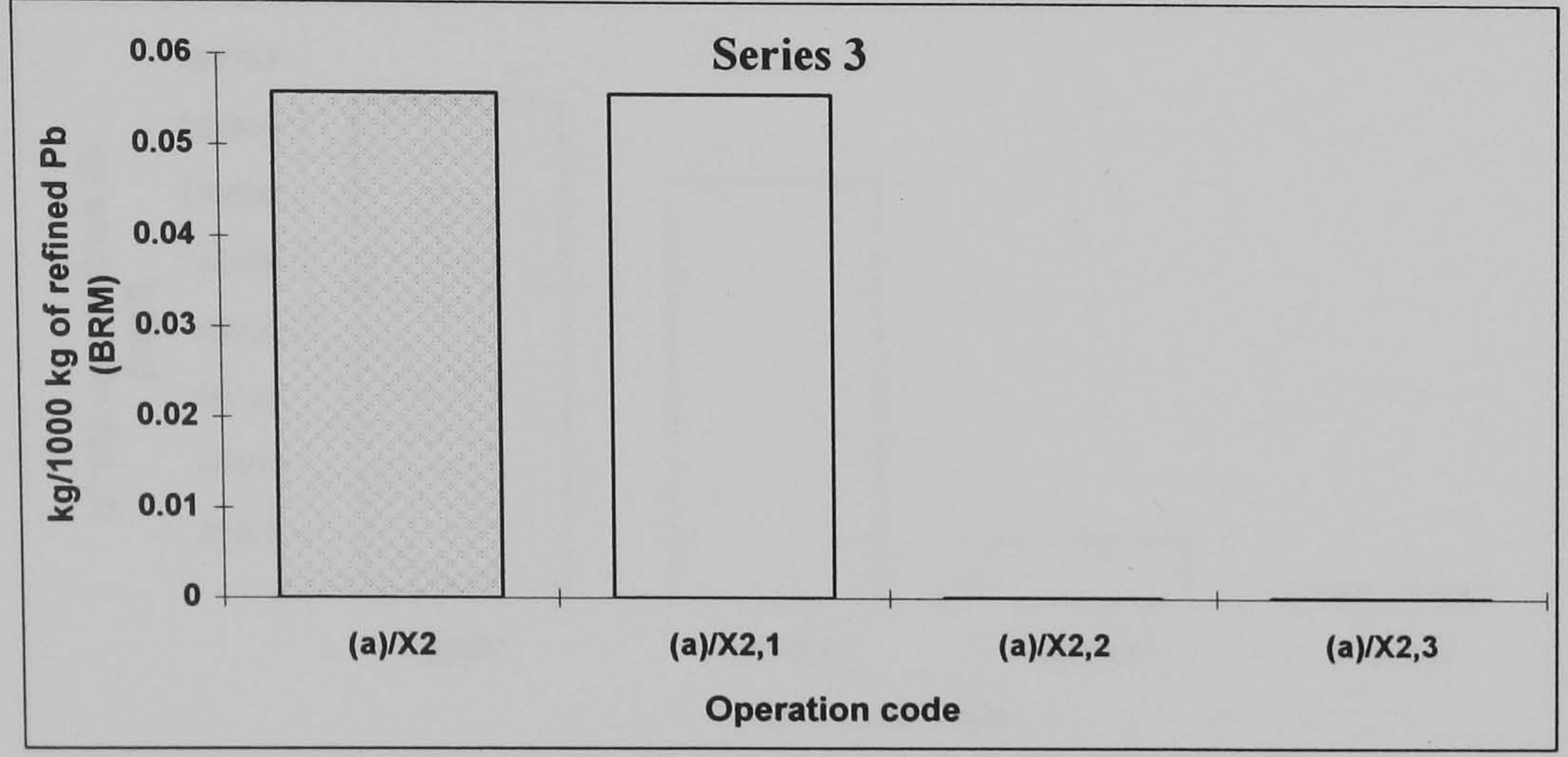
Graph 9.2: (r) Oil (in ground) -Modelled year (a). Series 1 to 4



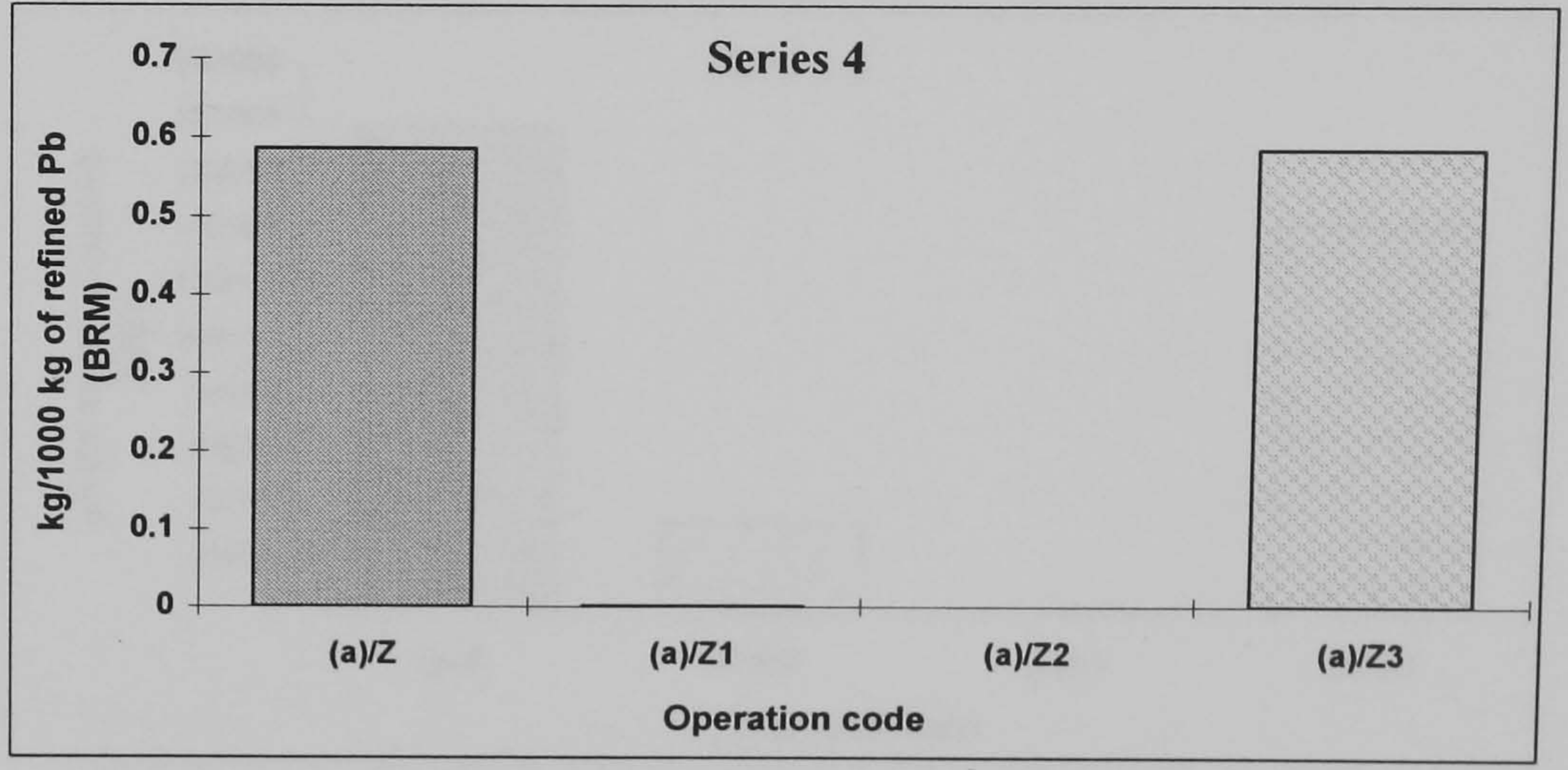
Code:	Operation:
(a)/-	BRM: Lead Material Processing Operations at Northfleet, Kent, UK
(a)/X	BRM: Primary Refined Lead Production Operations
(a)/X1	BRM: Refined Lead Production Operations (BRM stream throughput)
(a)/X2	BRM: Refined Lead production Operations (Isa stream throughput)
(a)/X3	BRM: Secondary Refined Lead Production Operations



Code:	Operation:
(a)/X1	BRM: Refined Lead Production Operations (BRM stream throughput)
(a)/X1,1	BRM: Lead Refining Operations (BRM floor throughput)
(a)/X1,2	BRM: On-site Electricity Generation Operations (BRM floor throughput)
(a)/X1,3	BRM: Overheads to BZL Input to BRM Floor Throughput Operations
(a)/X1,4	BRM: Overheads to MHD Input to BRM Floor Throughput Operations

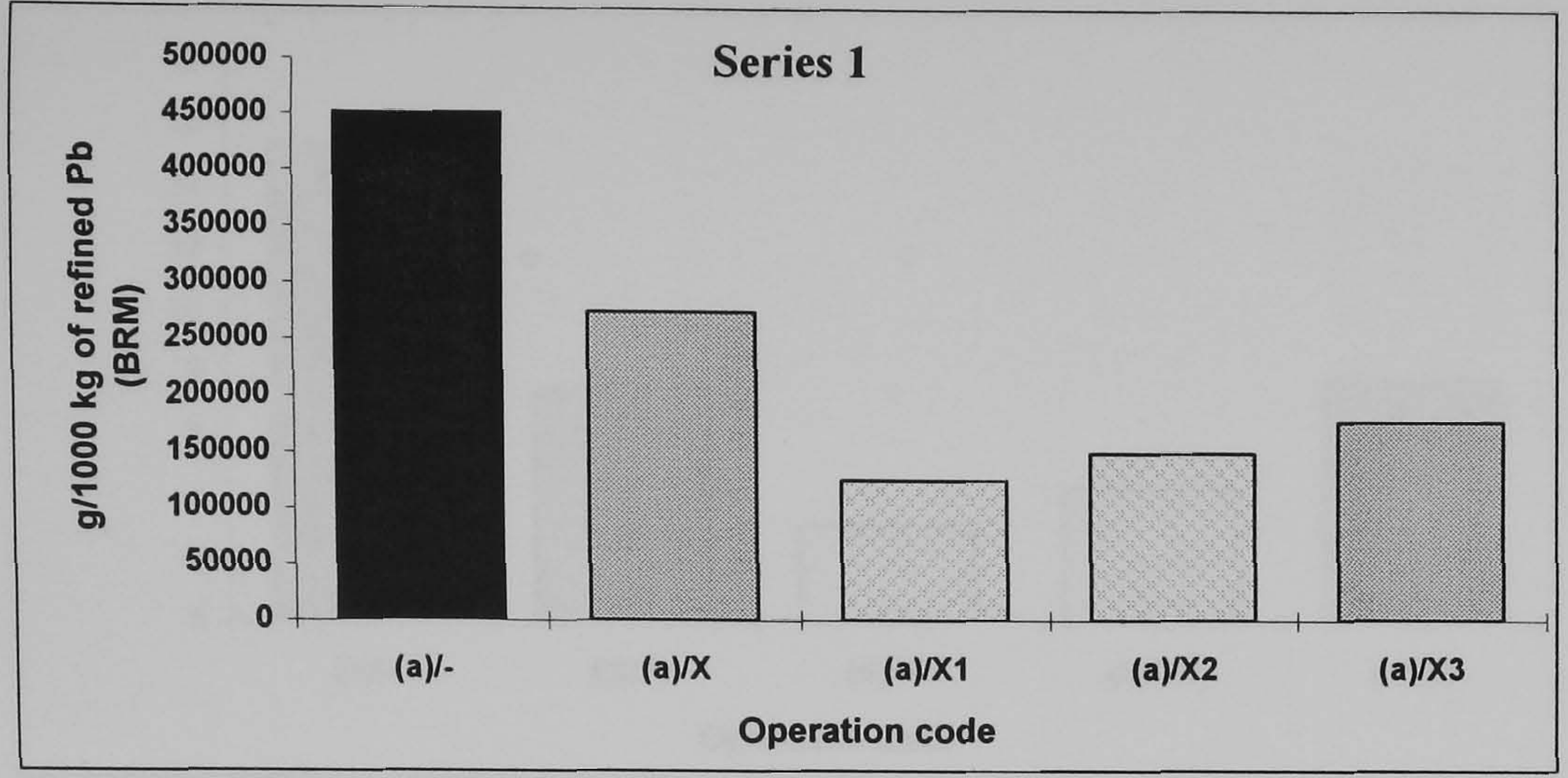


Code:	Operation:
(a)/X2	BRM: Refined Lead production Operations (Isa stream throughput)
(a)/X2,1	BRM: Lead Refining Operations (Isa stream)
(a)/X2,2	BRM: On-site Electricity Generation Operations (Isa floor throughput)
(a)/X2,3	BRM: Overheads (Isa stream operations)

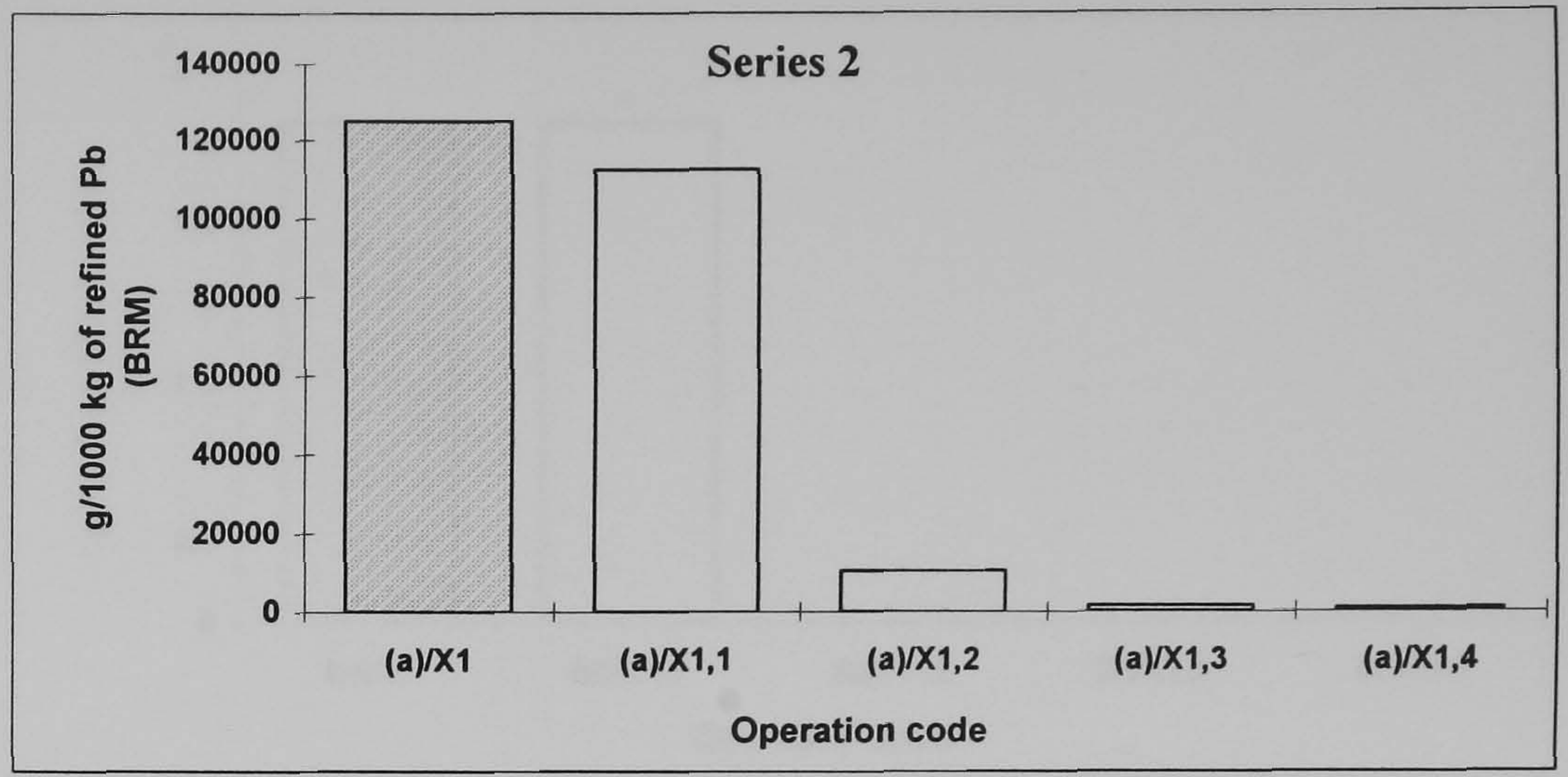


Code:	Operation:
(a)/Z	BRM: Secondary Refined Lead Production Operations
(a)/Z1	BRM (secondary): CX Plant Operations
(a)/Z2	BRM (secondary): Overheads of Secondary Processing Operations
(a)/Z3	BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations

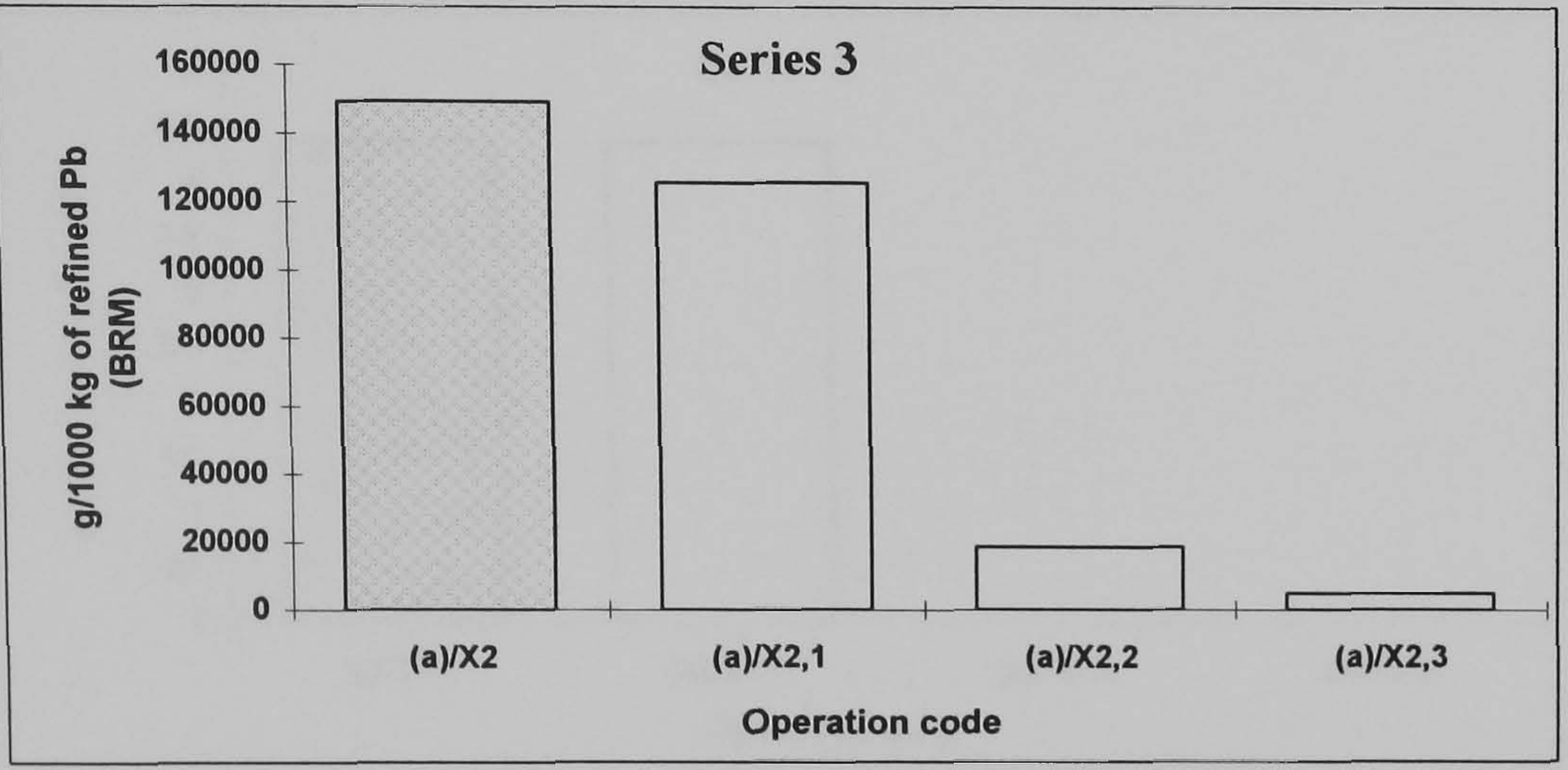
Graph 9.3: (a) Cadmium (Cd) -Modelled year (a). Series 1 to 4



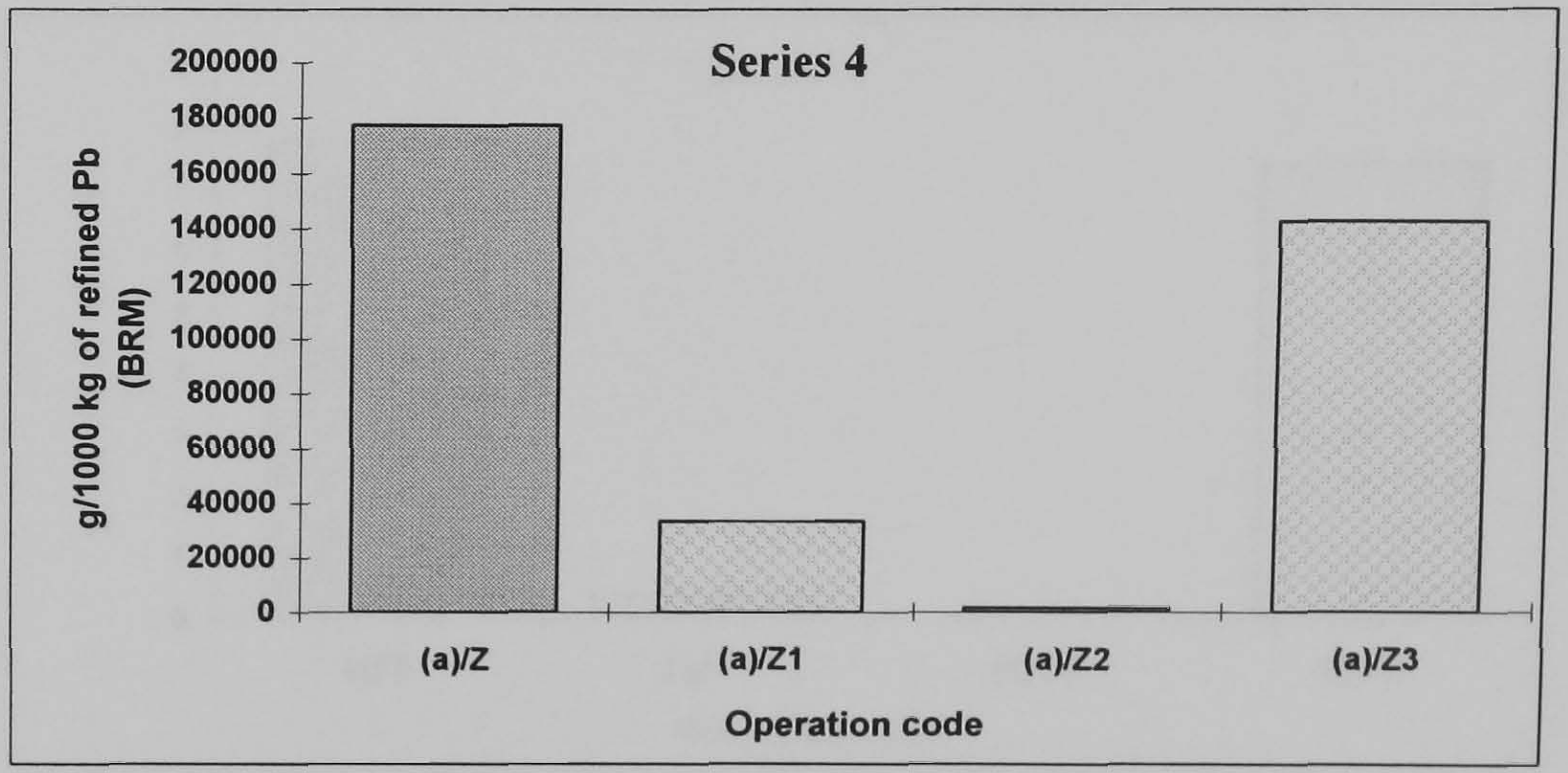
Code:	Operation:
(a)/-	BRM: Lead Material Processing Operations at Northfleet, Kent, UK
(a)/X	BRM: Primary Refined Lead Production Operations
(a)/X1	BRM: Refined Lead Production Operations (BRM stream throughput)
(a)/X2	BRM: Refined Lead production Operations (Isa stream throughput)
(a)/X3	BRM: Secondary Refined Lead Production Operations



Code:	Operation:
(a)/X1	BRM: Refined Lead Production Operations (BRM stream throughput)
(a)/X1,1	BRM: Lead Refining Operations (BRM floor throughput)
(a)/X1,2	BRM: On-site Electricity Generation Operations (BRM floor throughput)
(a)/X1,3	BRM: Overheads to BZL Input to BRM Floor Throughput Operations
(a)/X1,4	BRM: Overheads to MHD Input to BRM Floor Throughput Operations

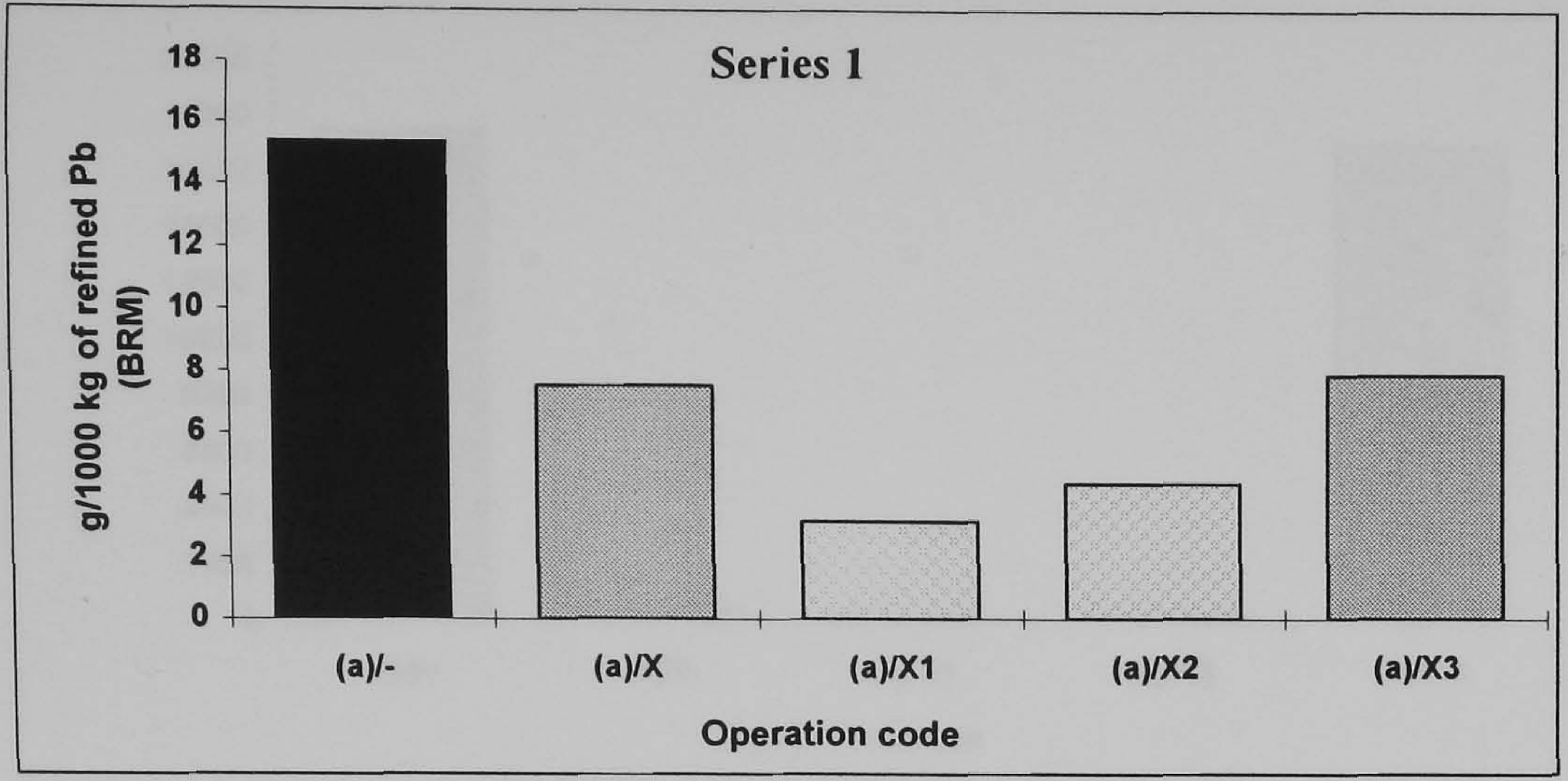


Code:	Operation:
(a)/X2	BRM: Refined Lead production Operations (Isa stream throughput)
(a)/X2,1	BRM: Lead Refining Operations (Isa stream)
(a)/X2,2	BRM: On-site Electricity Generation Operations (Isa floor throughput)
(a)/X2,3	BRM: Overheads (Isa stream operations)

