

Life Cycle Assessment for Environmental Improvement of Minerals' Production

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ABSTRACT

Life Cycle Assessment (LCA) is a relatively new, cutting edge environmental decision support tool recently standardised by the International Standardisation Organisation (ISO). The LCA provides quantitative environmental and energy data on products and processes. This paper provides an overview of the current status of LCA methodology, and reviews LCA case studies from the primary resources industries against this background. Moreover, potential applications for cradle-to-gate LCAs in the minerals sector are briefly explored. The paper concludes that LCA has significant potential for supporting environmental improvement and innovation in the minerals sector. To increase public trust in LCAs for the minerals sector, it is however imperative that current best available LCA practice is being applied.

INTRODUCTION

Life Cycle Assessment is a technique for assessing the environmental performance of a product, process or activity from '*cradle to grave*', i.e. from extraction of raw materials to final disposal (e.g. Curran, 1997; UNEP, 1999). Today's LCA originates from '*net energy analysis*' studies, which were first published in the 1970s and considered only energy consumption over the life cycle of a product or process. Some later studies, in particular for packaging systems and selected other consumer products (e.g. diapers) included wastes and emissions (also called '*Resource and Environmental Profile Analysis*'), but none of them went further than just quantifying materials and energy use. At this point it was clear that a more sophisticated approach to environmental issues was needed. As a result, in 1990, the Society for Environmental Toxicology and Chemistry (SETAC) initiated activities to define LCA and develop a '*technical framework*' (Fava et al, 1991) and a '*code of practice*' (Consoli, et al, 1993) for conducting LCA studies. Meanwhile the first comprehensive LCA methodologies were produced, both in Europe (most importantly Heijungs, 1992) and in the USA (USEPA, 1991). Soon afterwards, the International Standardisation Organisation (ISO) started similar work on developing principles and guidelines on the LCA methodology, resulting in the ISO 14040 series of standards (ISO 1997 a; b; 1999 a; b). Although SETAC and ISO worked independently of each other, a general consensus on the LCA framework between the two bodies started to emerge. The ISO standards focus on the procedures to be followed for conducting LCA with a view to assure transparency, independence and accountability of the LCA process. SETAC focuses through its

various working groups on best practicable methodologies for conducting the different parts of LCA with a view to achieve use of best scientific insights in conducting LCA.

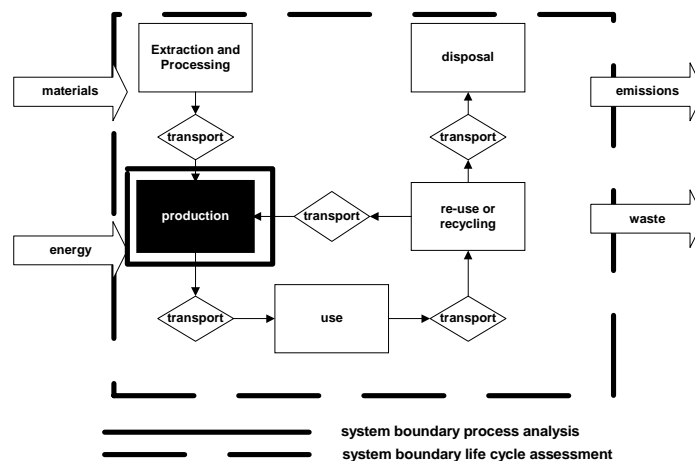
Although the use of LCA has traditionally been oriented towards improving the environmental performance of products, LCA can also be applied to processes for the selection of environmental best practices and technologies (Azapagic, 1999), and for optimisation of the environmental performance of existing processes (Azapagic et al, 1999). In its application to processes, the LCA system boundary normally excludes product use, recycling and waste disposal. Such LCA provides the manufacturer with insights in significant environmental performance issues that may then be addressed through improvement and innovation of processes, for instance through the application of Cleaner Production and Eco-Efficiency tools (e.g. Van Berkel, 2000). Moreover, such LCAs generate transparent baseline information as input for LCA studies by the different users of the manufacturer's products. This type of LCA is also referred to as 'cradle-to-gate' assessment, and has over the last few years been applied to a number of primary resources, in Australia and abroad.

This paper explores how LCA can be applied for environmental improvement in the minerals sector. First, the LCA methodology is discussed from a primary resources extraction and processing perspective, and examples are given to illustrate some of the bottlenecks encountered so far. Next, a few applications for LCA in the minerals sector are reviewed, and it is argued that applications for process selection and optimisation are most beneficial. The concluding section addresses the barriers towards the widespread use of LCA, and the potential benefits from LCA.

METHODOLOGY IN MINERALS CONTEXT

Life Cycle Assessment can be thought of as a form of environmental systems analysis. In terms of the system boundary, LCA is an extension to the conventional system analysis, in which the system boundary is drawn around the process of interest only (Stewart et al, 2000). Figure 1 illustrates the way in which LCA can