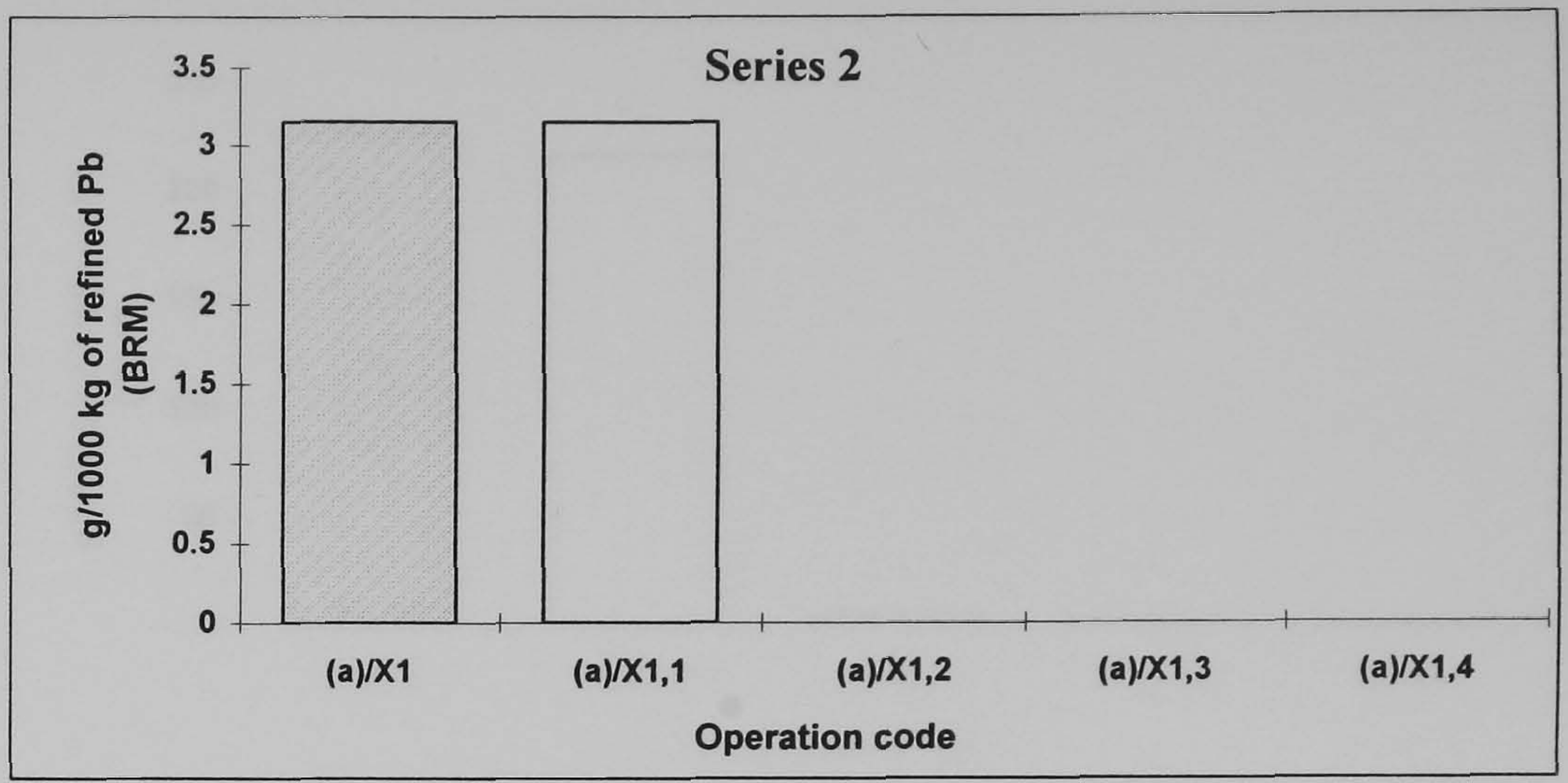


Code:	Operation:
(a)/Z	BRM: Secondary Refined Lead Production Operations
(a)/Z1	BRM (secondary): CX Plant Operations
(a)/Z2	BRM (secondary): Overheads of Secondary Processing Operations
(a)/Z3	BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations

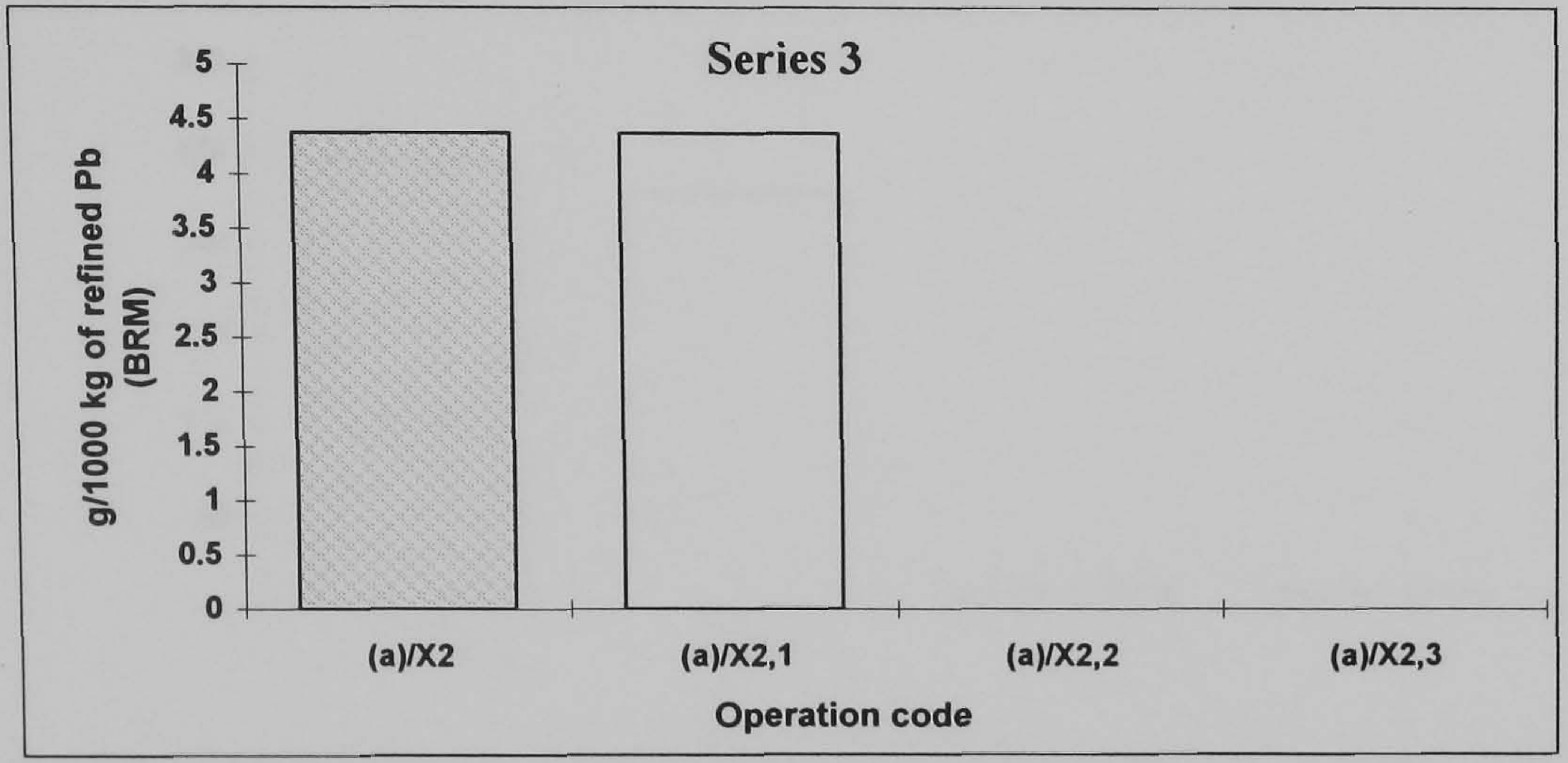
Graph 9.4: (a) Carbon Dioxide (CO2, fossil) -Modelled year (a). Series 1 to 4



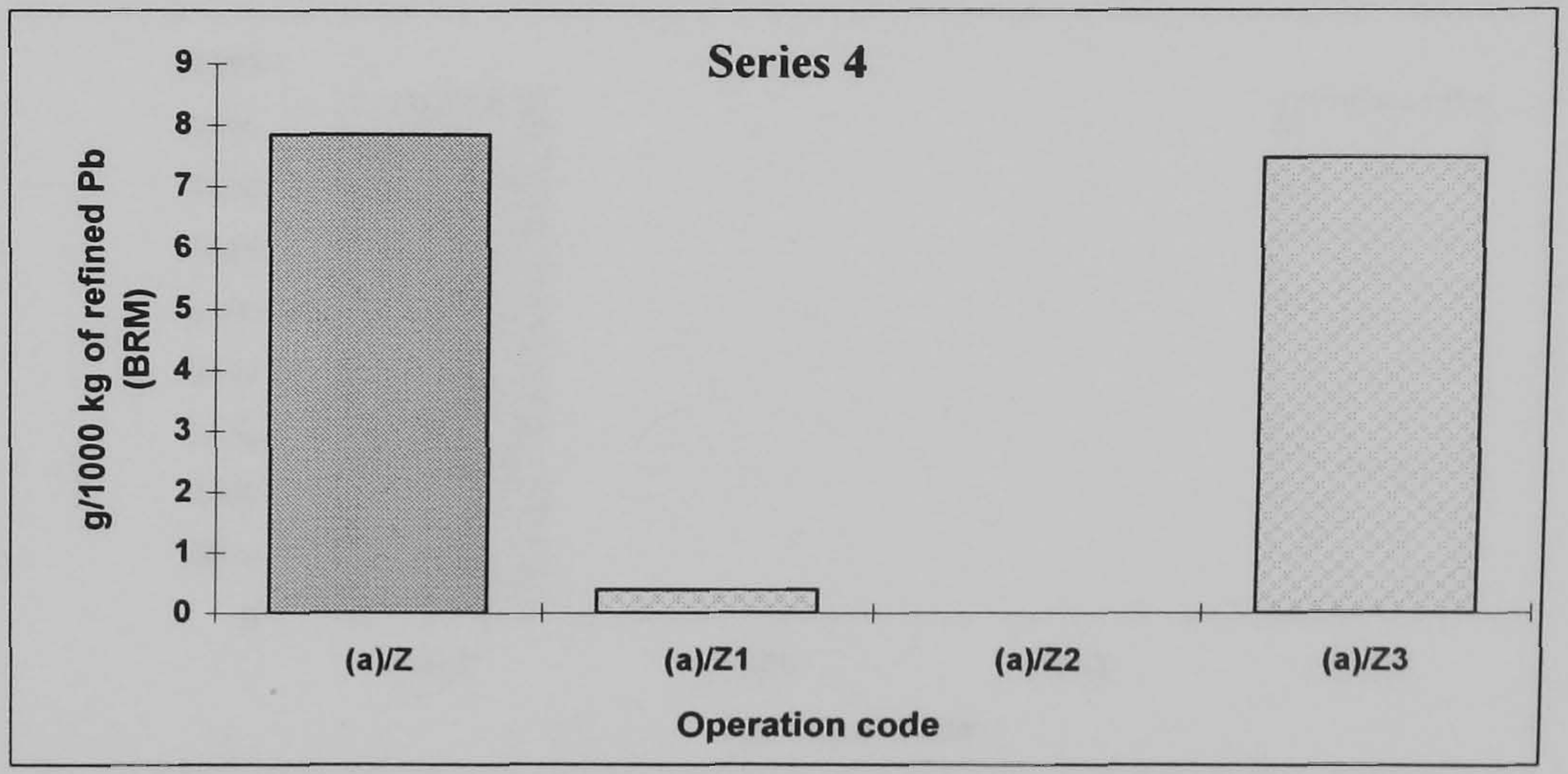
Code:	Operation:
(a)/-	BRM: Lead Material Processing Operations at Northfleet, Kent, UK
(a)/X	BRM: Primary Refined Lead Production Operations
(a)/X1	BRM: Refined Lead Production Operations (BRM stream throughput)
(a)/X2	BRM: Refined Lead production Operations (Isa stream throughput)
(a)/X3	BRM: Secondary Refined Lead Production Operations



Code:	Operation:
(a)/X1	BRM: Refined Lead Production Operations (BRM stream throughput)
(a)/X1,1	BRM: Lead Refining Operations (BRM floor throughput)
(a)/X1,2	BRM: On-site Electricity Generation Operations (BRM floor throughput)
(a)/X1,3	BRM: Overheads to BZL Input to BRM Floor Throughput Operations
(a)/X1,4	BRM: Overheads to MHD Input to BRM Floor Throughput Operations

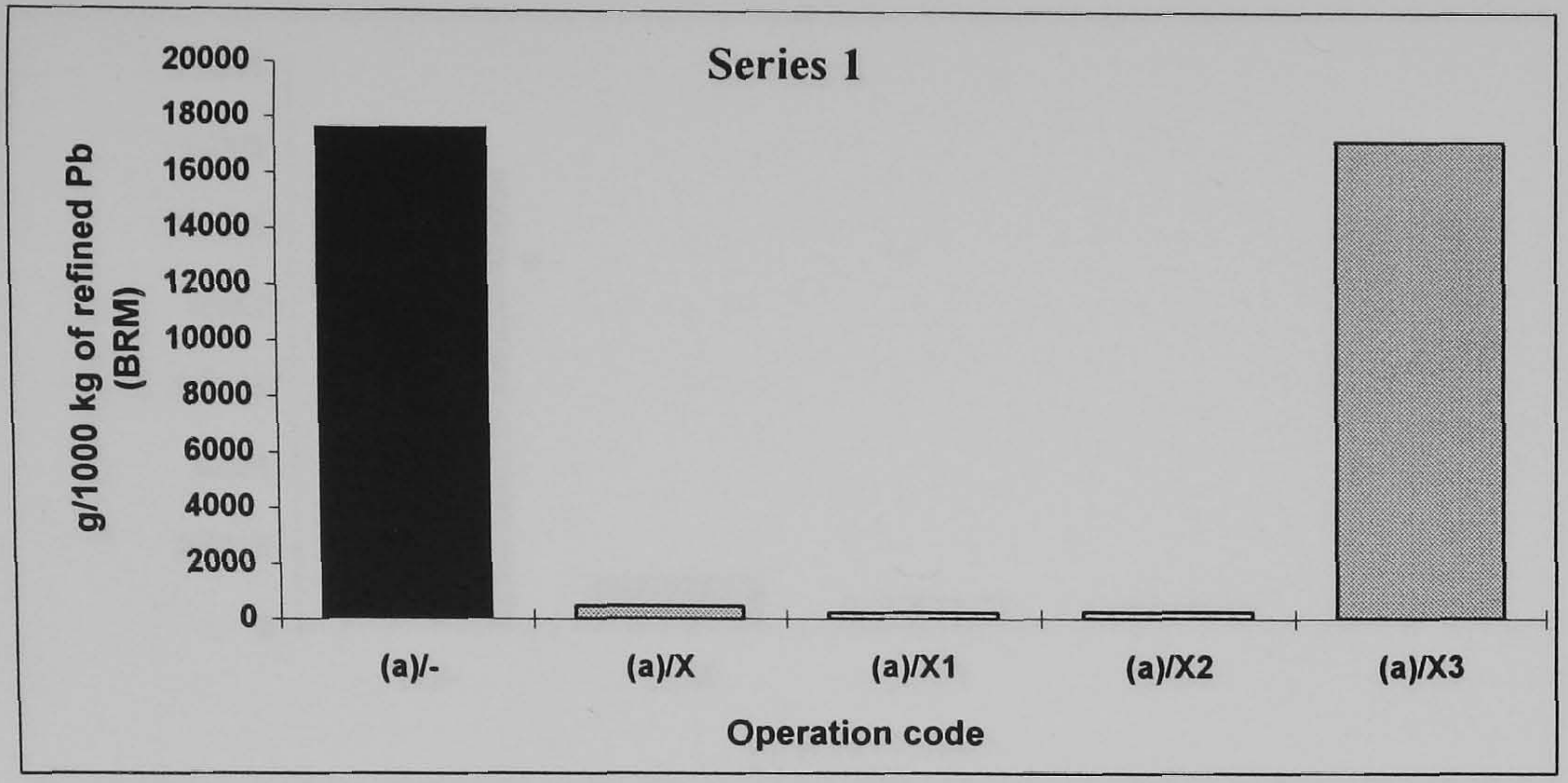


Code:	Operation:
(a)/X2	BRM: Refined Lead production Operations (Isa stream throughput)
(a)/X2,1	BRM: Lead Refining Operations (Isa stream)
(a)/X2,2	BRM: On-site Electricity Generation Operations (Isa floor throughput)
(a)/X2,3	BRM: Overheads (Isa stream operations)

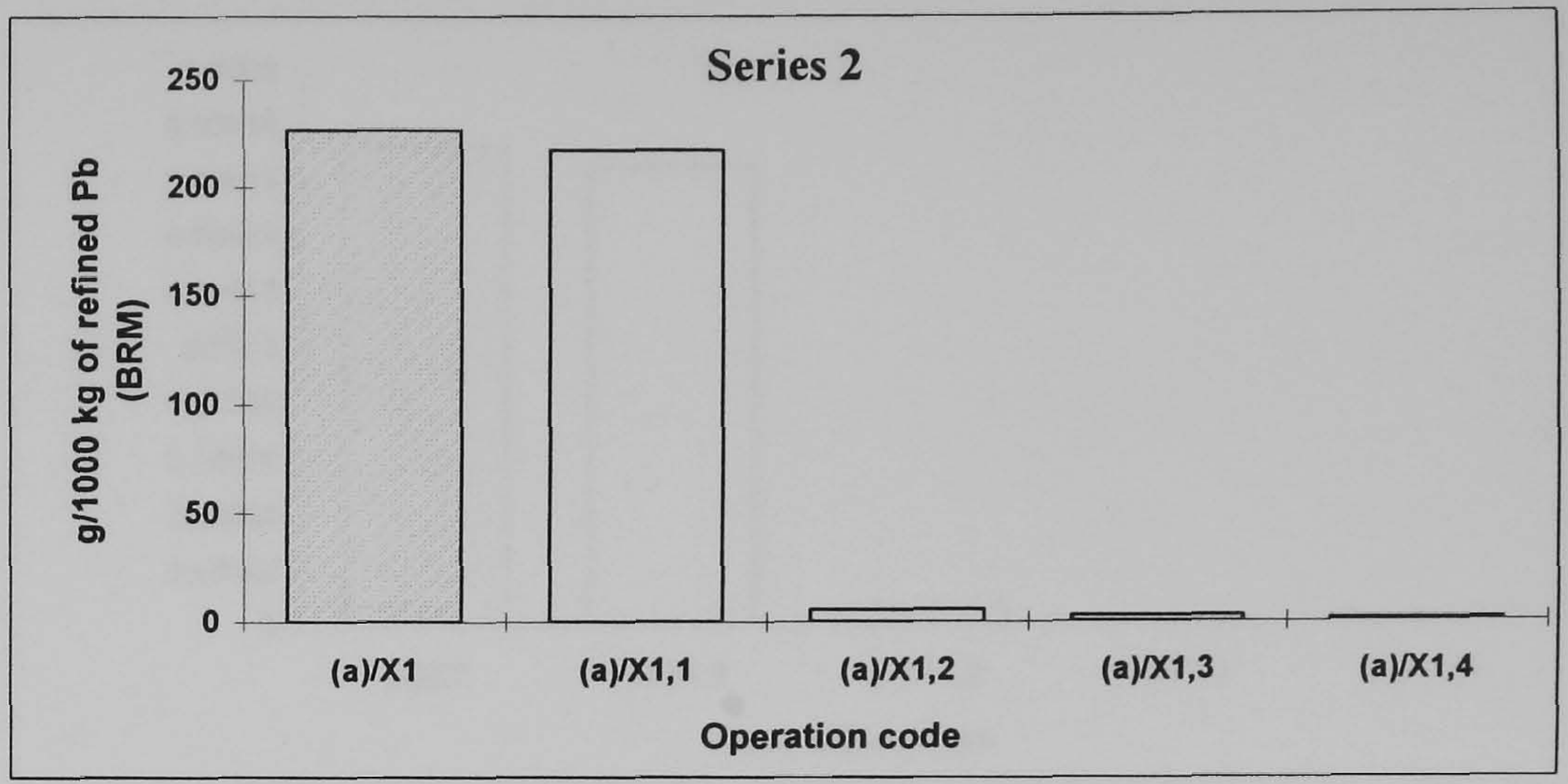


Code:	Operation:
(a)/Z	BRM: Secondary Refined Lead Production Operations
(a)/Z1	BRM (secondary): CX Plant Operations
(a)/Z2	BRM (secondary): Overheads of Secondary Processing Operations
(a)/Z3	BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations

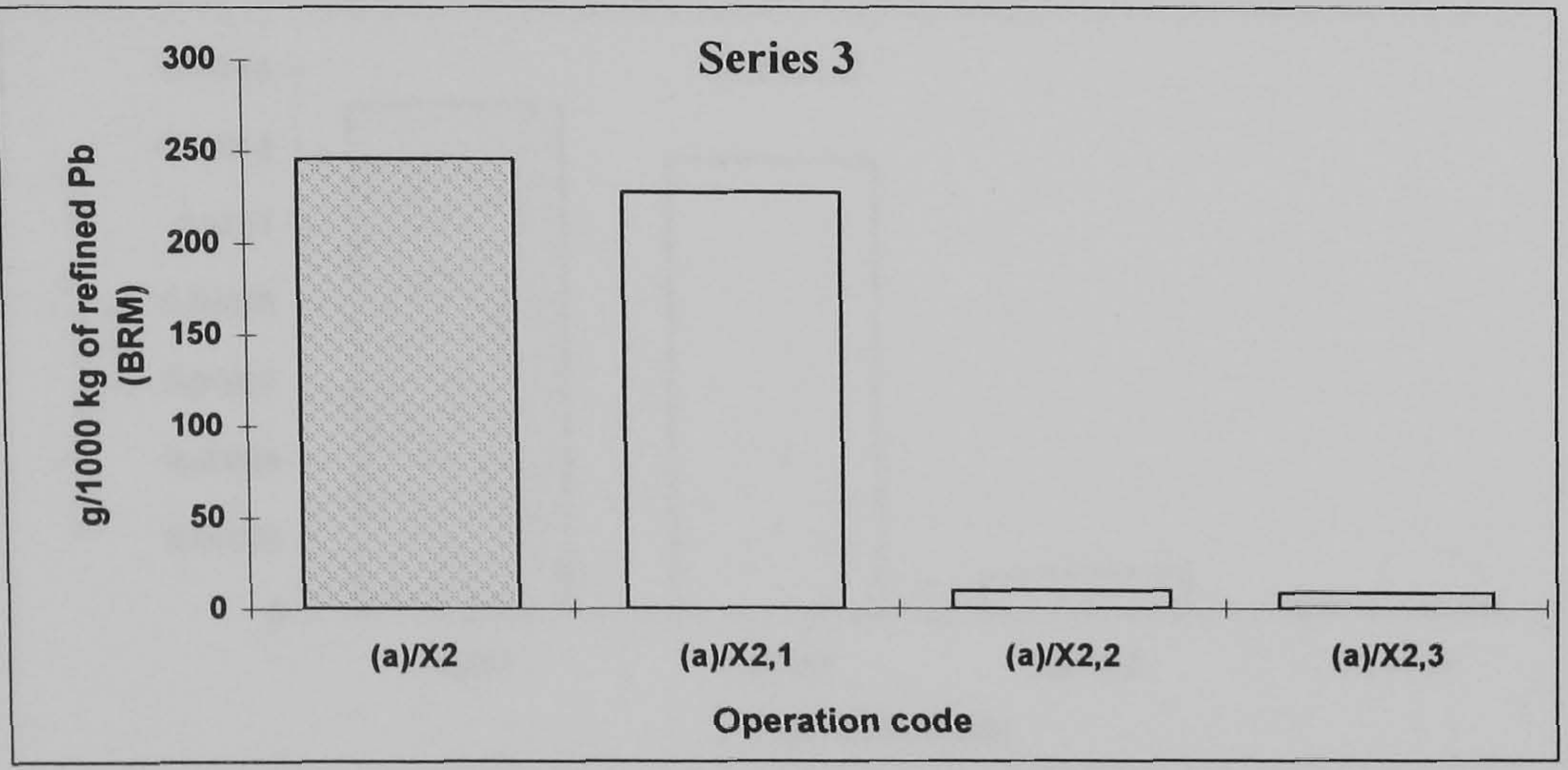
Graph 9.5: (a) Lead (Pb) -Modelled year (a). Series 1 to 4



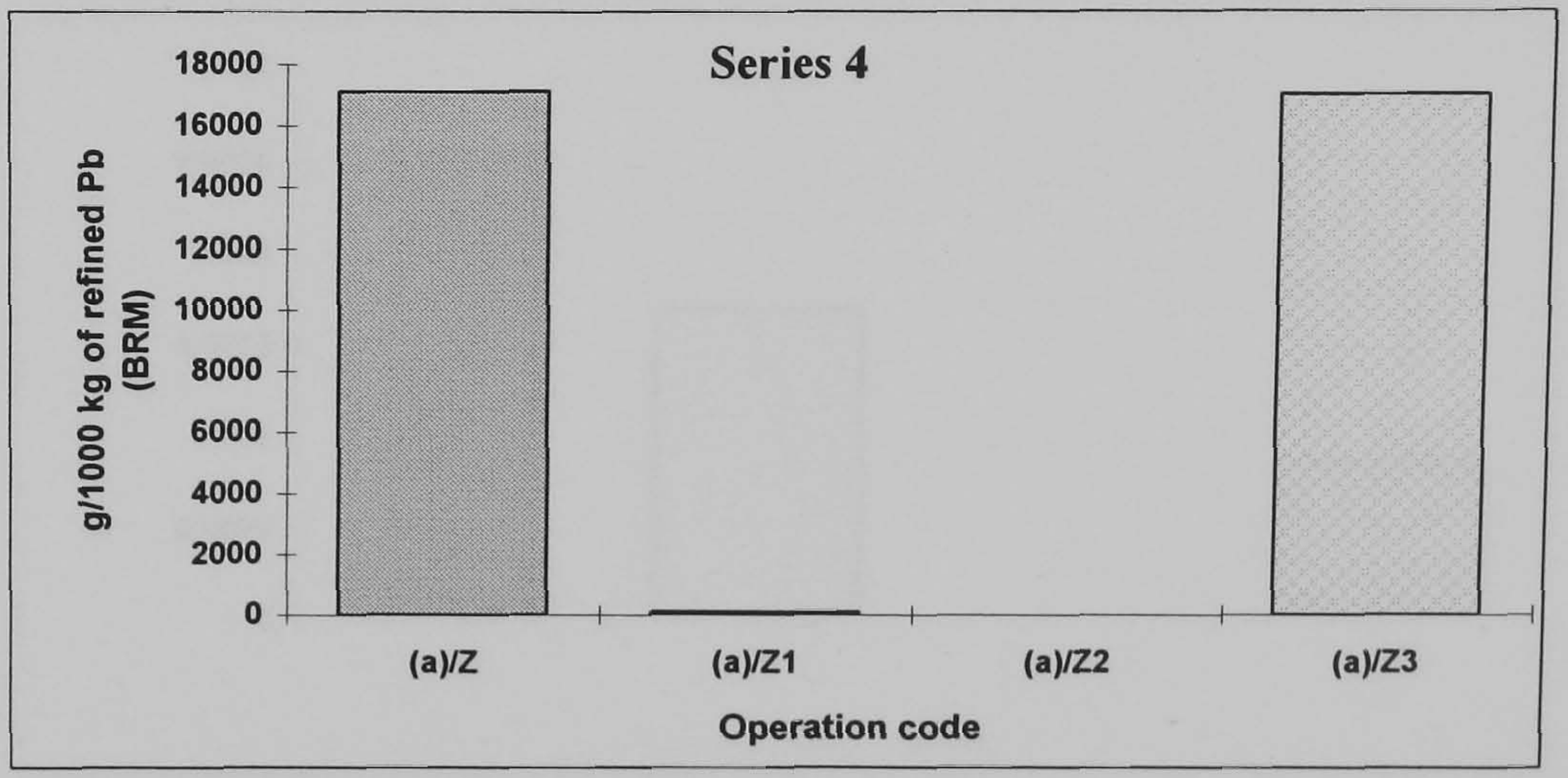
Code:	Operation:
(a)/-	BRM: Lead Material Processing Operations at Northfleet, Kent, UK
(a)/X	BRM: Primary Refined Lead Production Operations
(a)/X1	BRM: Refined Lead Production Operations (BRM stream throughput)
(a)/X2	BRM: Refined Lead production Operations (Isa stream throughput)
(a)/X3	BRM: Secondary Refined Lead Production Operations



Code:	Operation:
(a)/X1	BRM: Refined Lead Production Operations (BRM stream throughput)
(a)/X1,1	BRM: Lead Refining Operations (BRM floor throughput)
(a)/X1,2	BRM: On-site Electricity Generation Operations (BRM floor throughput)
(a)/X1,3	BRM: Overheads to BZL Input to BRM Floor Throughput Operations
(a)/X1,4	BRM: Overheads to MHD Input to BRM Floor Throughput Operations

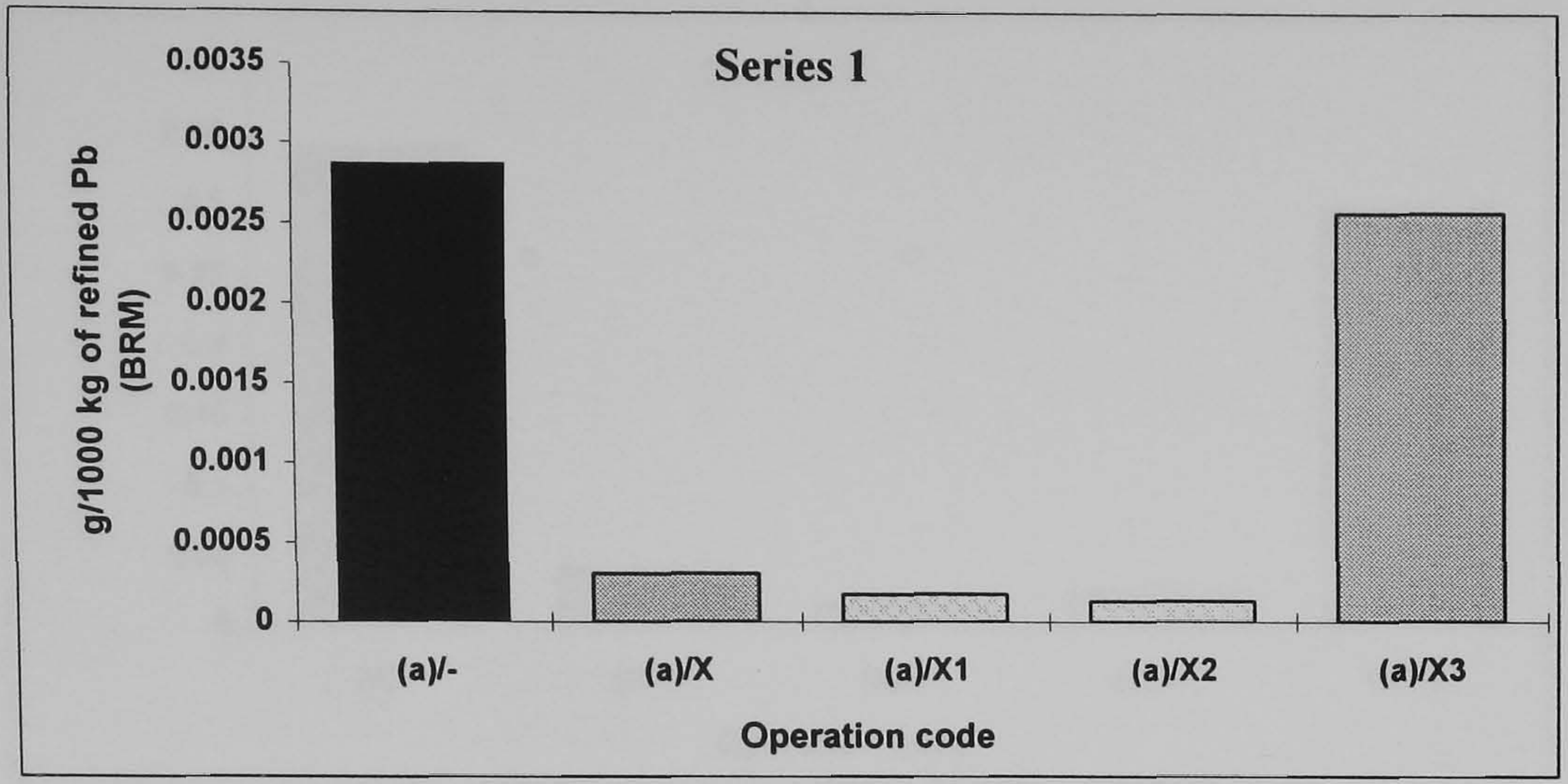


Code:	Operation:
(a)/X2	BRM: Refined Lead production Operations (Isa stream throughput)
(a)/X2,1	BRM: Lead Refining Operations (Isa stream)
(a)/X2,2	BRM: On-site Electricity Generation Operations (Isa floor throughput)
(a)/X2,3	BRM: Overheads (Isa stream operations)

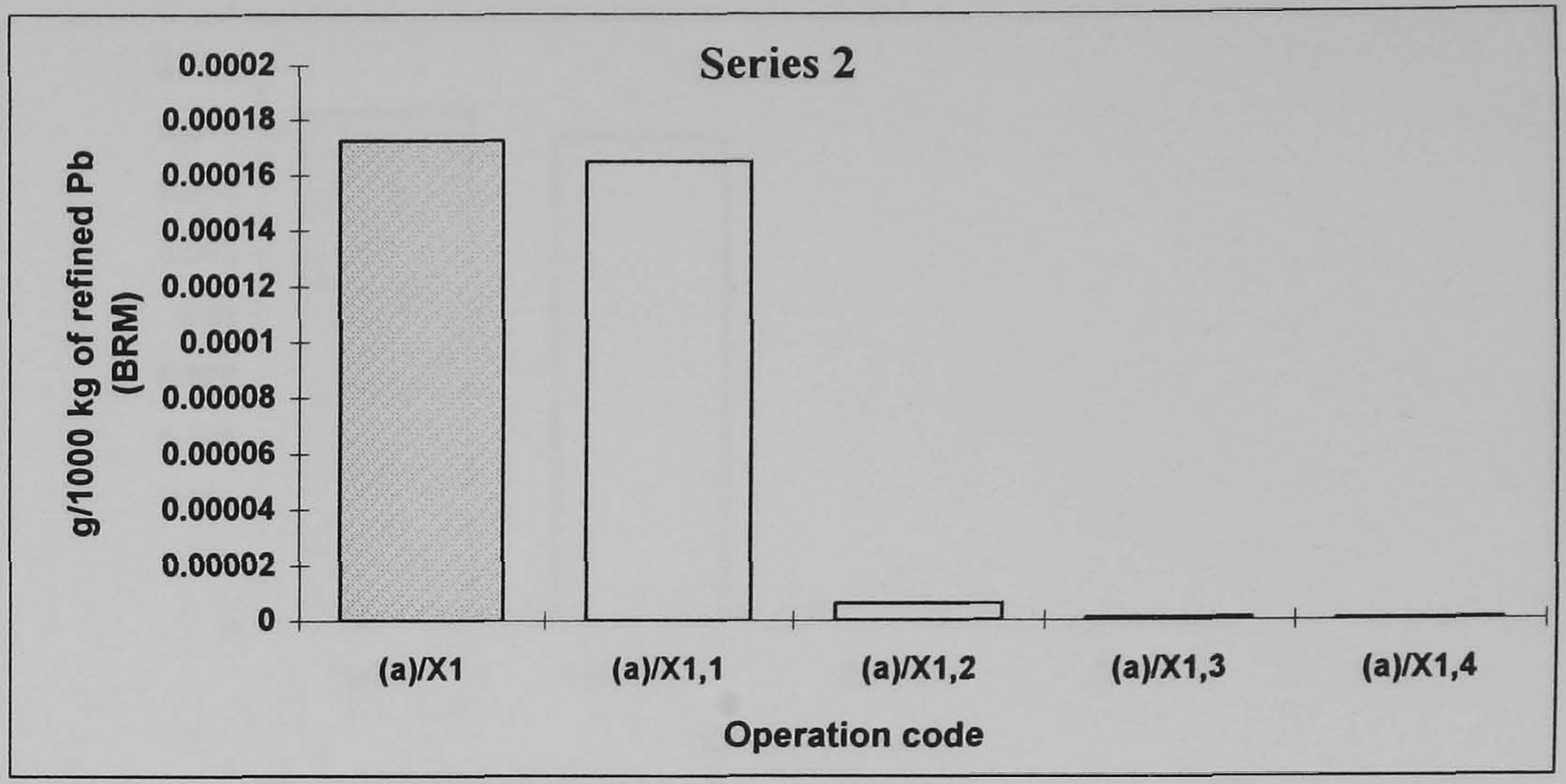


Code:	Operation:
(a)/Z	BRM: Secondary Refined Lead Production Operations
(a)/Z1	BRM (secondary): CX Plant Operations
(a)/Z2	BRM (secondary): Overheads of Secondary Processing Operations
(a)/Z3	BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations

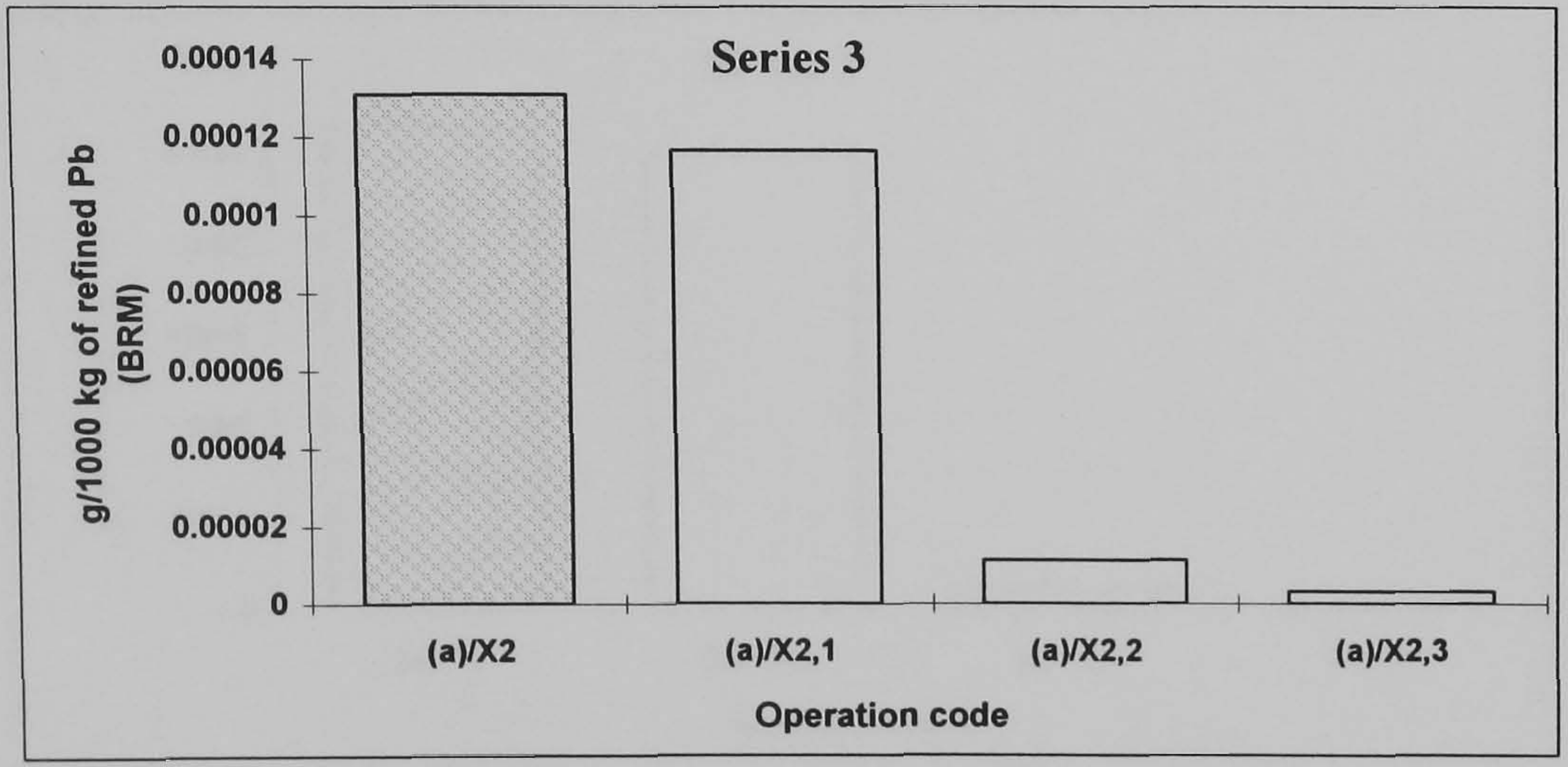
Graph 9.6: (a) Sulphur Oxides (SOx, as SO2) -Modelled year (a). Series 1 to 4



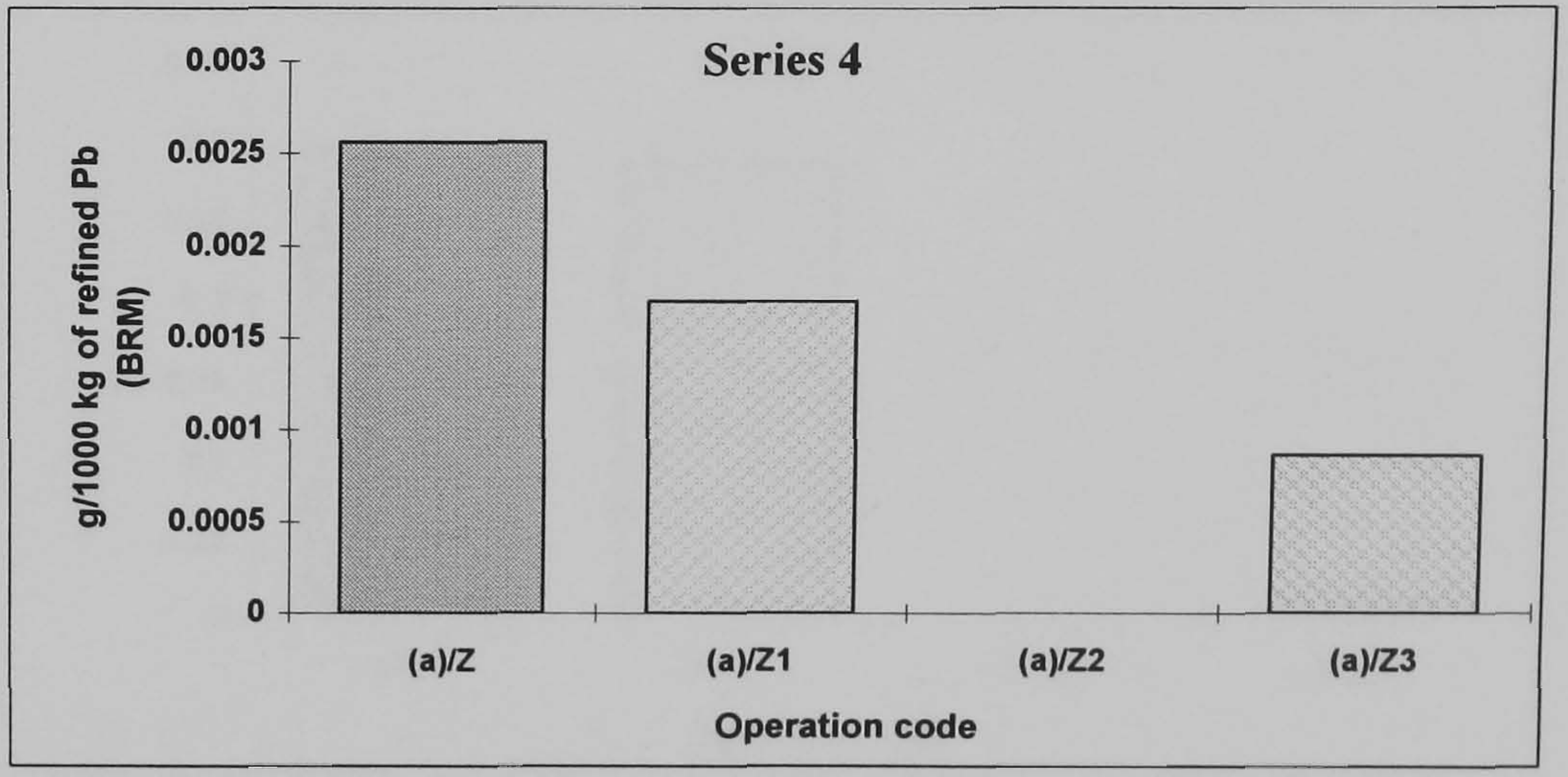
Code:	Operation:
(a)/-	BRM: Lead Material Processing Operations at Northfleet, Kent, UK
(a)/X	BRM: Primary Refined Lead Production Operations
(a)/X1	BRM: Refined Lead Production Operations (BRM stream throughput)
(a)/X2	BRM: Refined Lead production Operations (Isa stream throughput)
(a)/X3	BRM: Secondary Refined Lead Production Operations



Code:	Operation:
(a)/X1	BRM: Refined Lead Production Operations (BRM stream throughput)
(a)/X1,1	BRM: Lead Refining Operations (BRM floor throughput)
(a)/X1,2	BRM: On-site Electricity Generation Operations (BRM floor throughput)
(a)/X1,3	BRM: Overheads to BZL Input to BRM Floor Throughput Operations
(a)/X1,4	BRM: Overheads to MHD Input to BRM Floor Throughput Operations

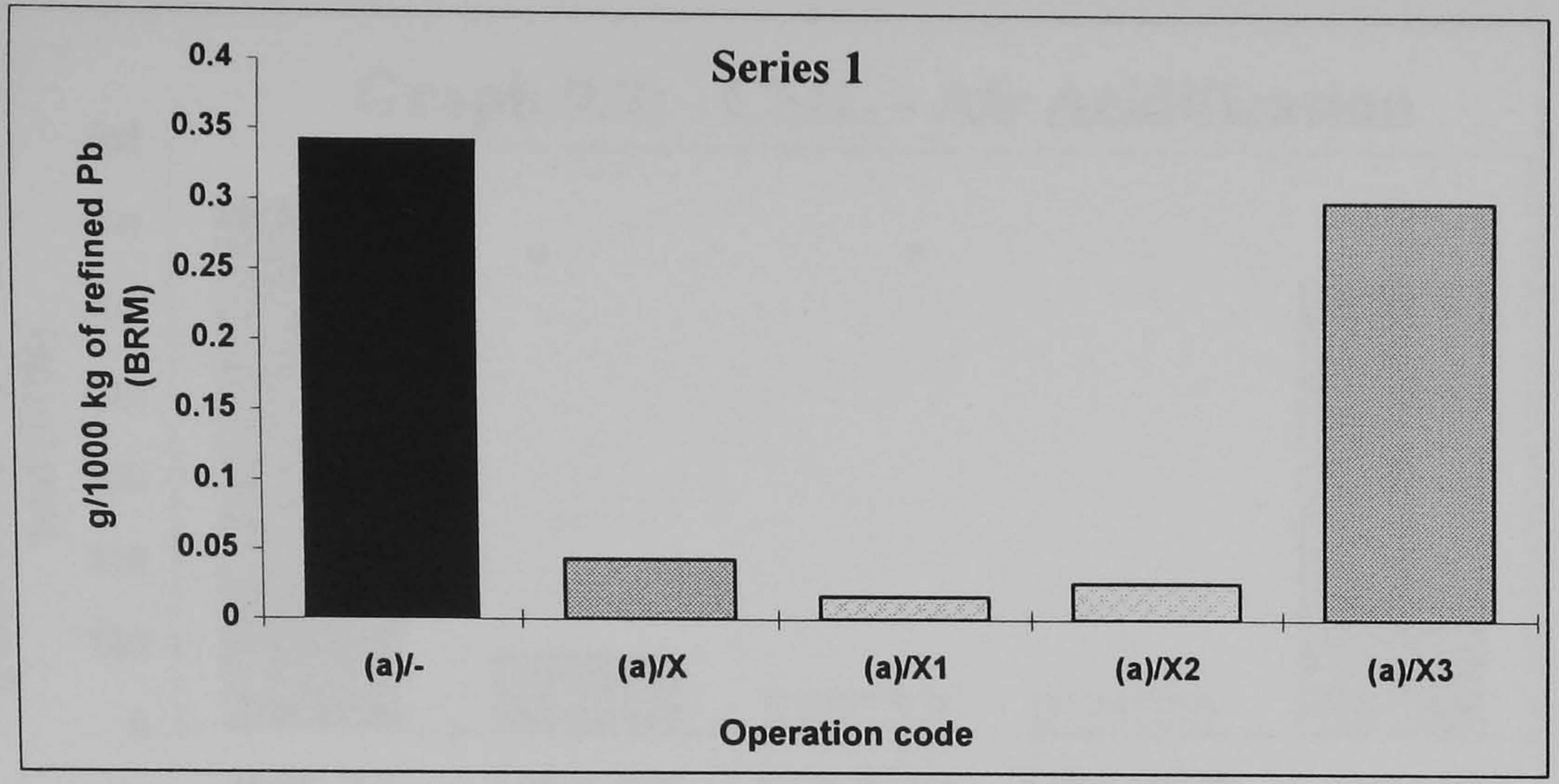


Code:	Operation:
(a)/X2	BRM: Refined Lead production Operations (Isa stream throughput)
(a)/X2,1	BRM: Lead Refining Operations (Isa stream)
(a)/X2,2	BRM: On-site Electricity Generation Operations (Isa floor throughput)
(a)/X2,3	BRM: Overheads (Isa stream operations)

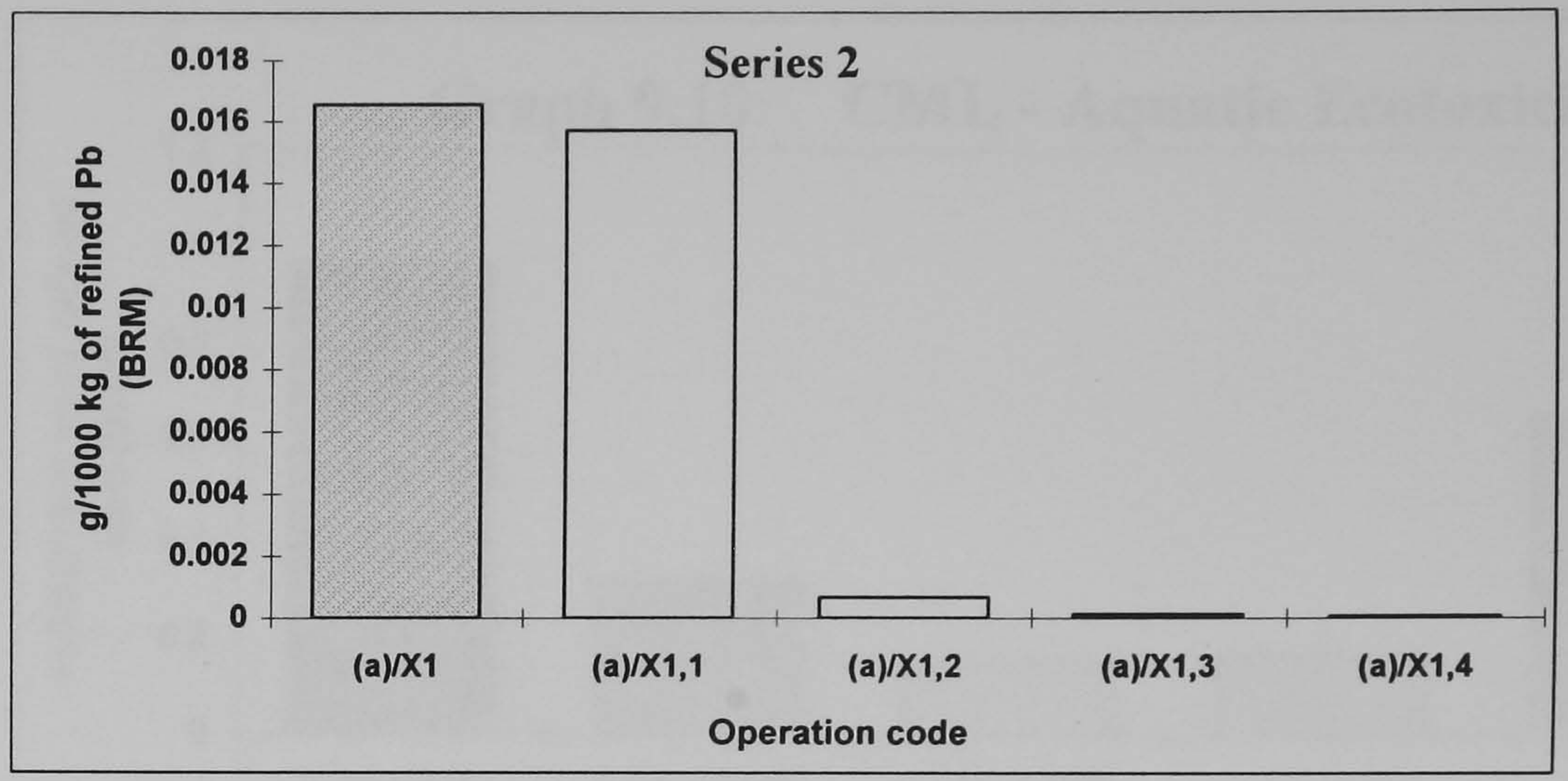


Code:	Operation:
(a)/Z	BRM: Secondary Refined Lead Production Operations
(a)/Z1	BRM (secondary): CX Plant Operations
(a)/Z2	BRM (secondary): Overheads of Secondary Processing Operations
(a)/Z3	BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations

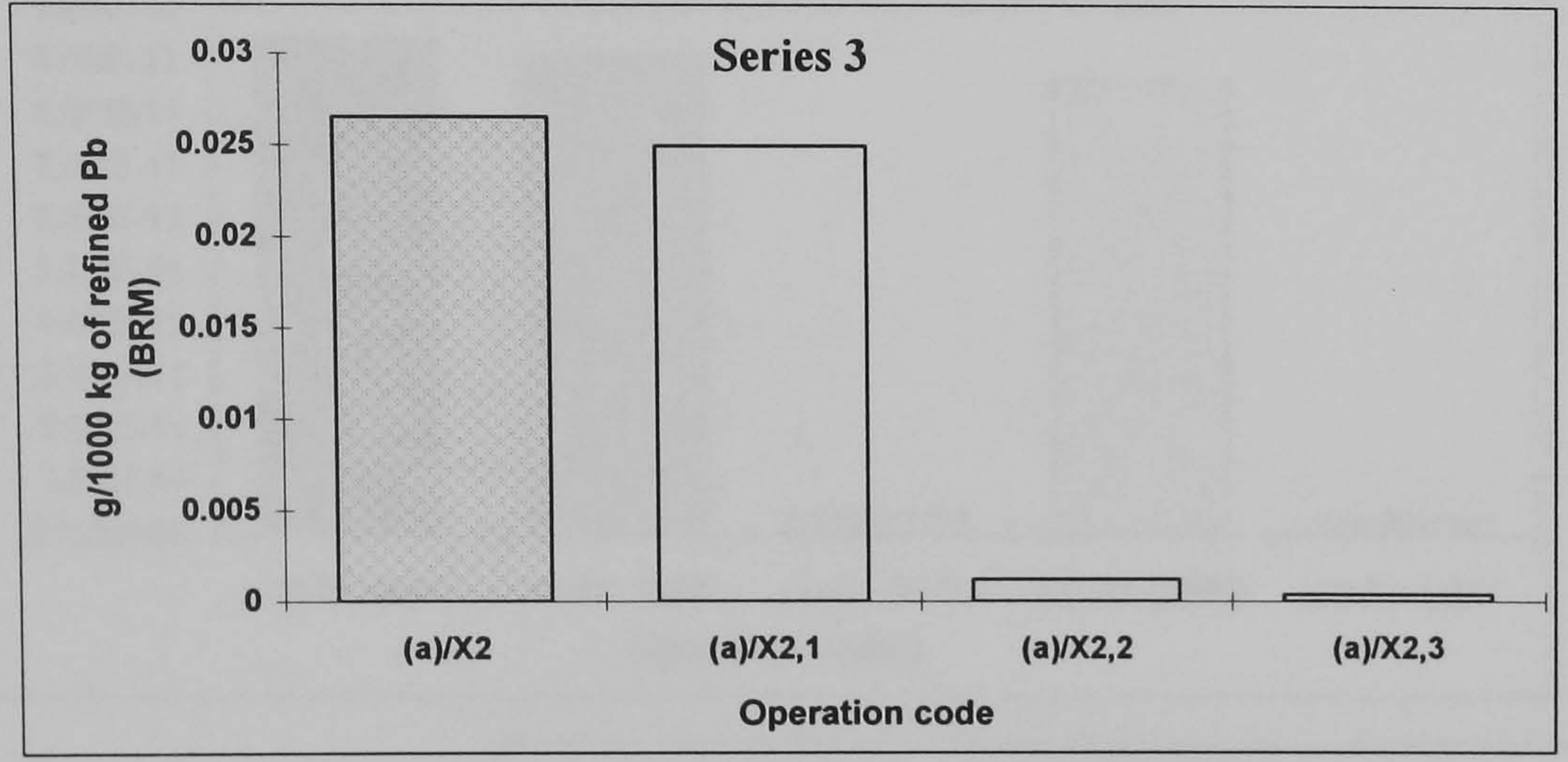
Graph 9.7: (w) Cadmium (Cd++) -Modelled year (a). Series 1 to 4



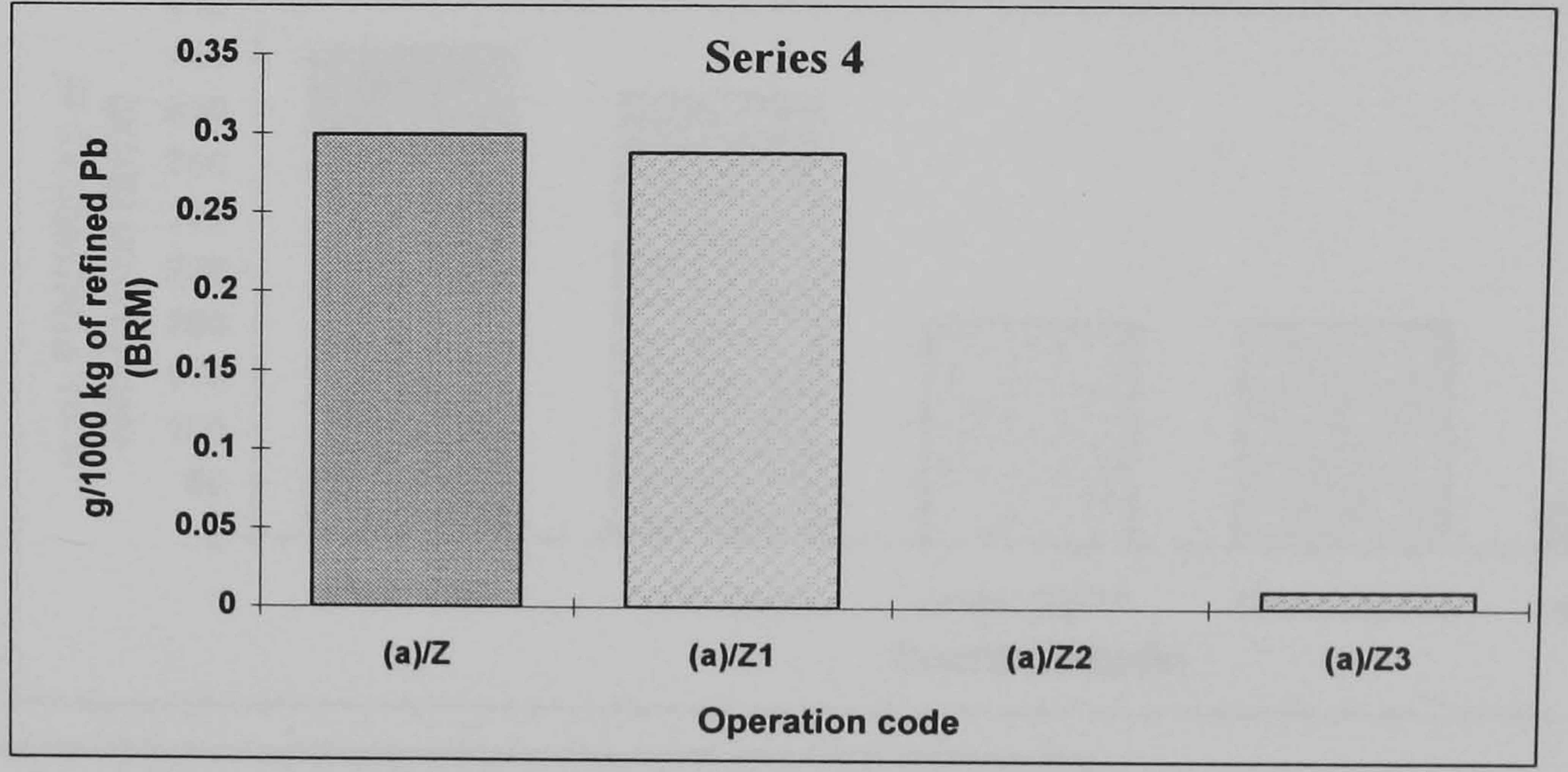
Code:	Operation:
(a)/-	BRM: Lead Material Processing Operations at Northfleet, Kent, UK
(a)/X	BRM: Primary Refined Lead Production Operations
(a)/X1	BRM: Refined Lead Production Operations (BRM stream throughput)
(a)/X2	BRM: Refined Lead production Operations (Isa stream throughput)
(a)/X3	BRM: Secondary Refined Lead Production Operations



Code:	Operation:
(a)/X1	BRM: Refined Lead Production Operations (BRM stream throughput)
(a)/X1,1	BRM: Lead Refining Operations (BRM floor throughput)
(a)/X1,2	BRM: On-site Electricity Generation Operations (BRM floor throughput)
(a)/X1,3	BRM: Overheads to BZL Input to BRM Floor Throughput Operations
(a)/X1,4	BRM: Overheads to MHD Input to BRM Floor Throughput Operations



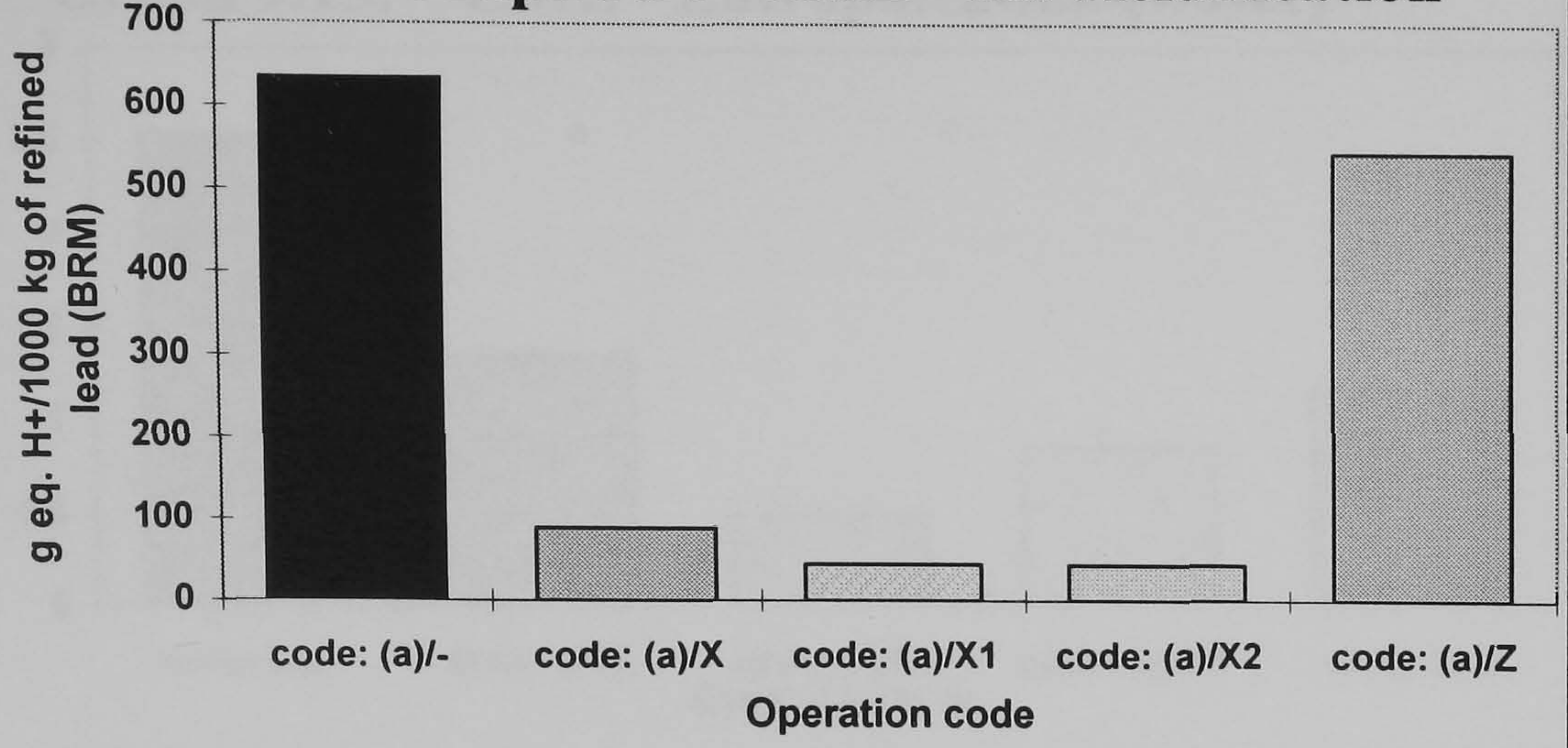
Code:	Operation:
(a)/X2	BRM: Refined Lead production Operations (Isa stream throughput)
(a)/X2,1	BRM: Lead Refining Operations (Isa stream)
(a)/X2,2	BRM: On-site Electricity Generation Operations (Isa floor throughput)
(a)/X2,3	BRM: Overheads (Isa stream operations)



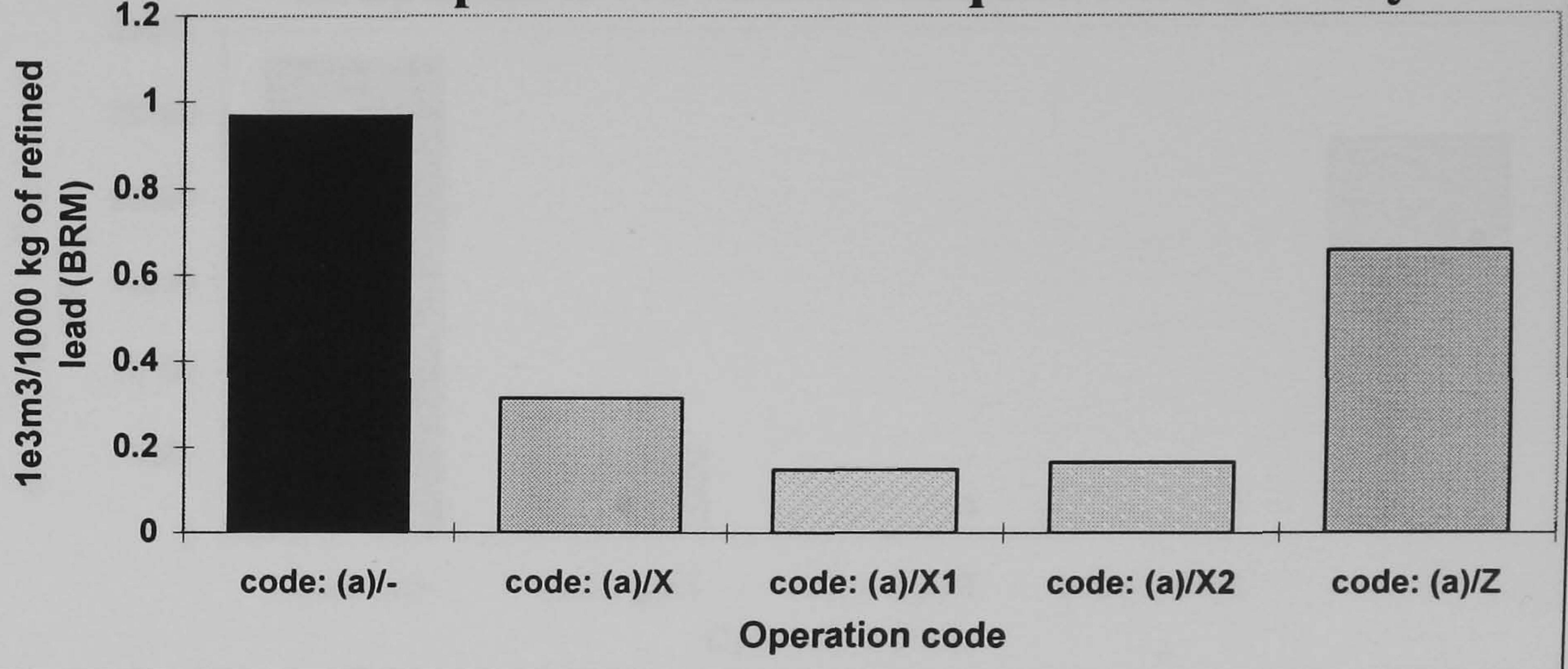
Code:	Operation:
(a)/Z	BRM: Secondary Refined Lead Production Operations
(a)/Z1	BRM (secondary): CX Plant Operations
(a)/Z2	BRM (secondary): Overheads of Secondary Processing Operations
(a)/Z3	BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations

Graph 9.8: (w) Lead (Pb++, Pb4+) -Modelled year (a). Series 1 to 4

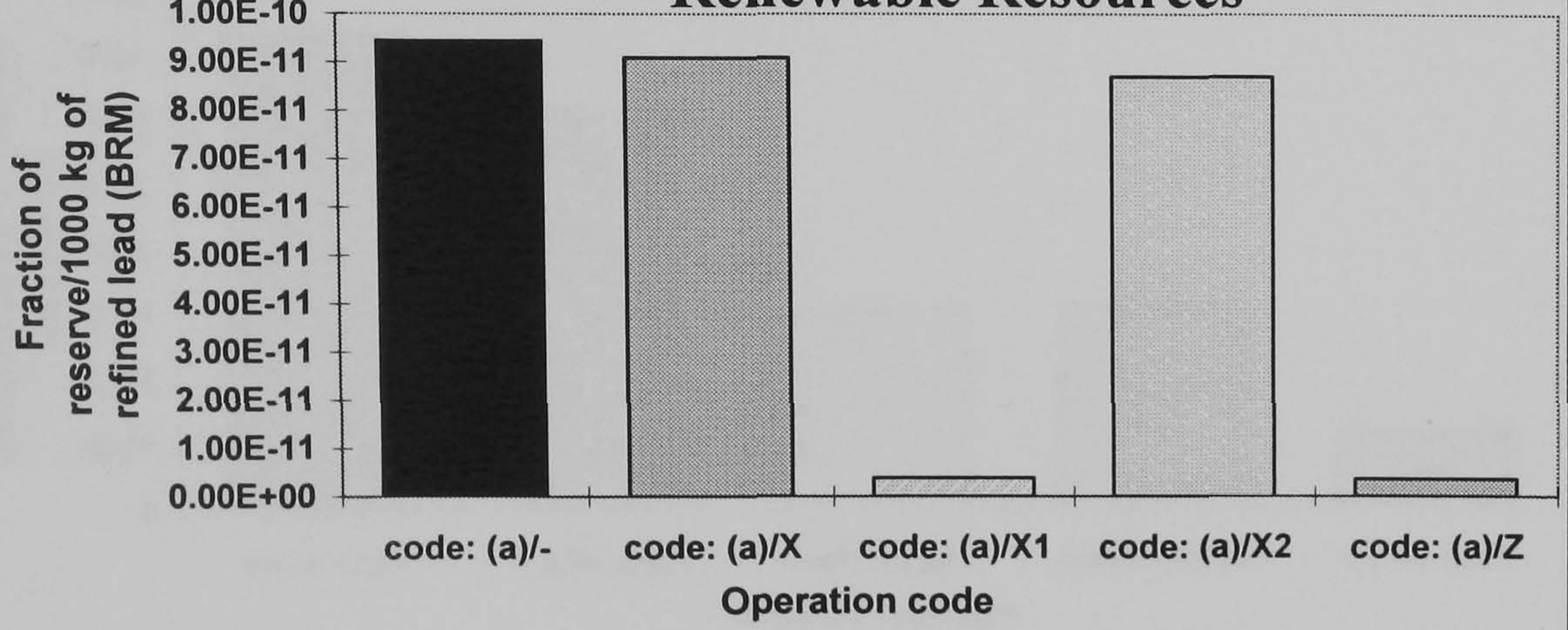
Graph 9.9: CML - Air Acidification



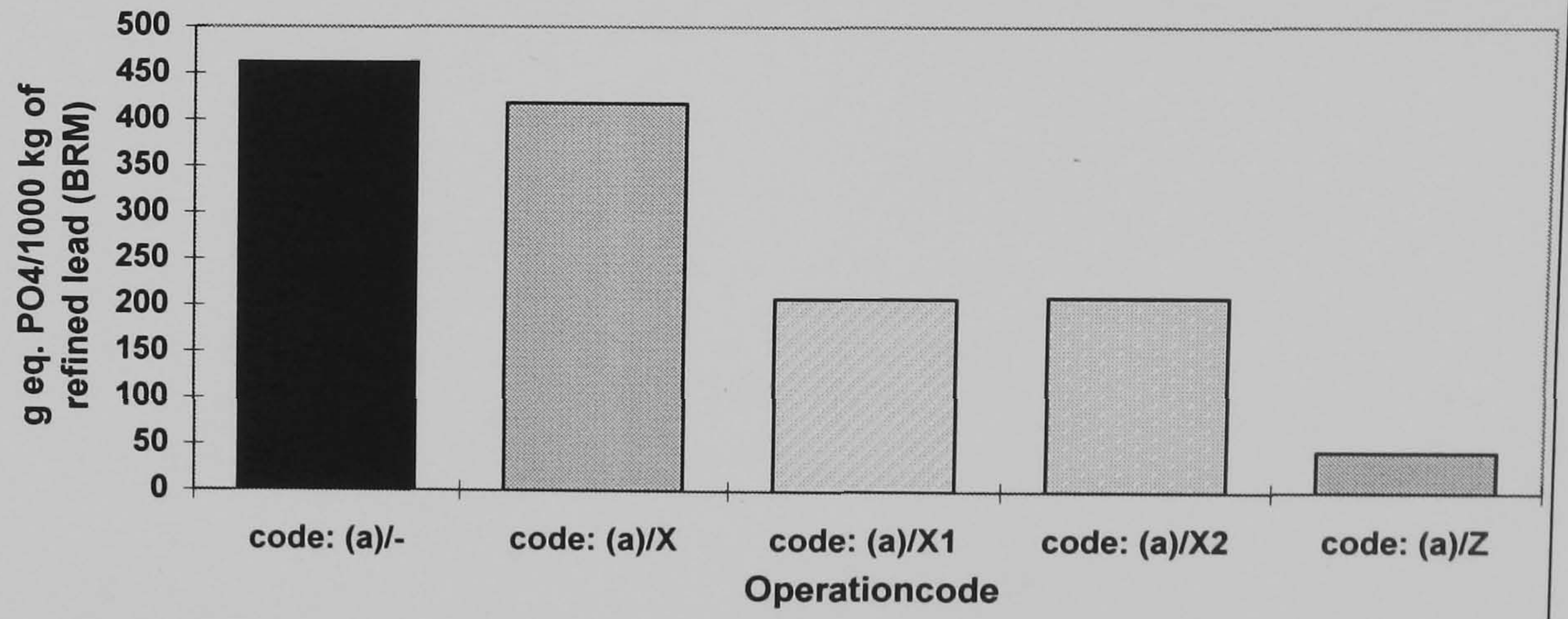
Graph 9.10: CML - Aquatic Ecotoxicity



Graph 9.11: CML - Depletion of Non-Renewable Resources

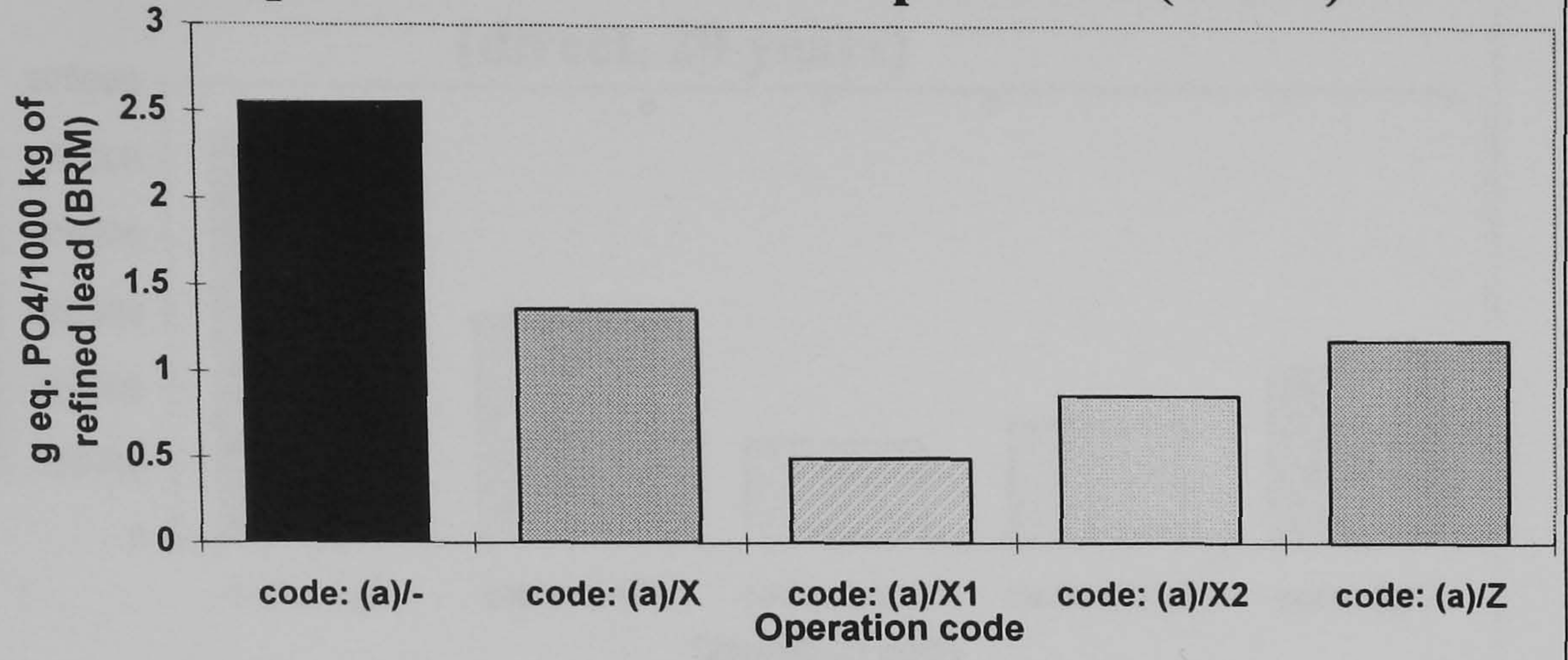


Graph 9.12: CML - Eutrophication

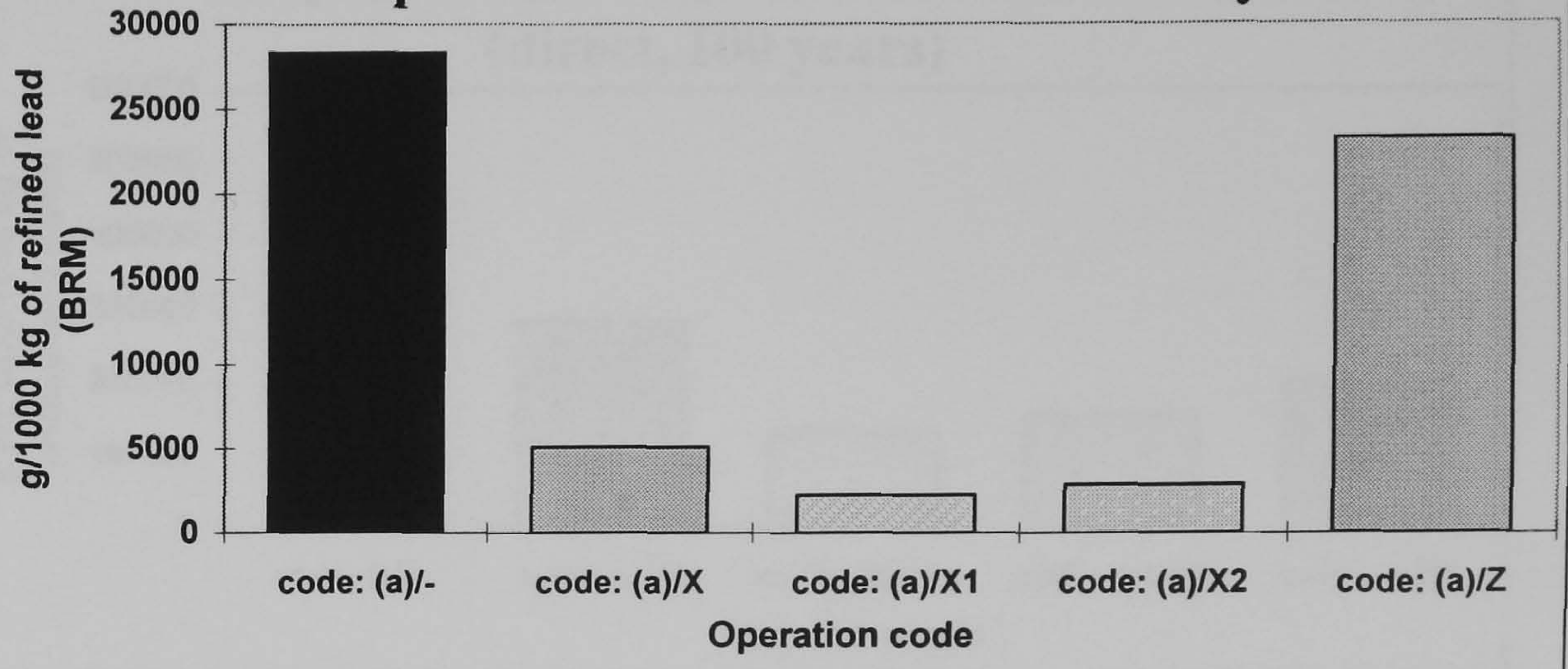


Code:	Operation:
(a)/-	BRM: Lead Material Processing Operations at Northfleet, Kent, UK
(a)/X	BRM: Primary Refined Lead Production Operations
(a)/X1	BRM: Refined Lead Production Operations (BRM stream throughput)
(a)/X2	BRM: Refined Lead production Operations (Isa stream throughput)
(a)/Z	BRM: Secondary Refined Lead Production Operations

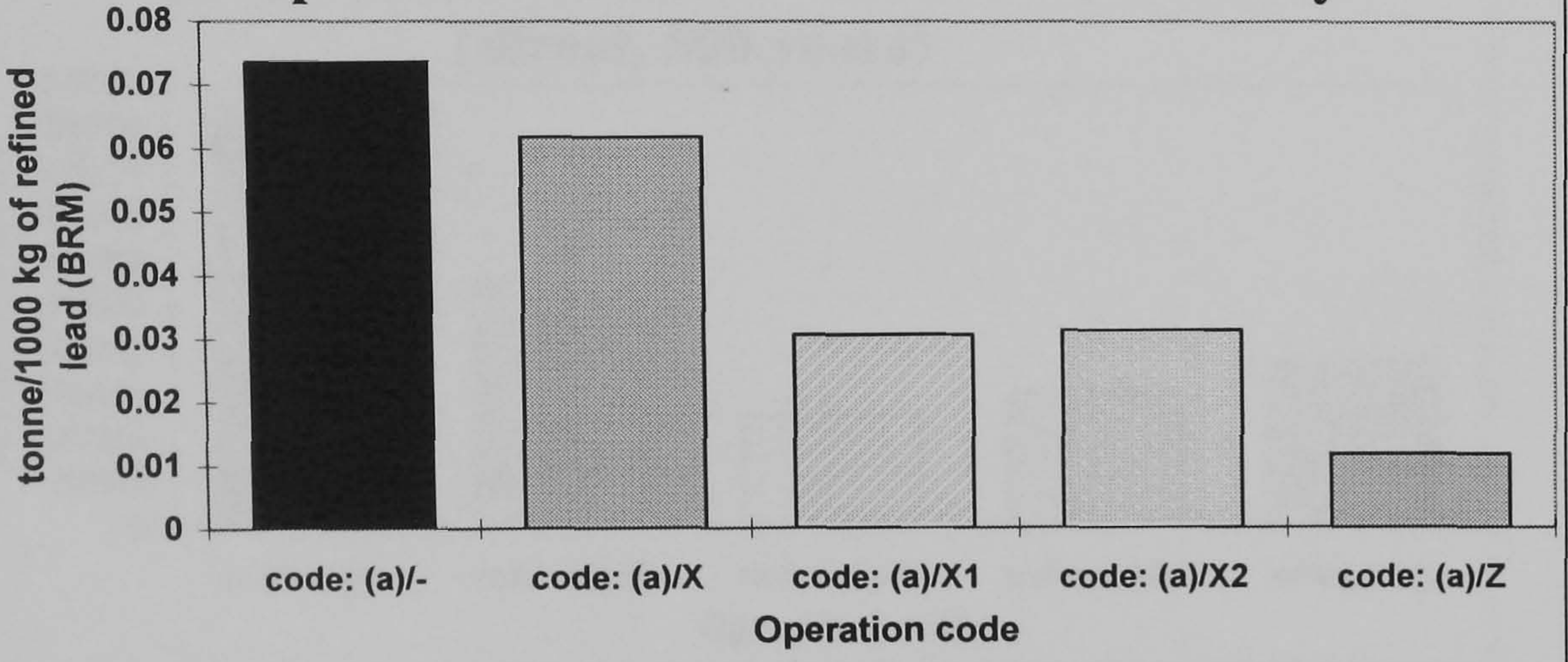
Graph 9.13: CML - Eutrophication (water)



Graph 9.14: CML - Human Toxicity

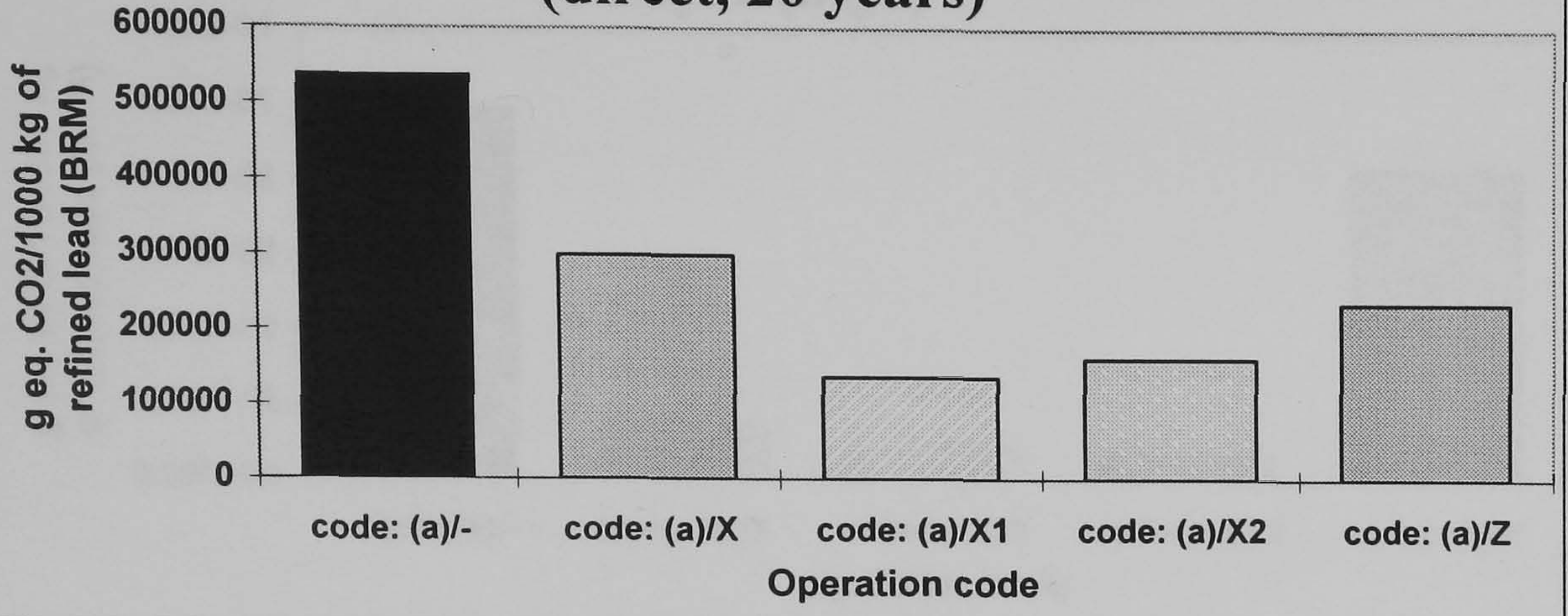


Graph 9.15: CML - Terrestrial Ecotoxicity

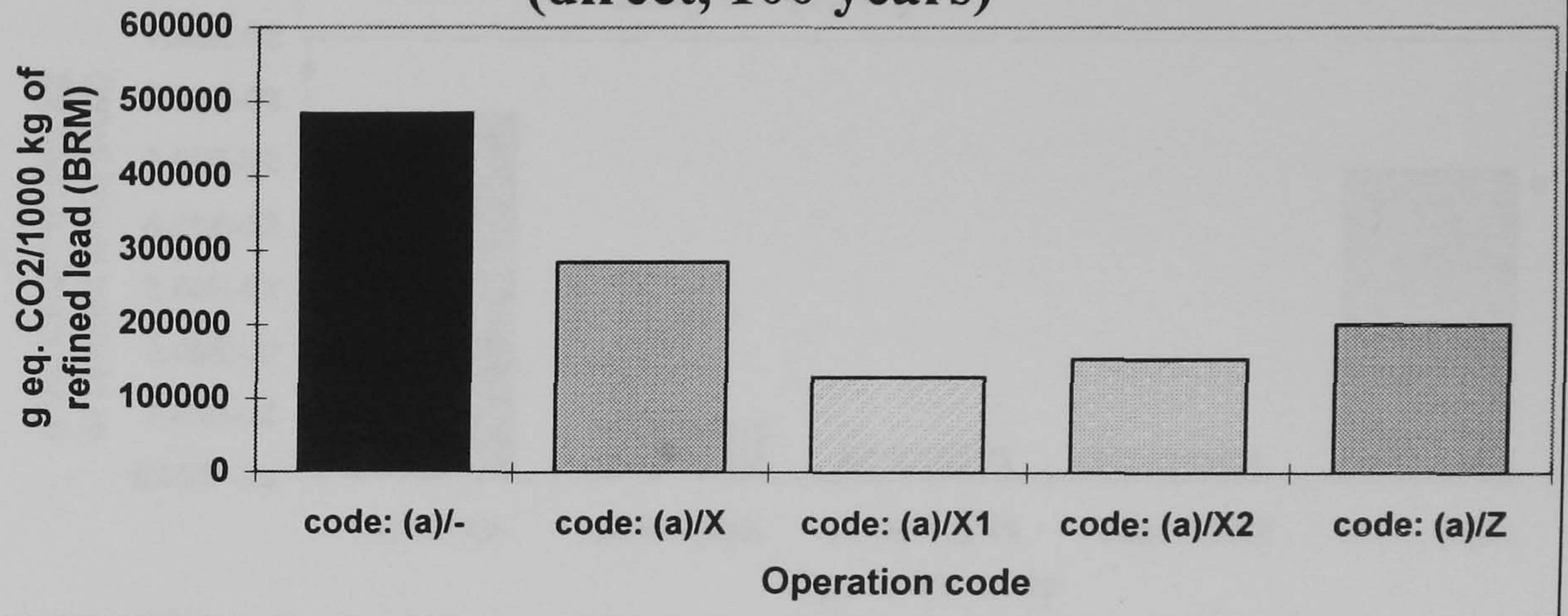


Code:	Operation:
(a)/-	BRM: Lead Material Processing Operations at Northfleet, Kent, UK
(a)/X	BRM: Primary Refined Lead Production Operations
(a)/X1	BRM: Refined Lead Production Operations (BRM stream throughput)
(a)/X2	BRM: Refined Lead production Operations (Isa stream throughput)
(a)/Z	BRM: Secondary Refined Lead Production Operations

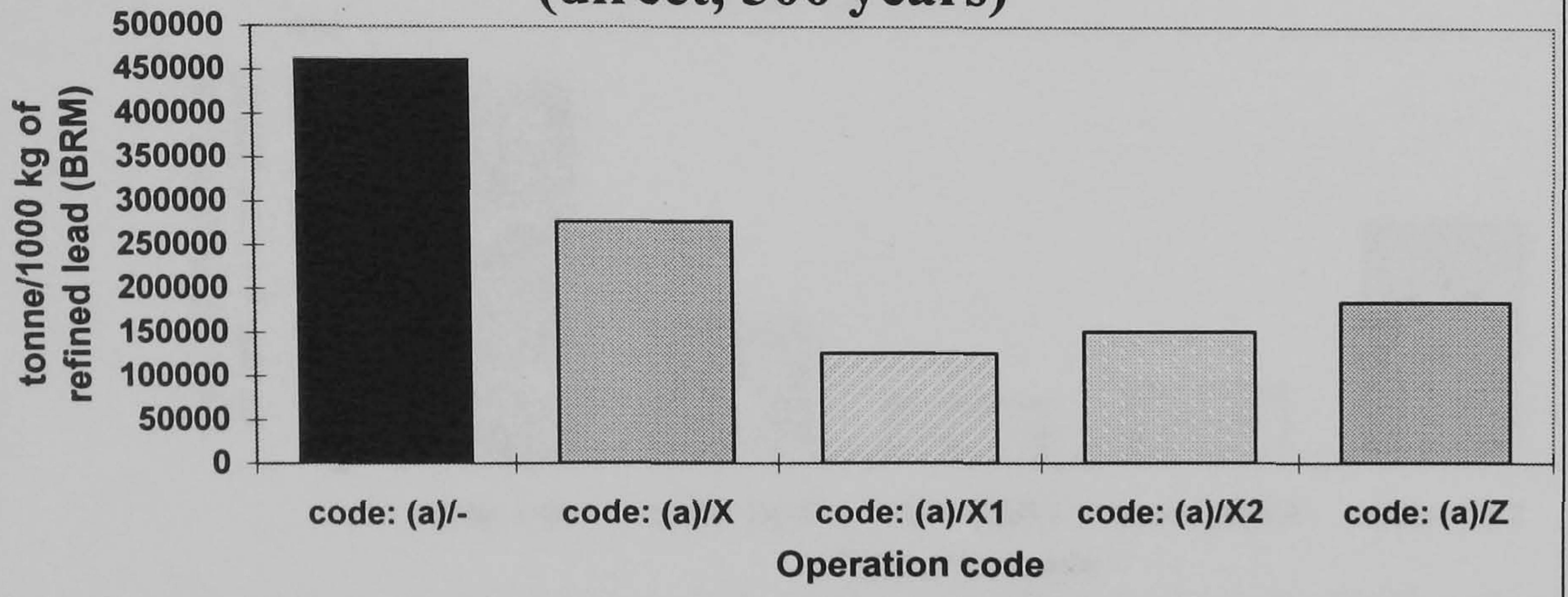
Graph 9.16: IPCC - Greenhouse Effect (direct, 20 years)



Graph 9.17: IPCC - Greenhouse Effect (direct, 100 years)

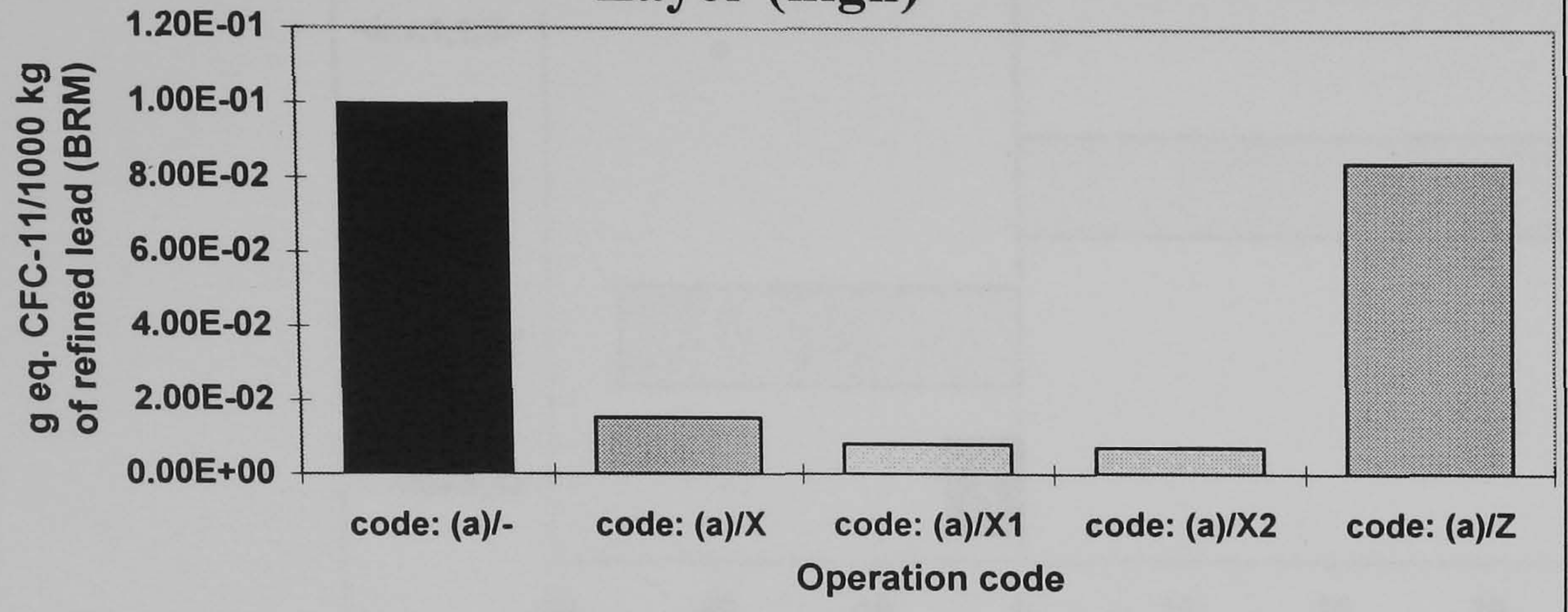


Graph 9.18: IPCC - Greenhouse Effect (direct, 500 years)

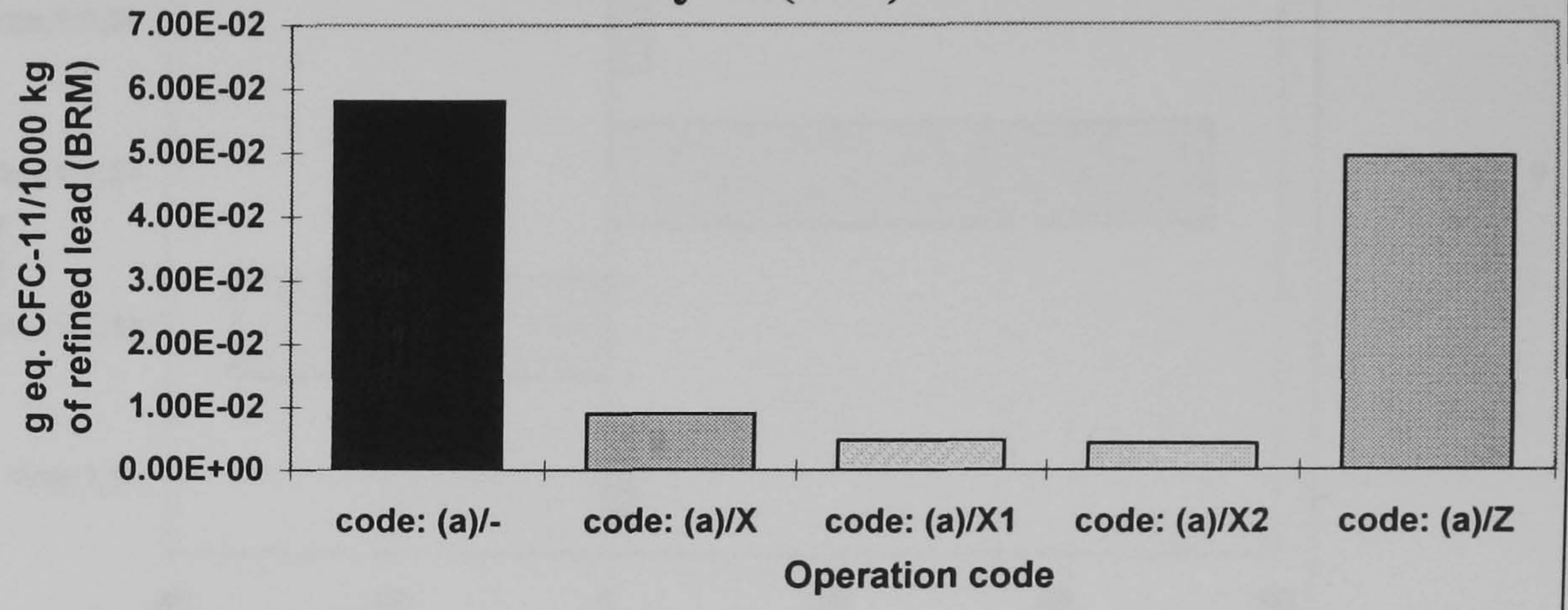


Code:	Operation:
(a)/-	BRM: Lead Material Processing Operations at Northfleet, Kent, UK
(a)/X	BRM: Primary Refined Lead Production Operations
(a)/X1	BRM: Refined Lead Production Operations (BRM stream throughput)
(a)/X2	BRM: Refined Lead production Operations (Isa stream throughput)
(a)/Z	BRM: Secondary Refined Lead Production Operations

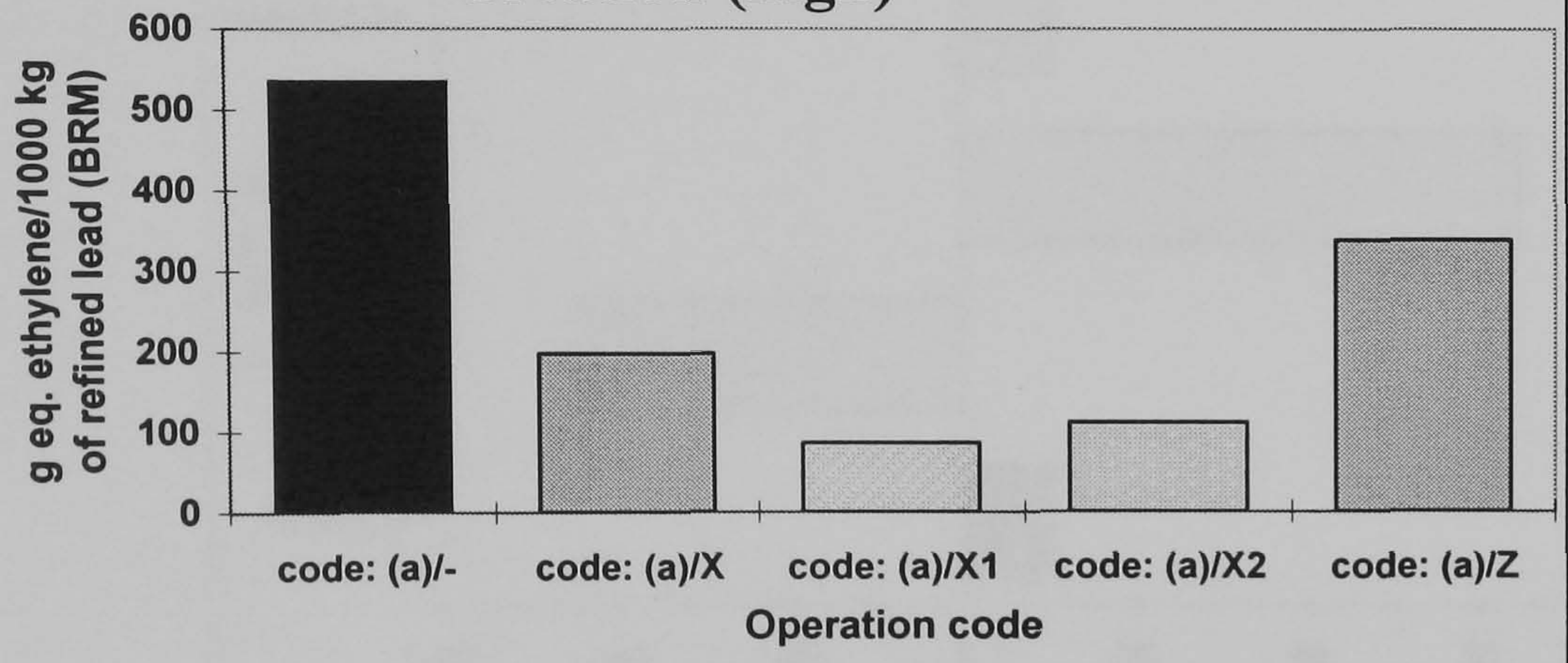
Graph 9.19: WMO - Depletion of the Ozone Layer (high)



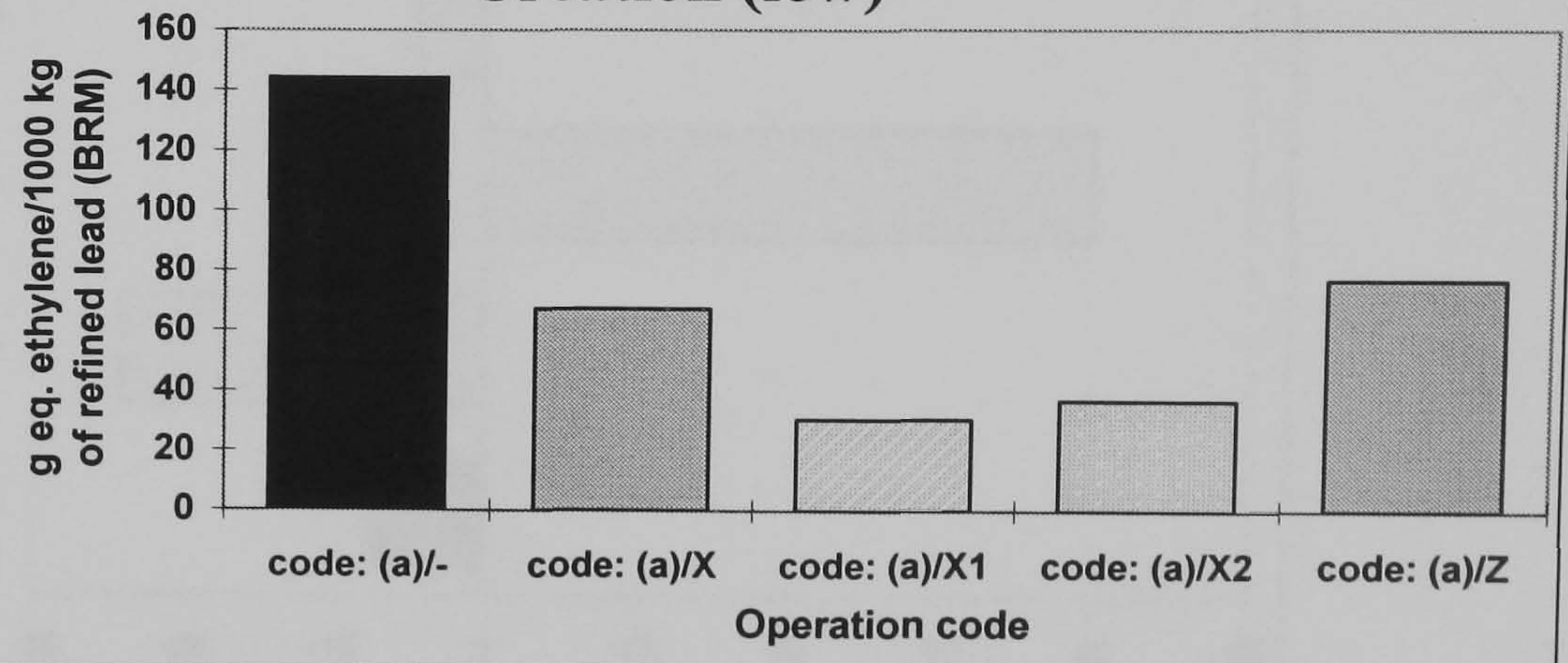
Graph 9.20: WMO - Depletion of the Ozone Layer (low)



Graph 9.21: WMO - Photochemical Oxidant Creation (high)



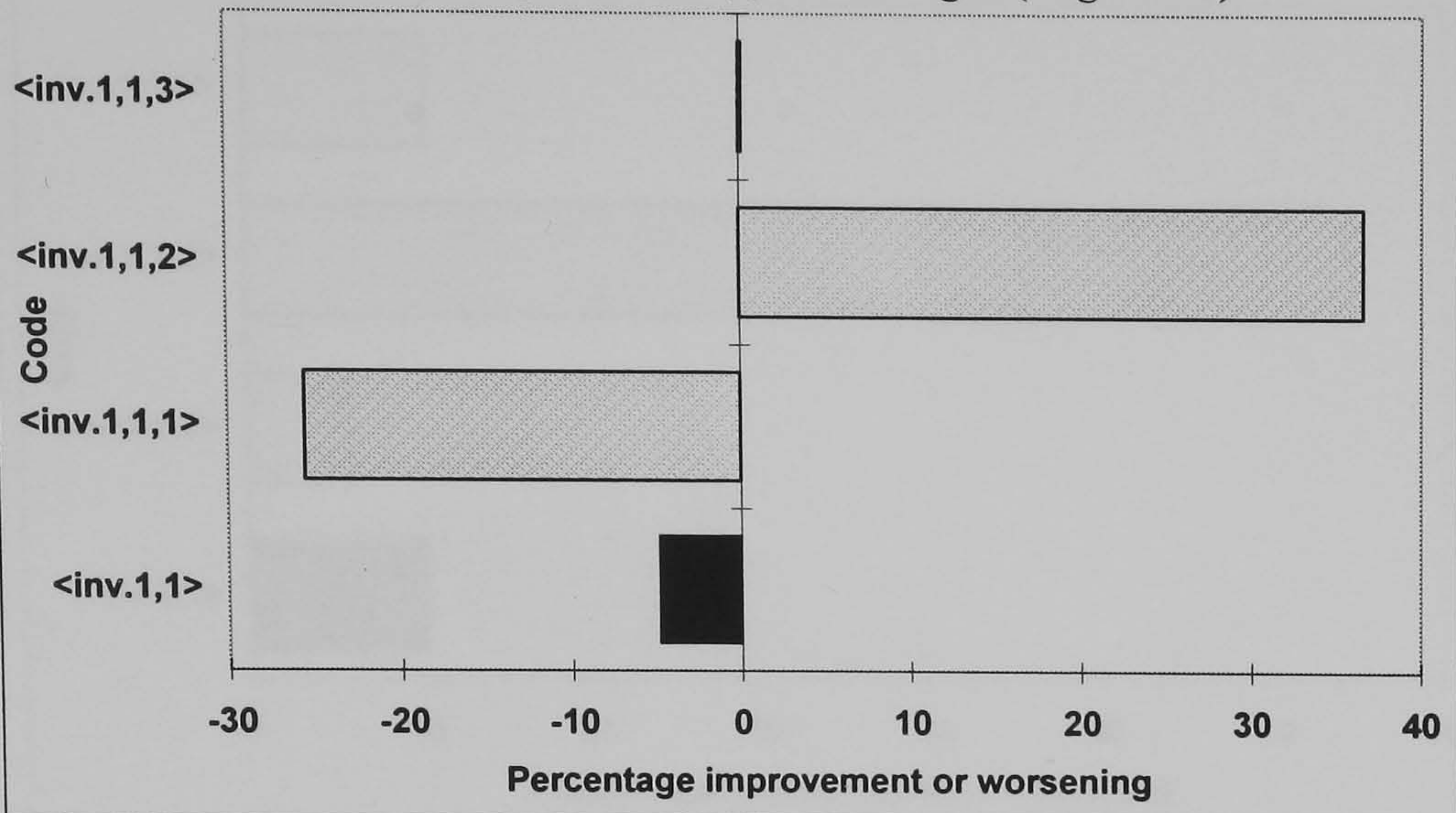
Graph 9.22: WMO - Photochemical Oxidant Creation (low)



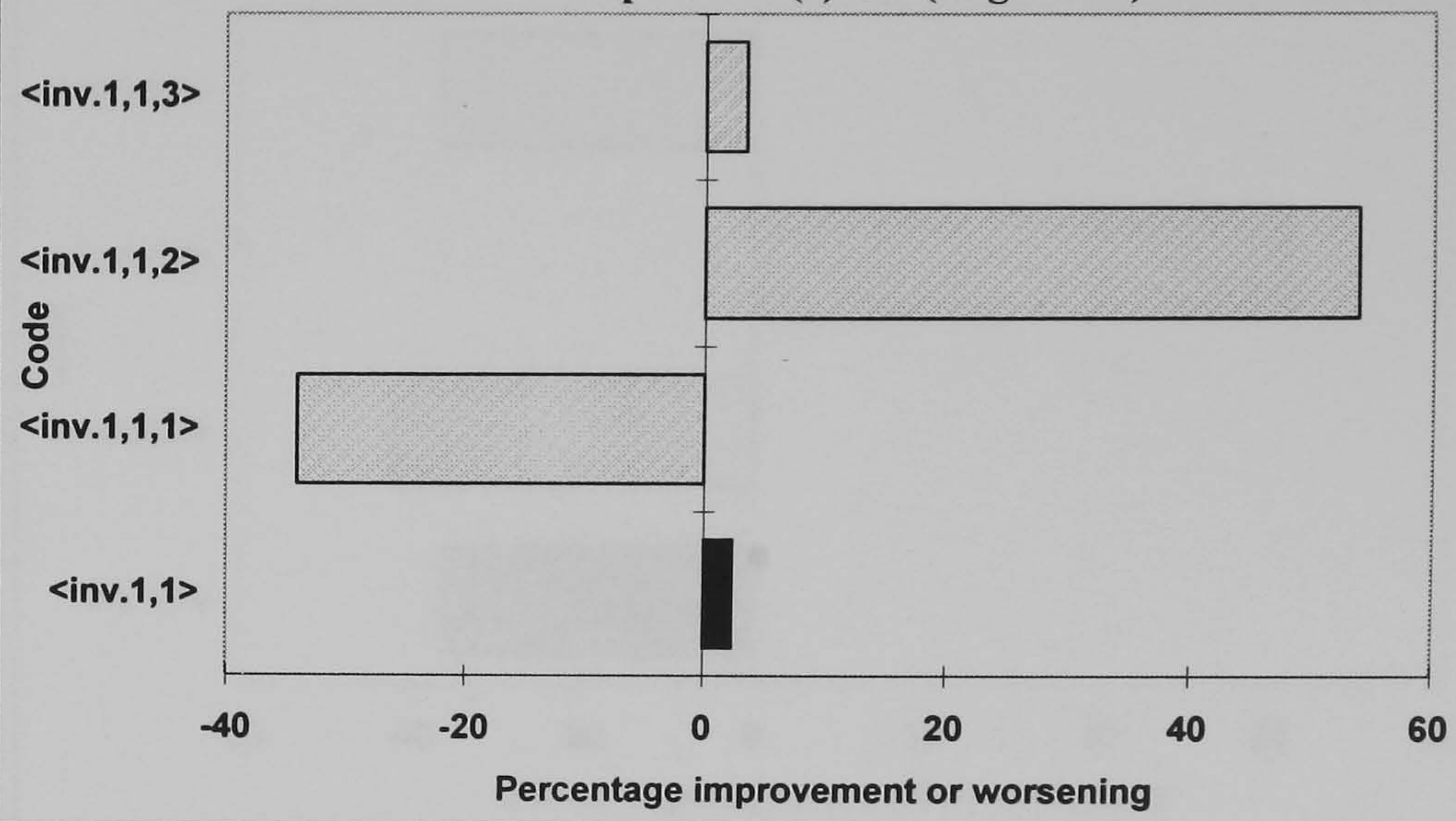
<u>Code</u>:	<u>Operation</u>:
(a)/-	BRM: Lead Material Processing Operations at Northfleet, Kent, UK
(a)/X	BRM: Primary Refined Lead Production Operations
(a)/X1	BRM: Refined Lead Production Operations (BRM stream throughput)
(a)/X2	BRM: Refined Lead production Operations (Isa stream throughput)
(a)/Z	BRM: Secondary Refined Lead Production Operations

Graphs comparing inventories for the BRM company year 1998-9 with 1997-8

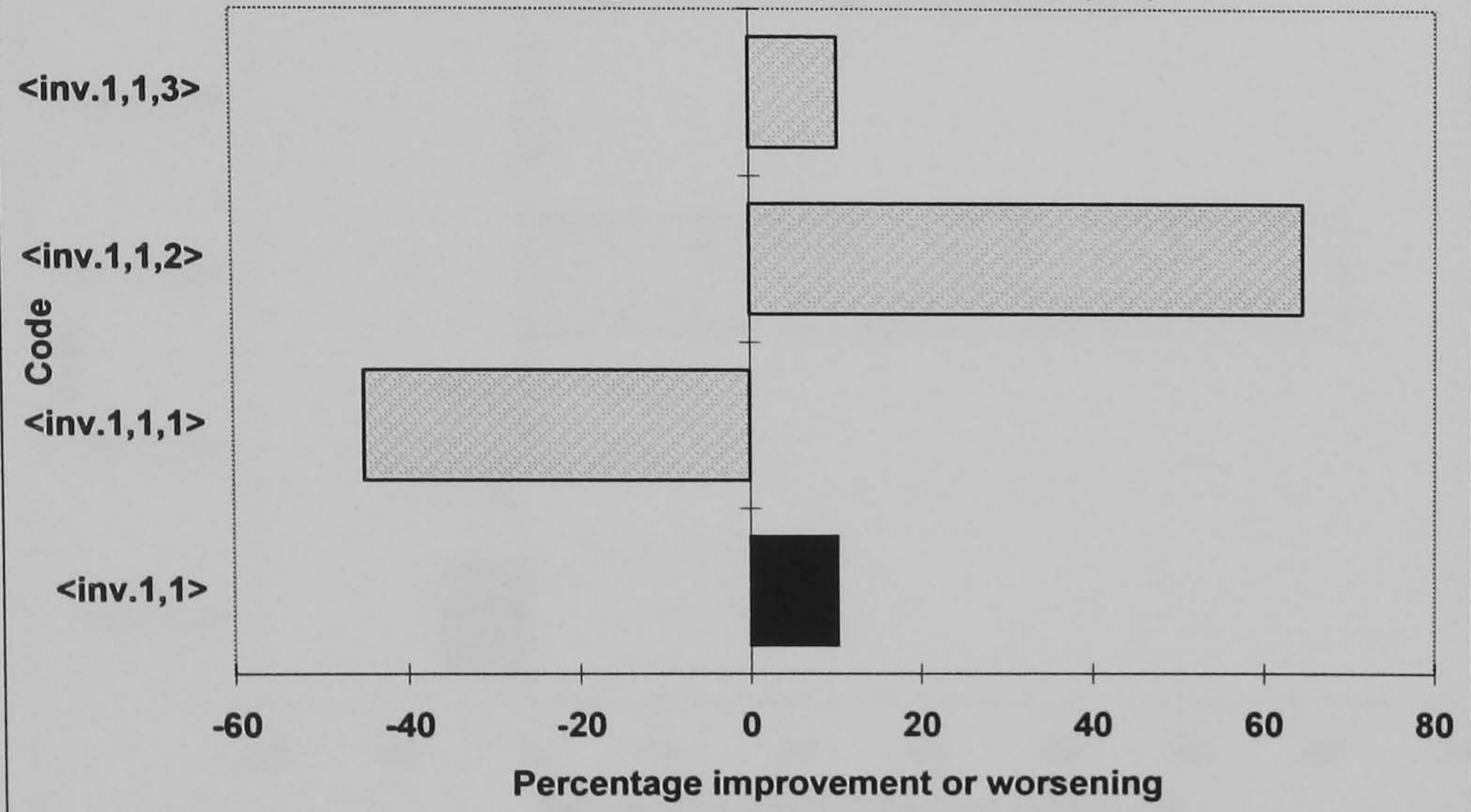
Graph 9.23: (r) Natural gas (in ground)



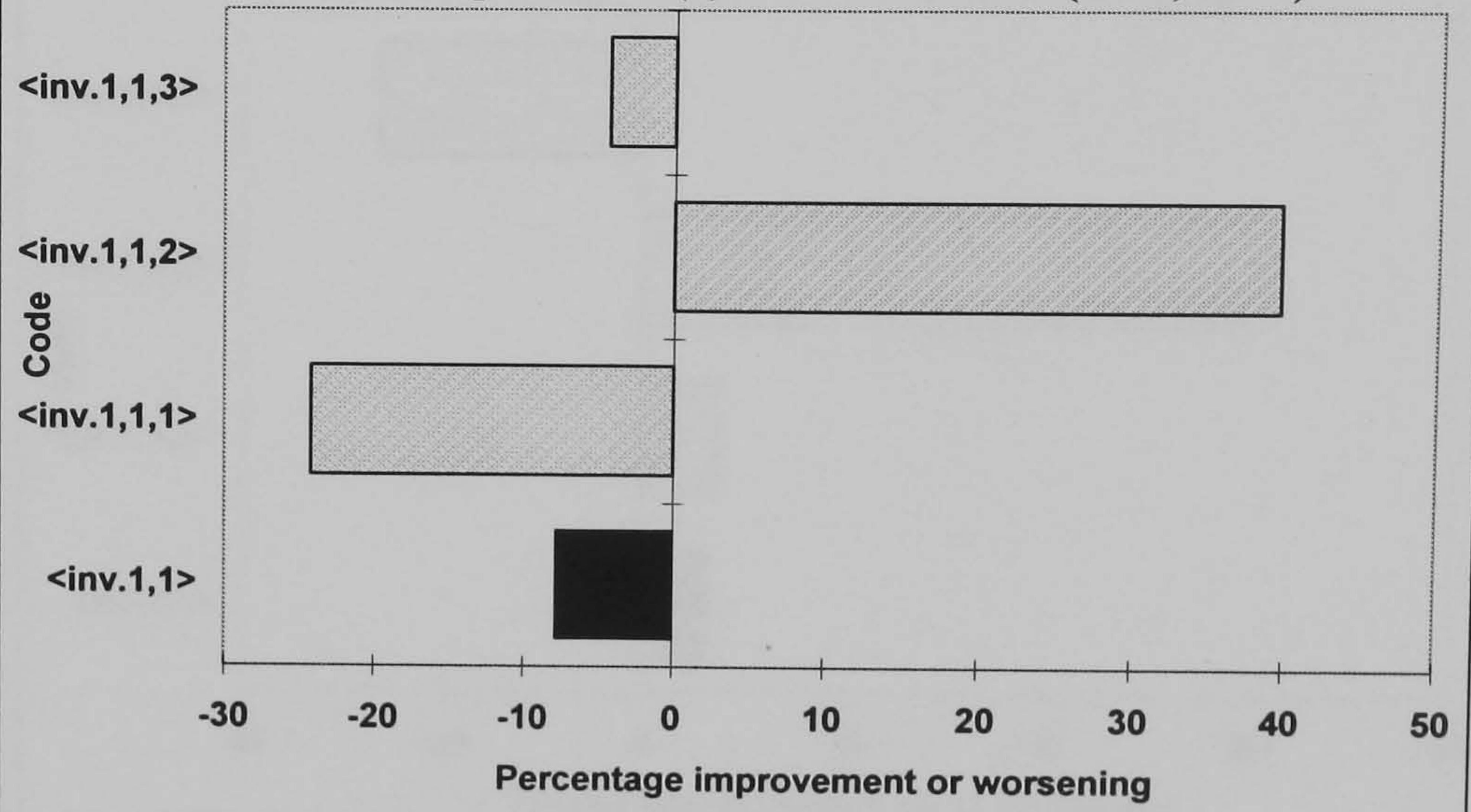
Graph 9.24: (r) Oil (in ground)



Graph 9.25: (a) Cadmium (Cd)



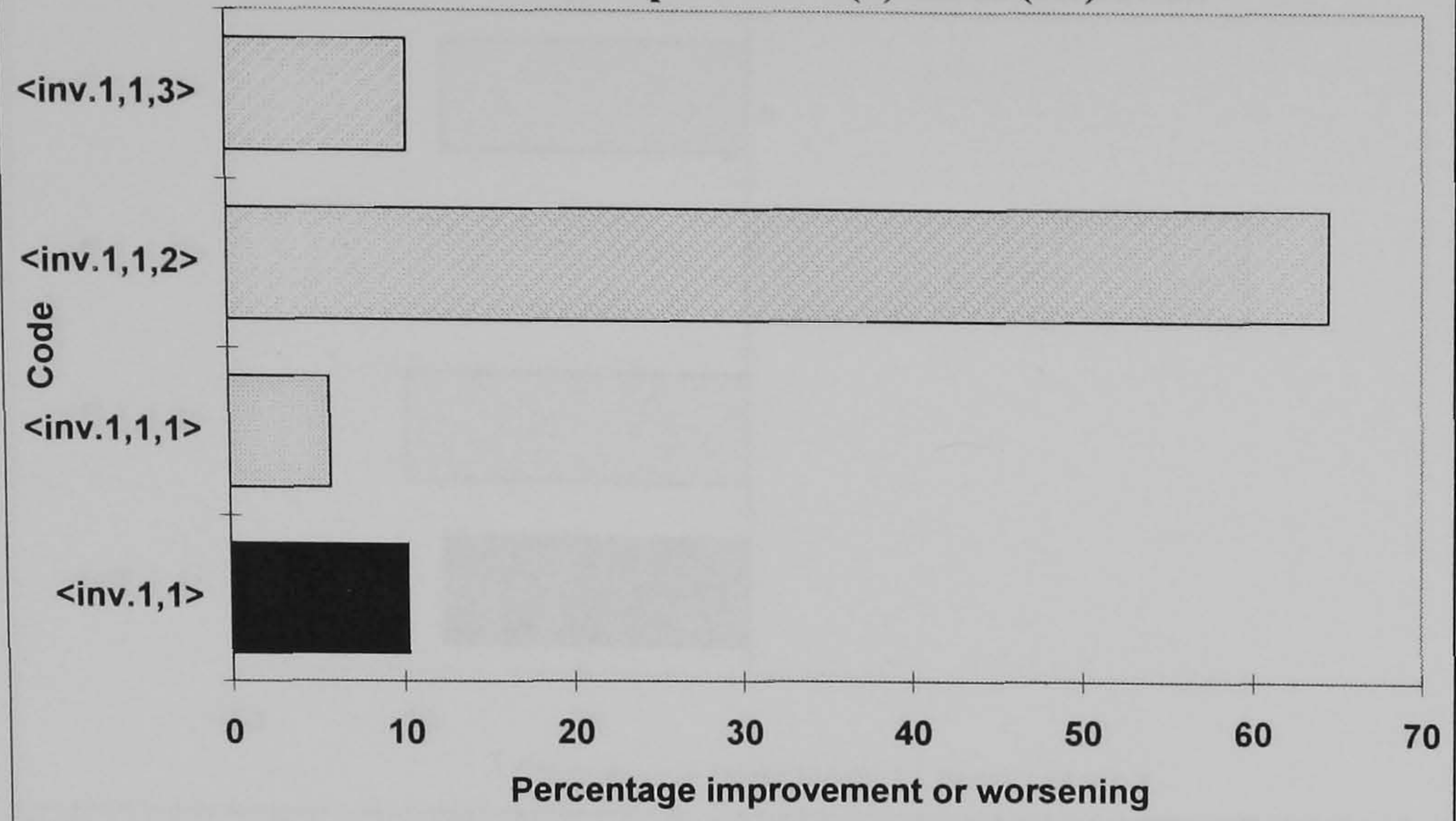
Graph 9.26: (a) Carbon Dioxide (CO2, fossil)



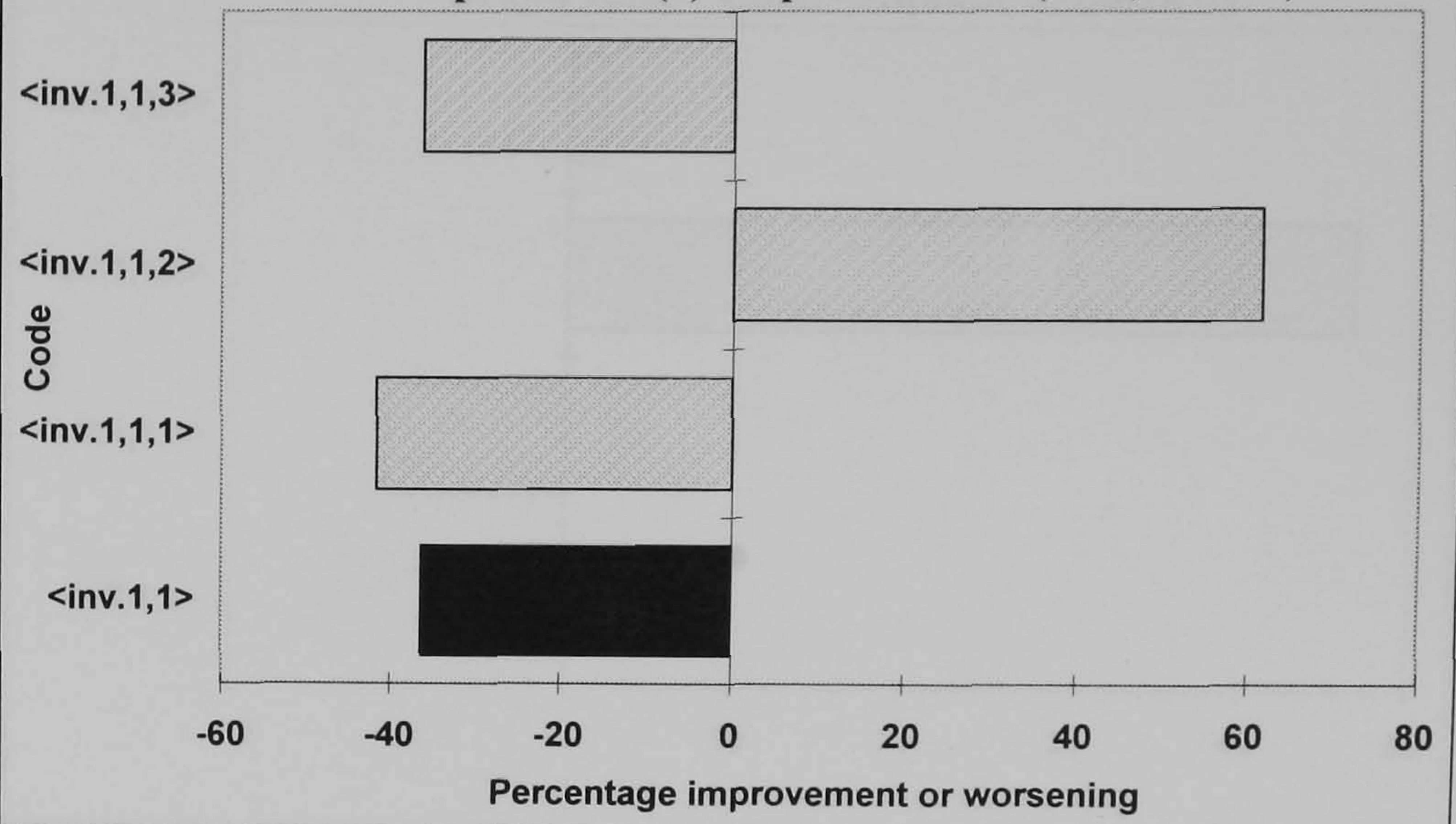
Code:	Operation being compared:	Level:
<inv.1,1>	BRM: Secondary Refined Lead Production Operations	level 2
<inv.1,1,1>	BRM (secondary): CX Plant Operations	level 3
<inv.1,1,2>	BRM (secondary): Overheads of Secondary Processing Operations	level 3
<inv.1,1,3>	BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations	level 3

Graphs comparing inventories for the BRM company year 1998-9 with 1997-8

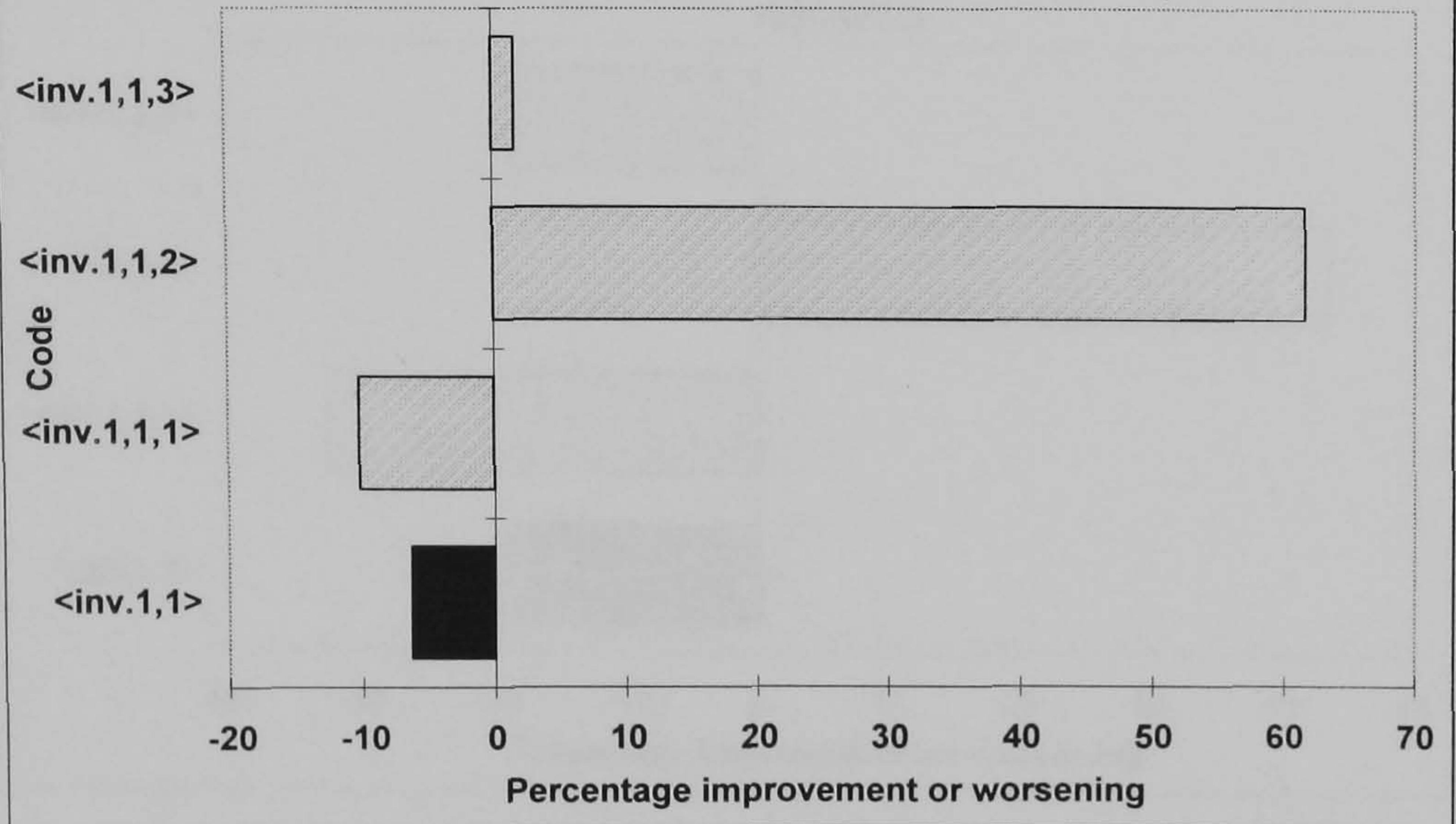
Graph 9.27: (a) Lead (Pb)



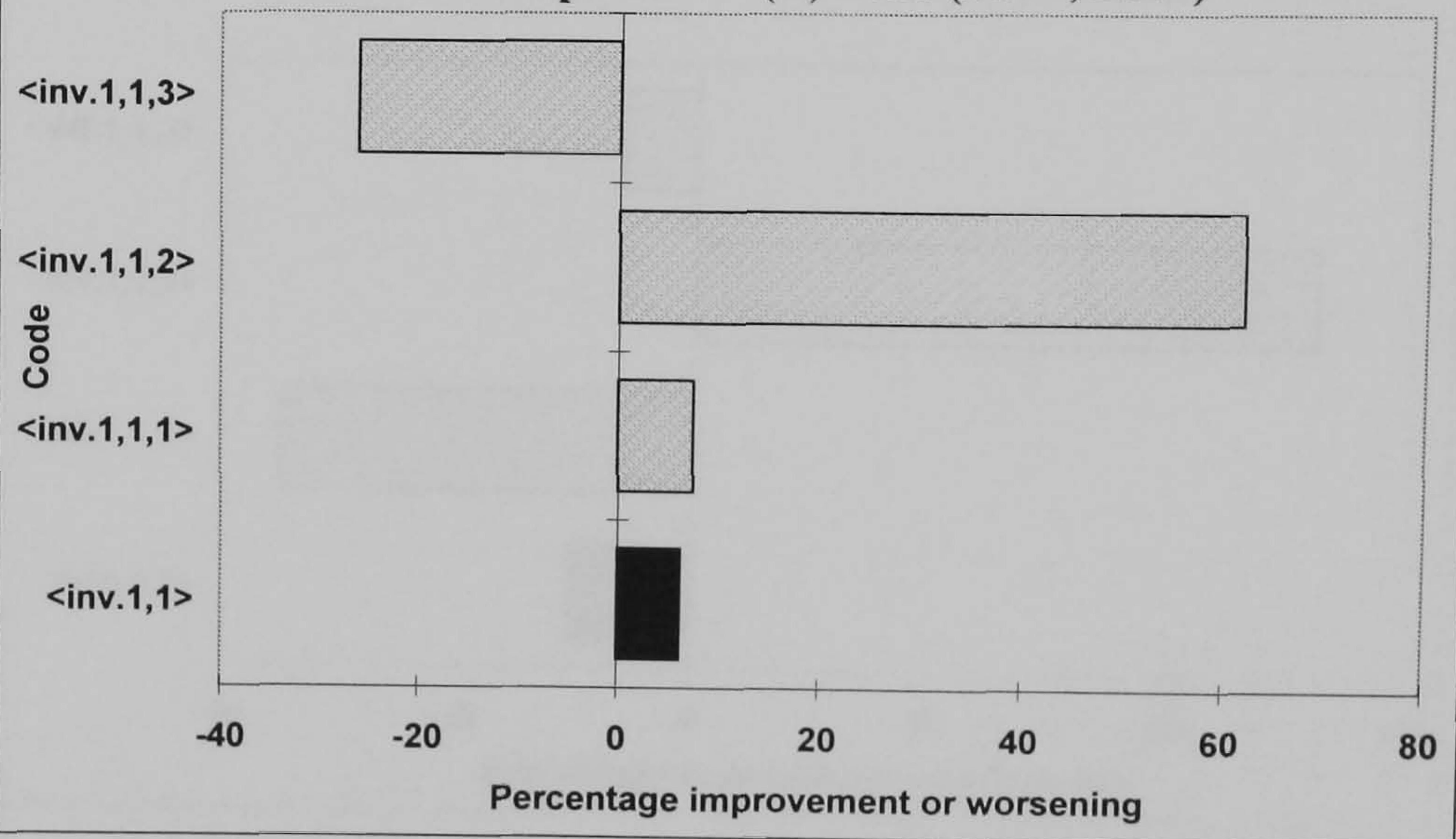
Graph 9.28: (a) Sulphur Oxides (SOx, as SO2)



Graph 9.29: (w) Cadmium (Cd++)



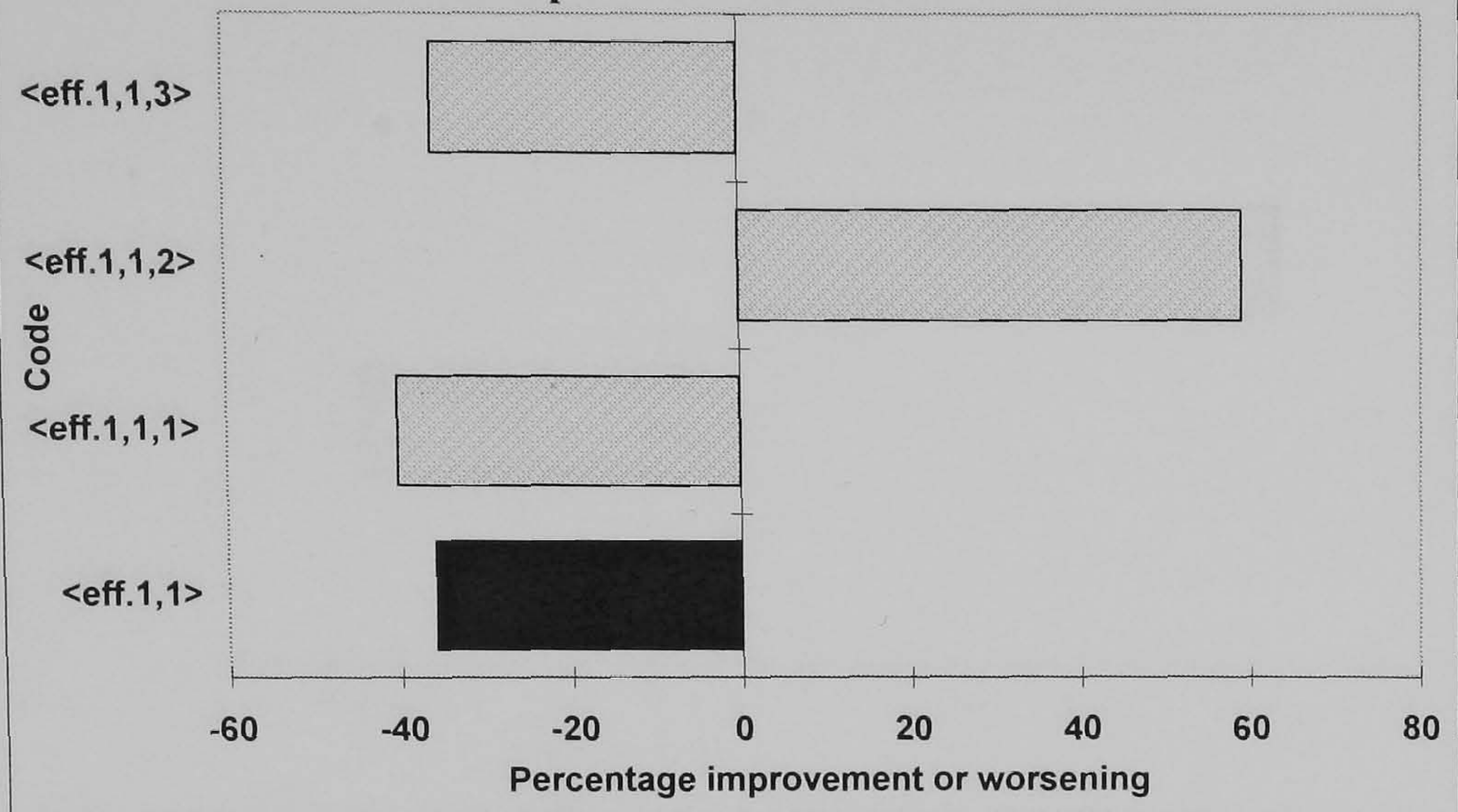
Graph 9.30: (w) Lead (Pb++, Pb4+)



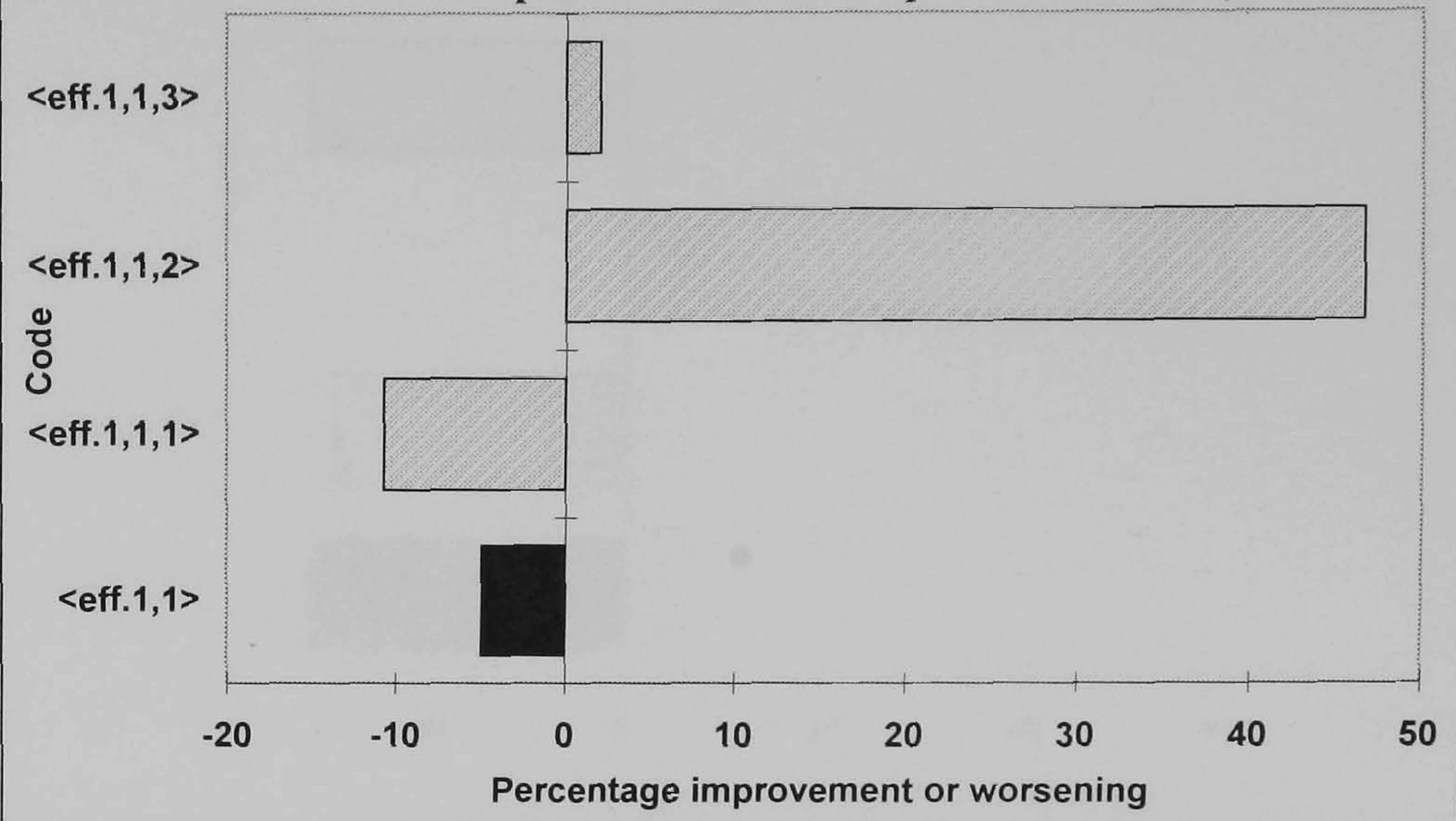
Code:	Operation being compared:	Level:
<inv.1,1>	BRM: Secondary Refined Lead Production Operations	level 2
<inv.1,1,1>	BRM (secondary): CX Plant Operations	level 3
<inv.1,1,2>	BRM (secondary): Overheads of Secondary Processing Operations	level 3
<inv.1,1,3>	BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations	level 3

Graphs comparing the BRM company year 1998-9 with 1997-8

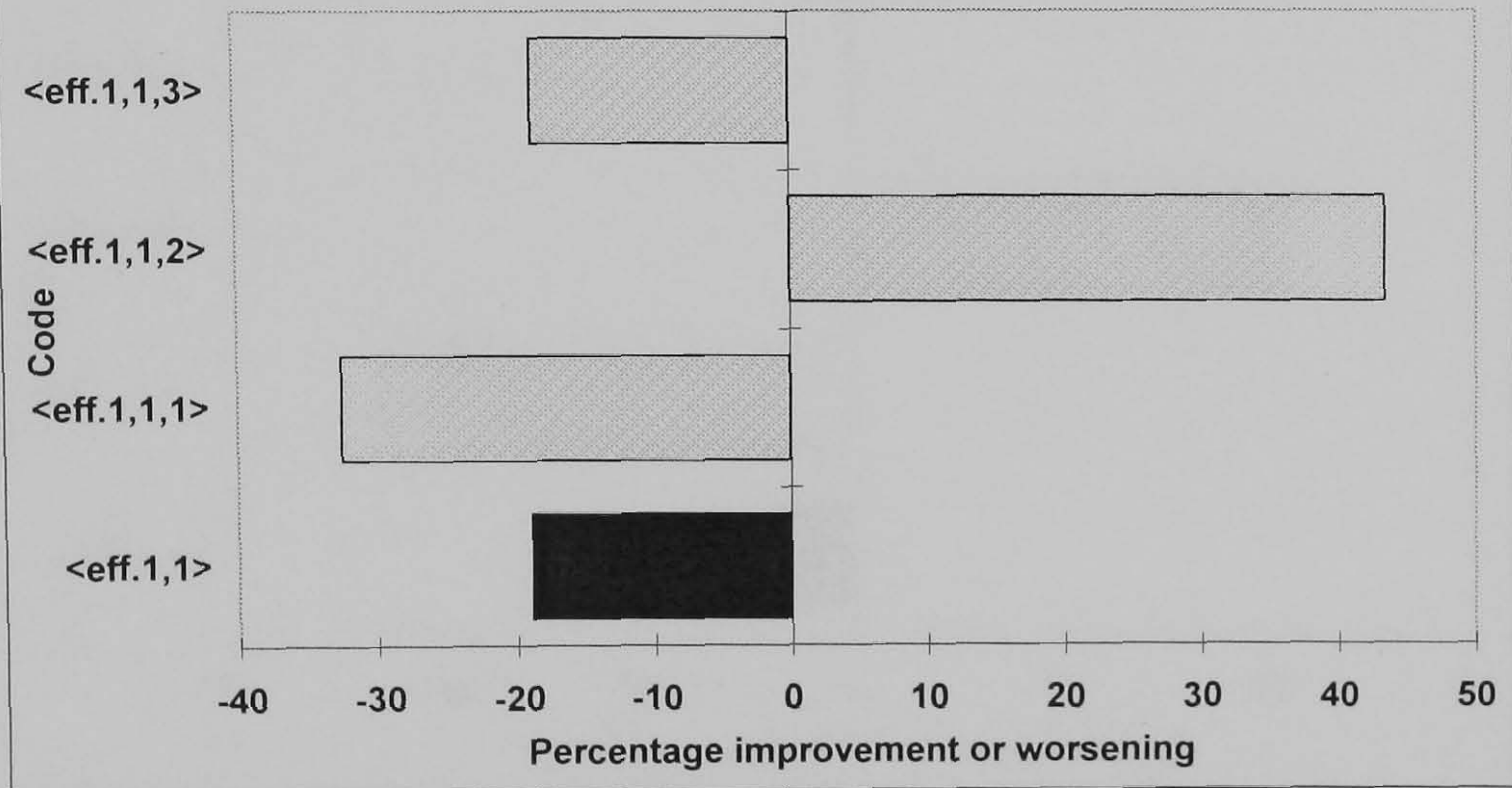
Graph 9.31: CML - Air Acidifcation



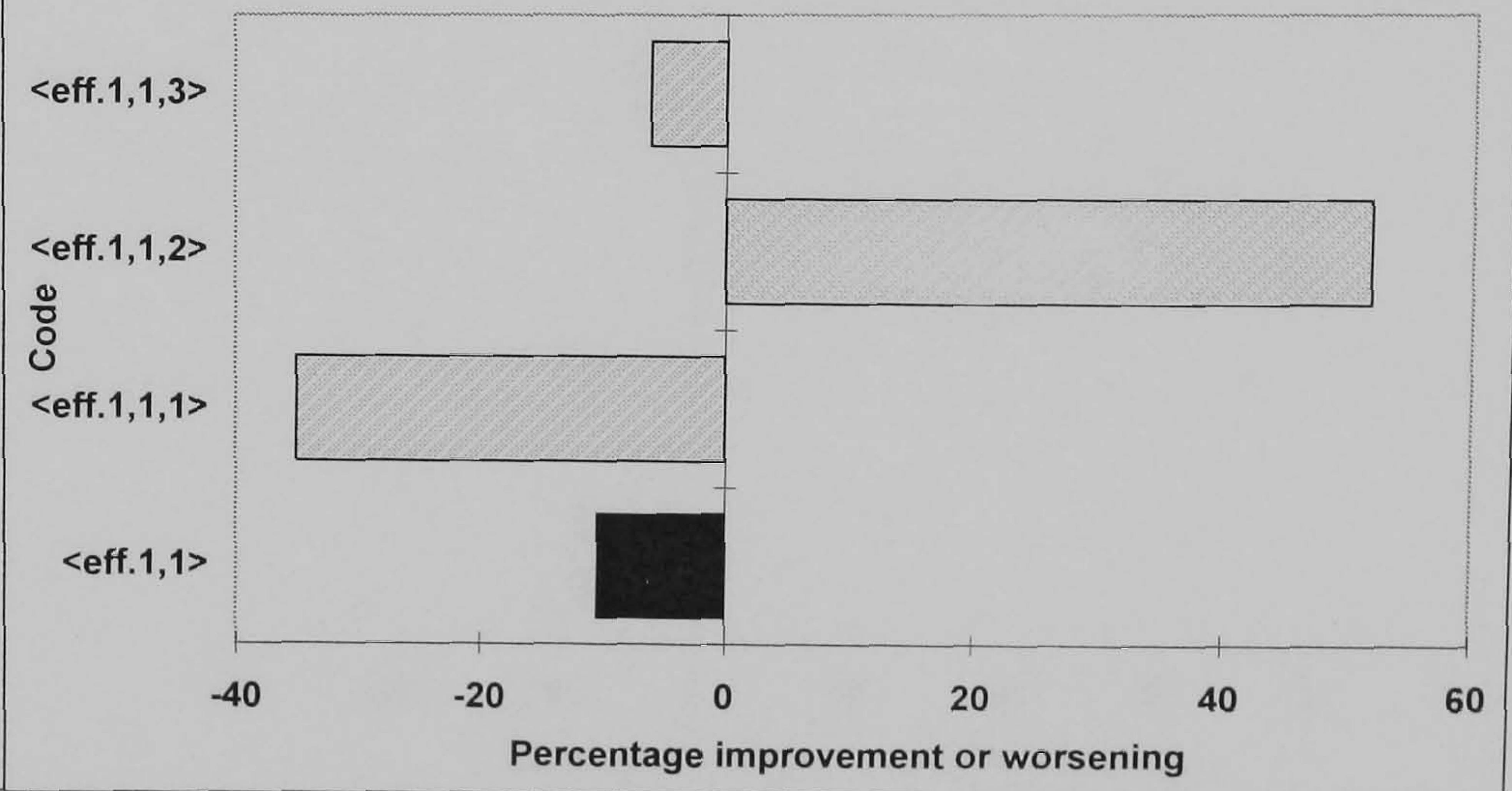
Graph 9.32: CML - Aquatic Ecotoxicity



Graph 9.33: CML - Depletion of non- renewable resources



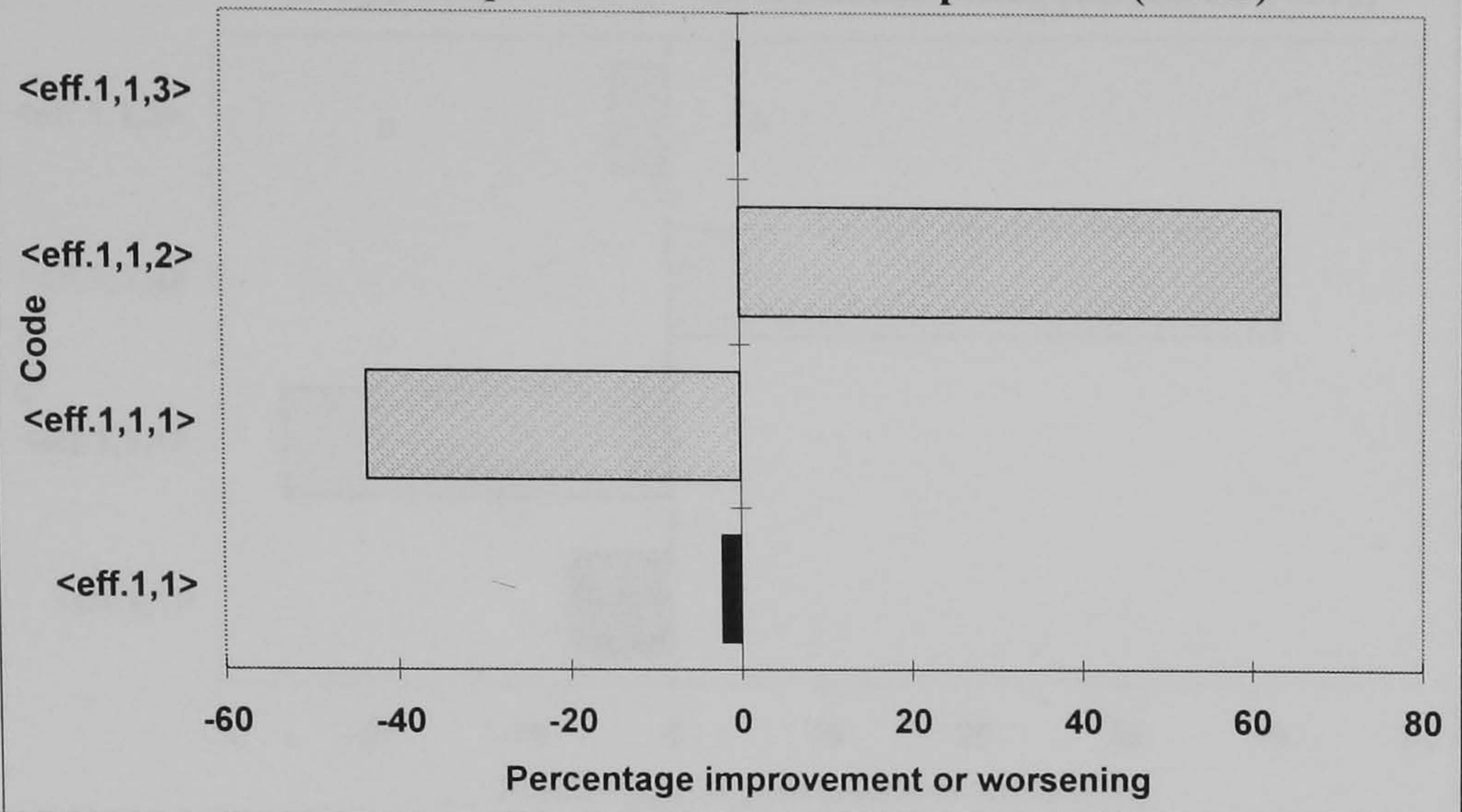
Graph 9.34: CML - Eutrophication



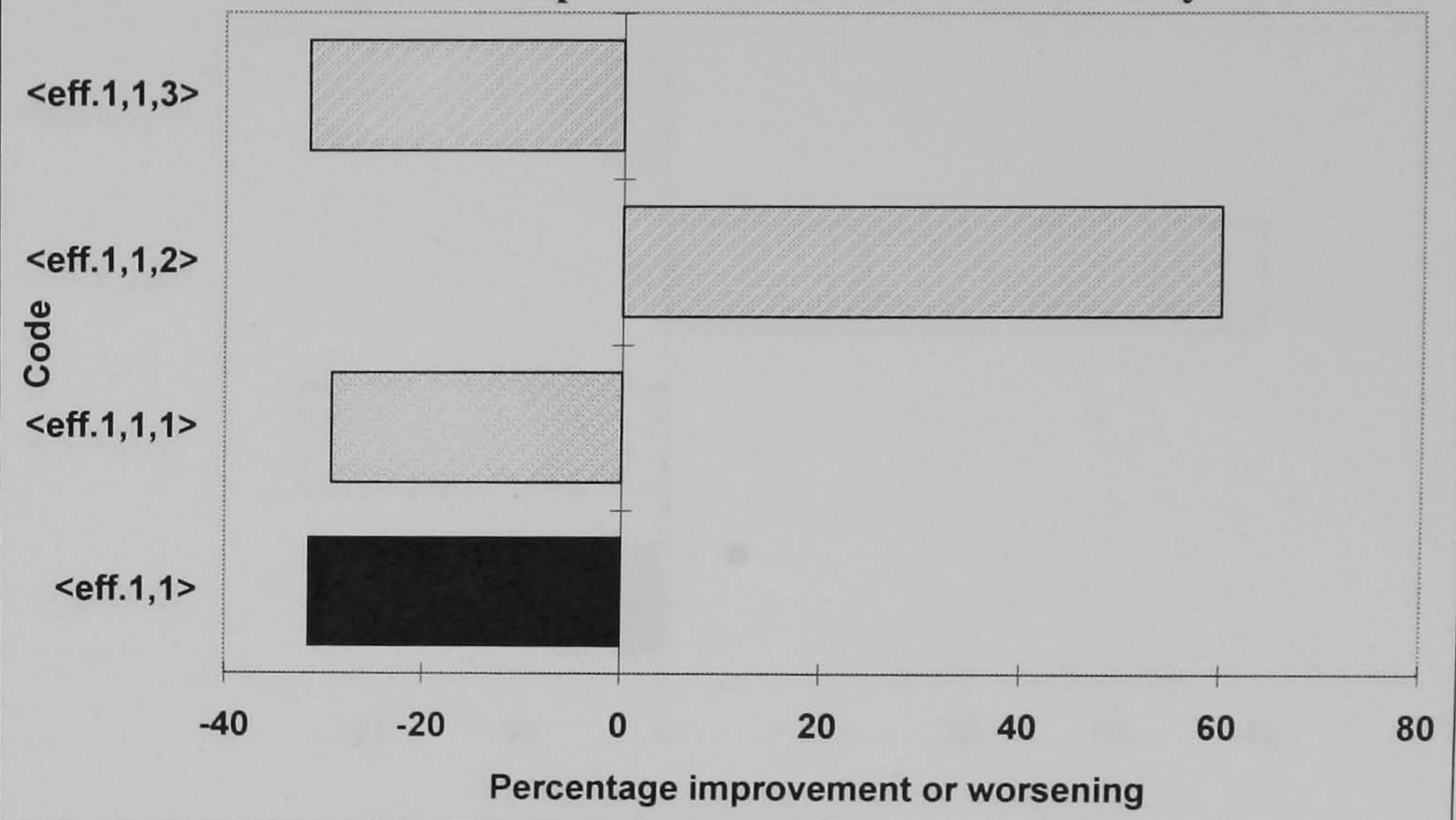
Code:	Operation being compared:	Level:
<eff.1,1>	BRM: Secondary Refined Lead Production Operations	level 2
<eff.1,1,1>	BRM (secondary): CX Plant Operations	level 3
<eff.1,1,2>	BRM (secondary): Overheads of Secondary Processing Operations	level 3
<eff.1,1,3>	BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations	level 3

Graphs comparing the BRM company year 1998-9 with 1997-8

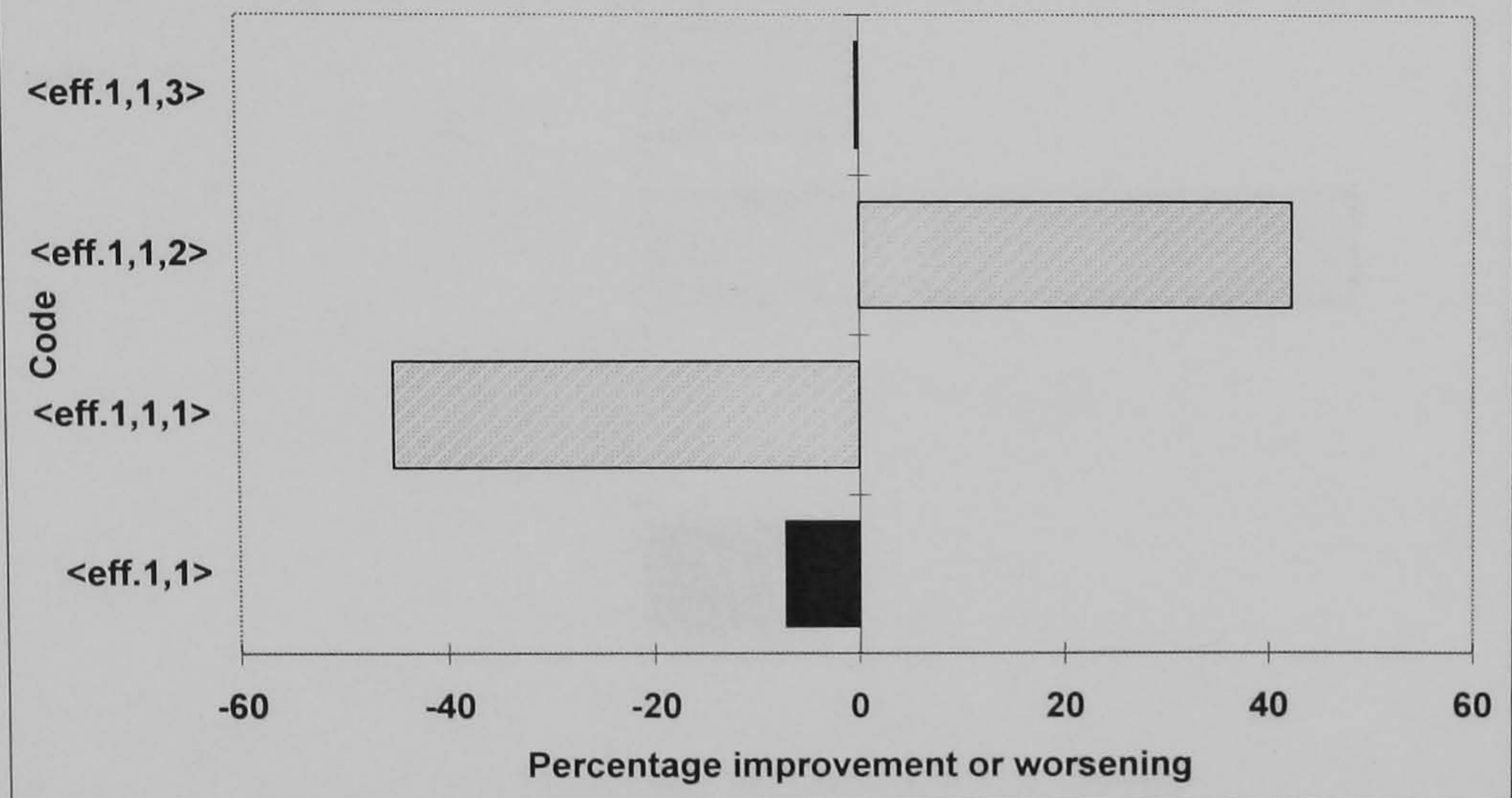
Graph 9.35: CML - Eutrophication (water)



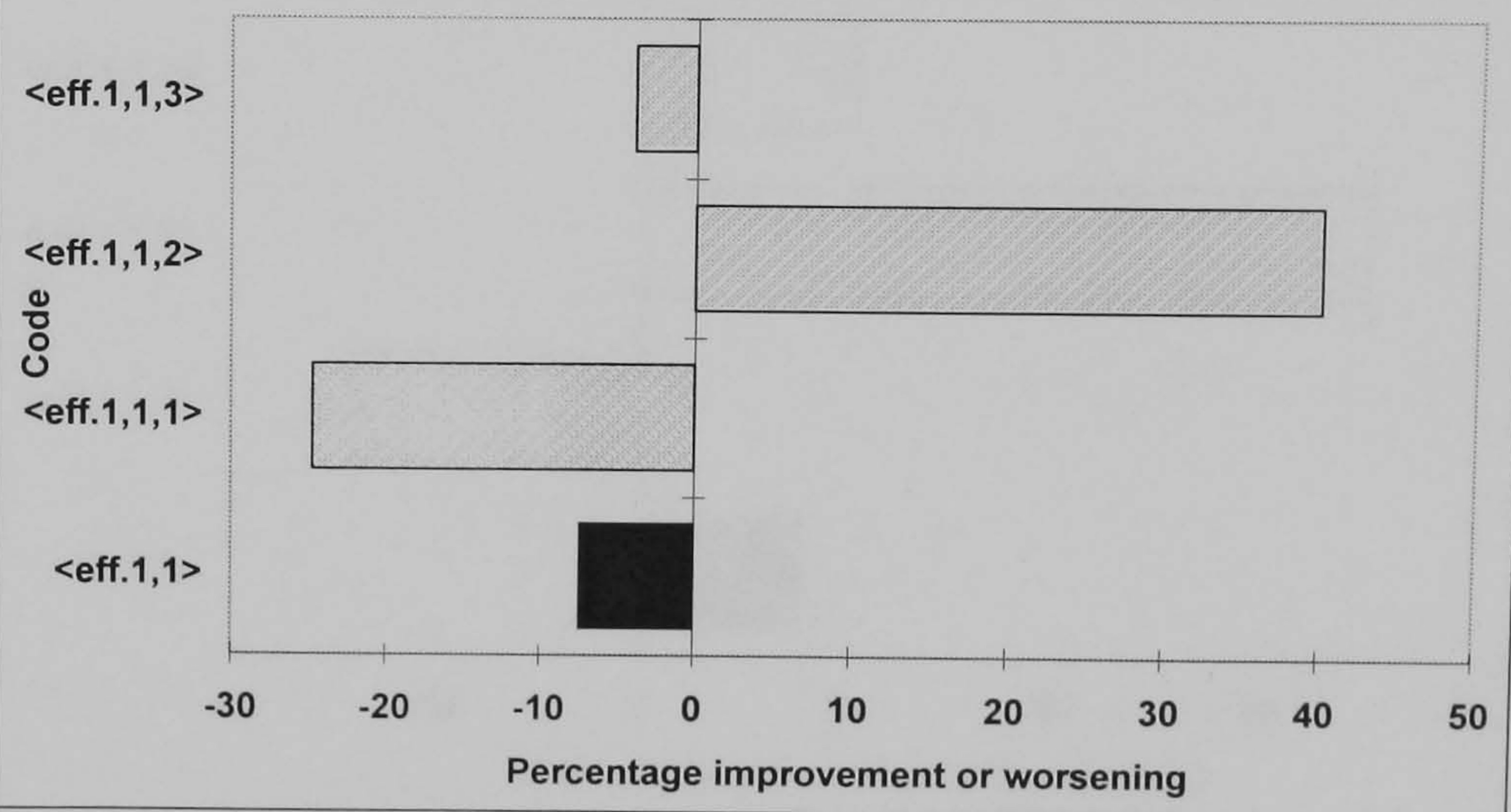
Graph 9.36: CML - Human toxicity



Graph 9.37: CML - Terrestrial ecotoxicity



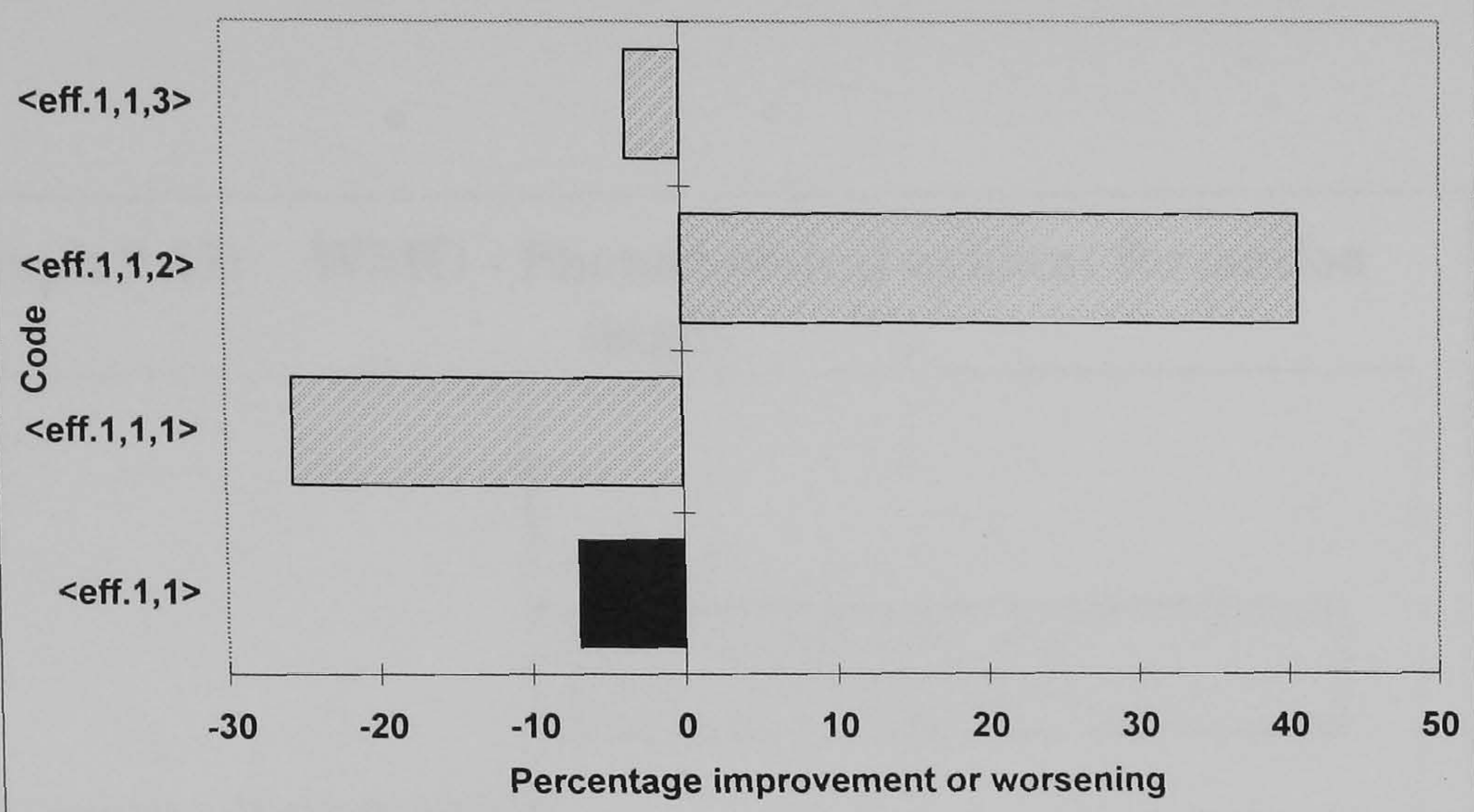
Graph 9.38: IPCC - Greenhouse effect (direct, 100 years)



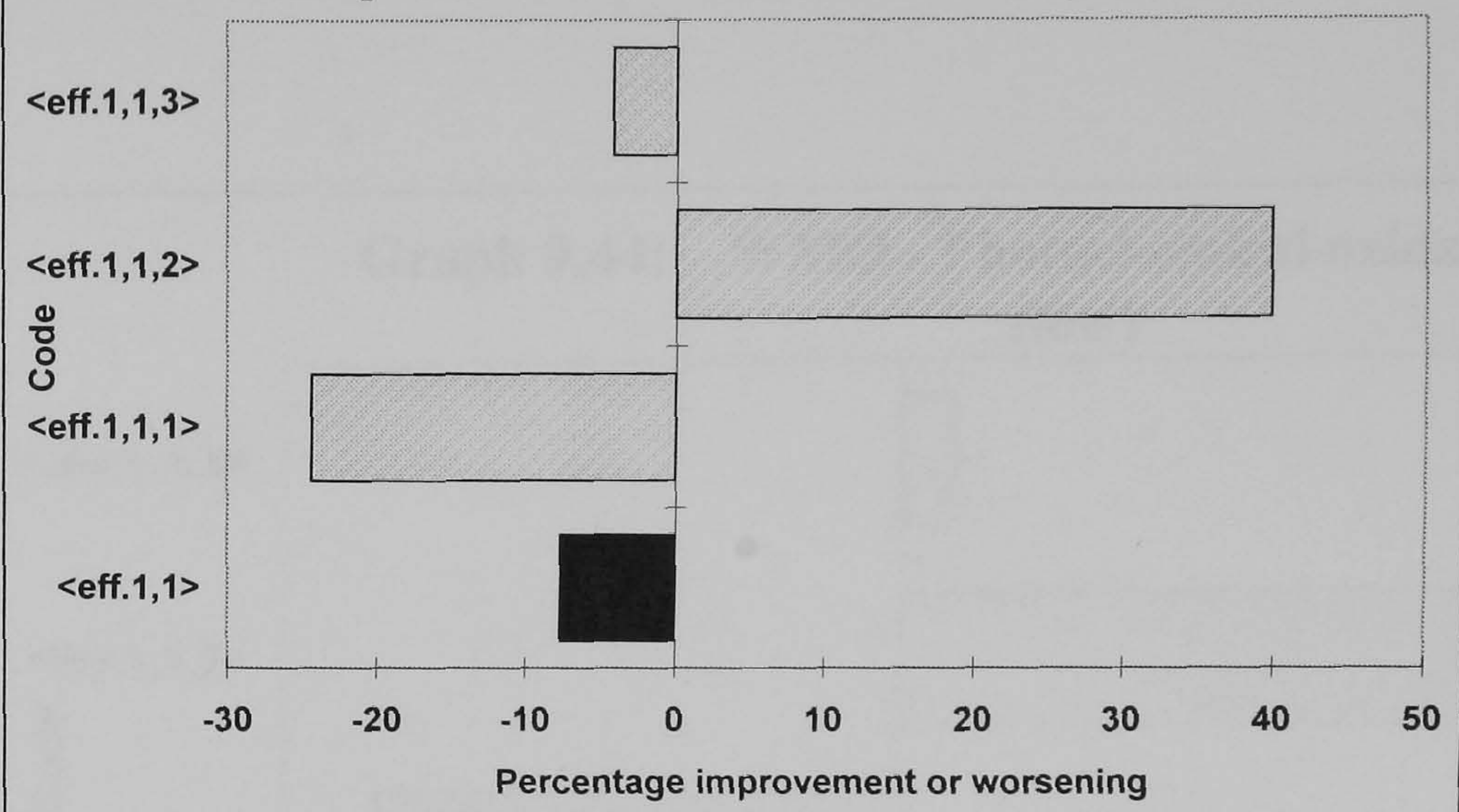
Code:	Operation being compared:	Level:
<eff.1,1>	BRM: Secondary Refined Lead Production Operations	level 2
<eff.1,1,1>	BRM (secondary): CX Plant Operations	level 3
<eff.1,1,2>	BRM (secondary): Overheads of Secondary Processing Operations	level 3
<eff.1,1,3>	BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations	level 3

Graphs comparing the BRM company year 1998-9 with 1997-8

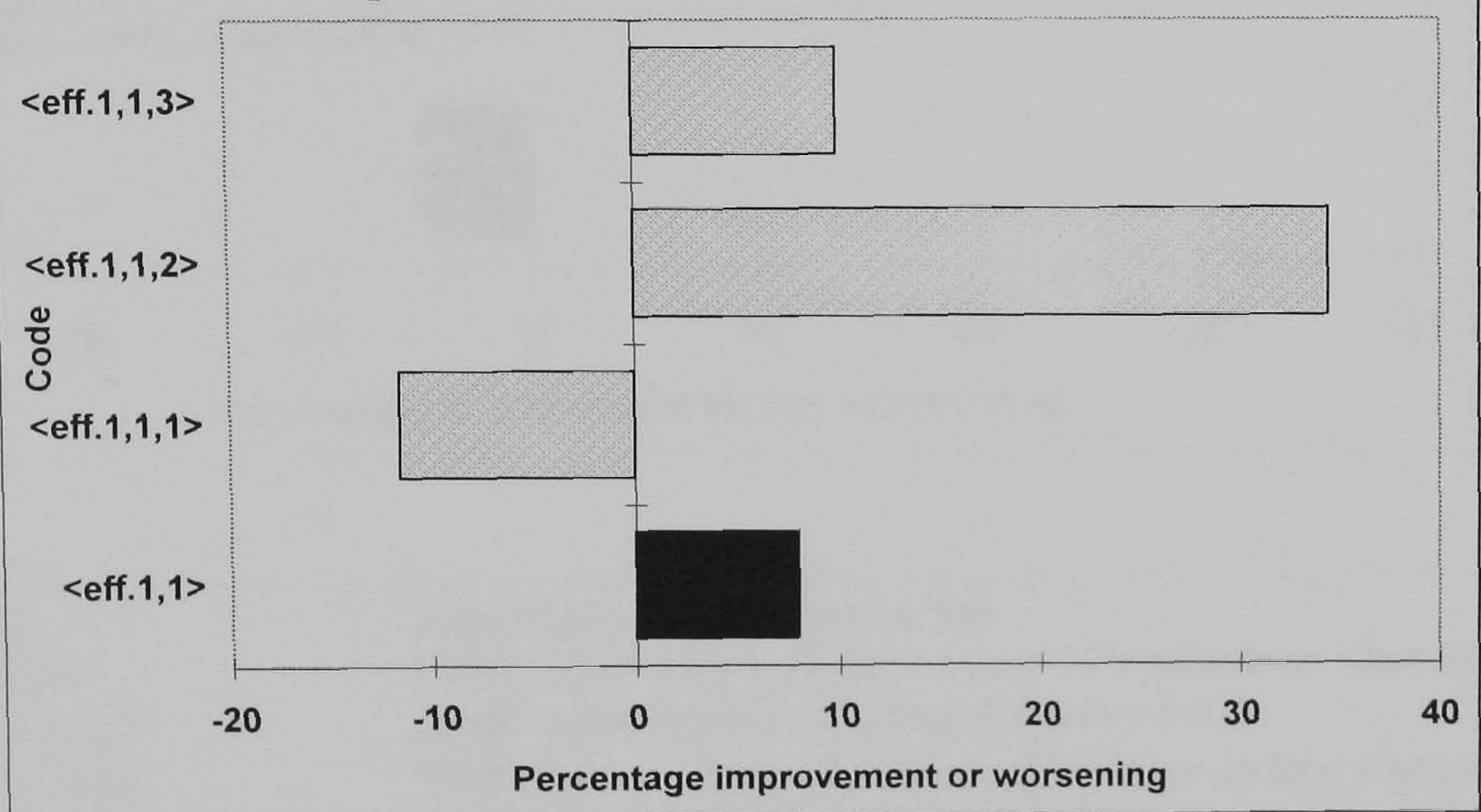
Graph 9.39: IPCC - Greenhouse effect (direct, 20 years)



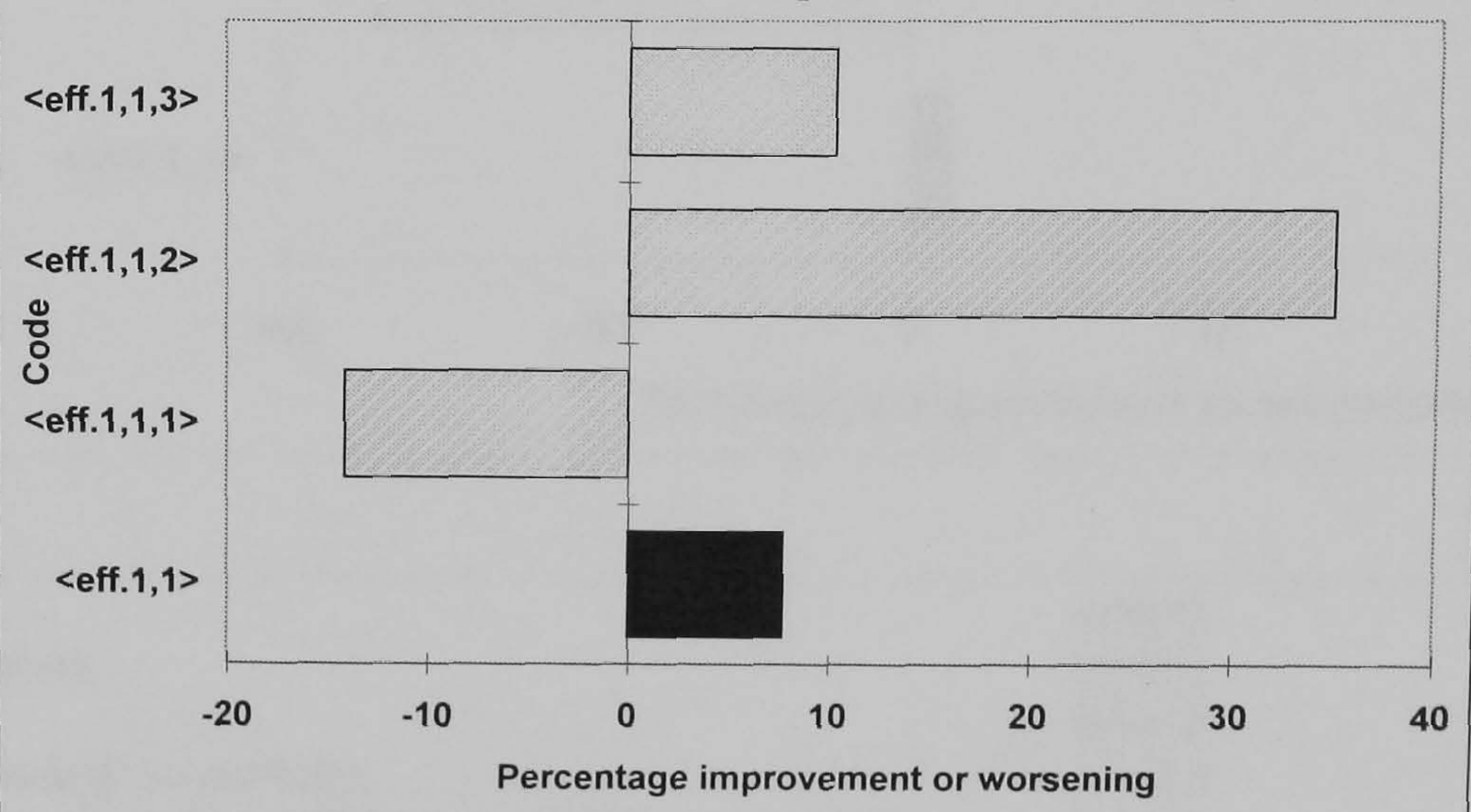
Graph 9.40: IPCC - Greenhouse effect (direct, 500 years)



Graph 9.41: WMO - Depletion of the ozone layer (high)

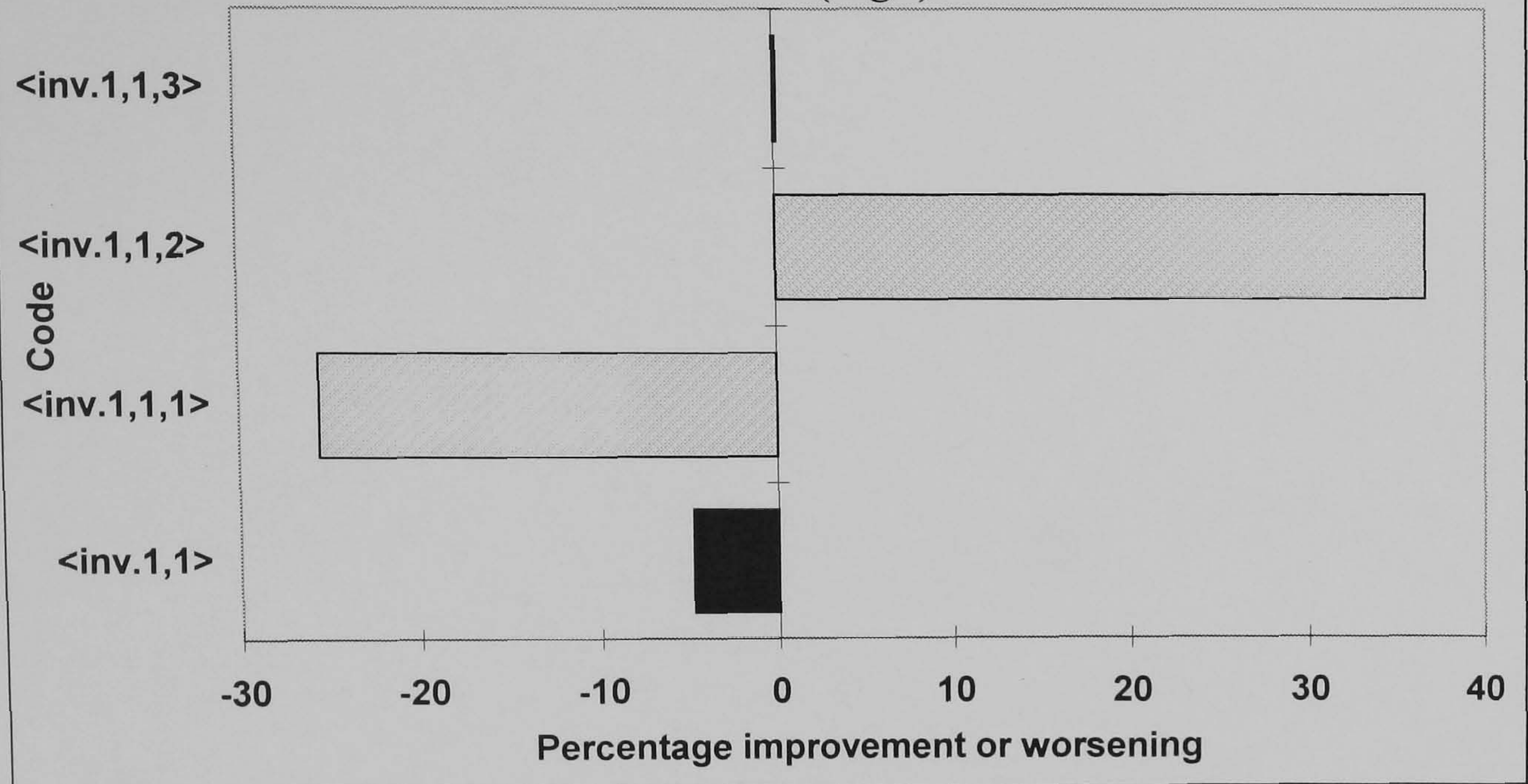


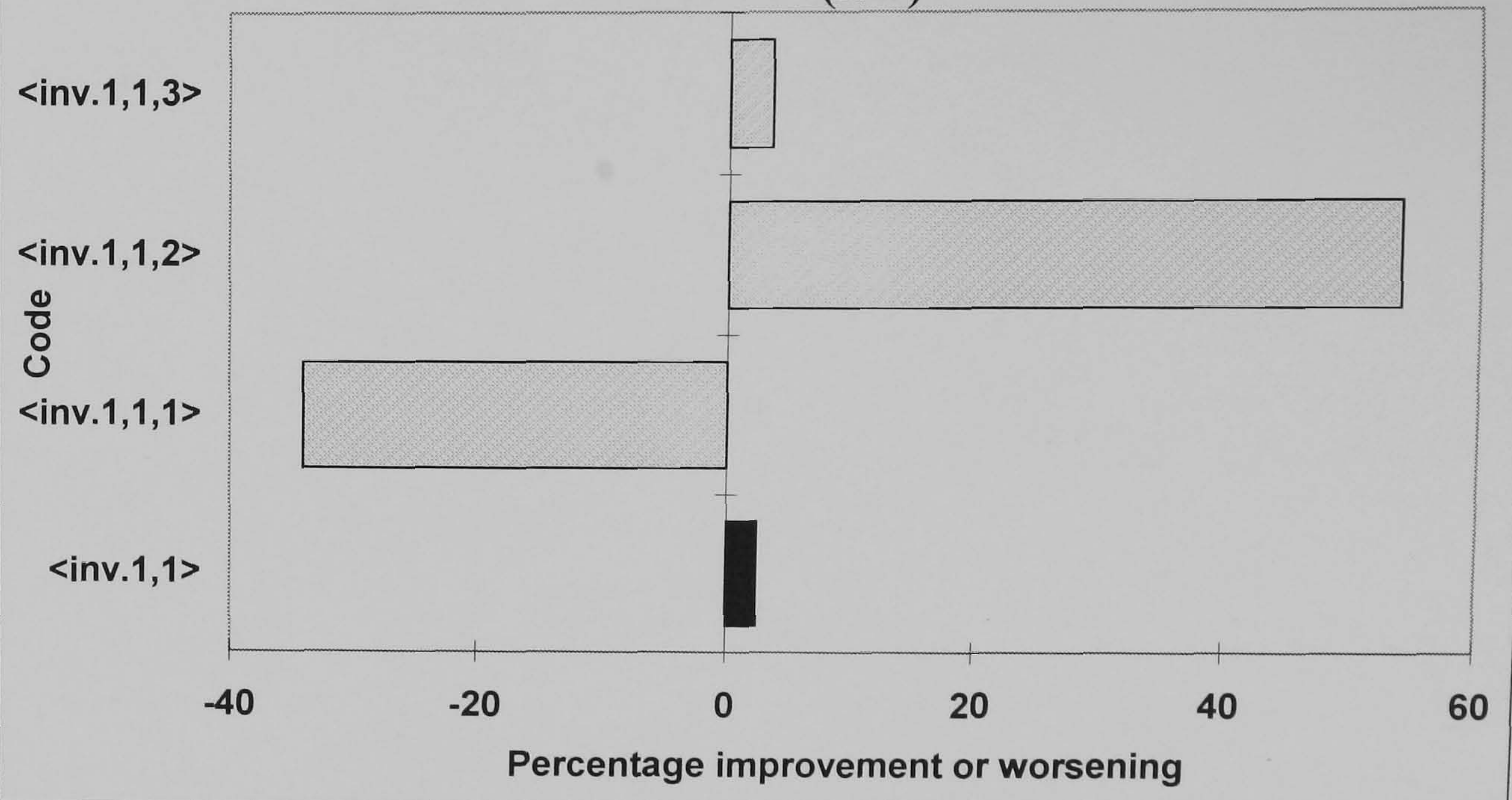
Graph 9.42: WMO - Depletion of the ozone layer (low)



Code:	Operation being compared:	Level:
<eff.1,1>	BRM: Secondary Refined Lead Production Operations	level 2
<eff.1,1,1>	BRM (secondary): CX Plant Operations	level 3
<eff.1,1,2>	BRM (secondary): Overheads of Secondary Processing Operations	level 3
<eff.1,1,3>	BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations	level 3

Graphs comparing the BRM company year 1998-9 with 1997-8


Graph 9.43: WMO - Photochemical oxidant formation (high)


Graph 9.44: WMO - Photochemical oxidant formation (low)

Code:	Operation being compared:	Level:
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<eff.1,1,1>	BRM (secondary): CX Plant Operations	level 3
<eff.1,1,2>	BRM (secondary): Overheads of Secondary Processing Operations	level 3
<eff.1,1,3>	BRM (secondary): Smelting, Rotary Furnace, Refining and Moulding Operations	level 3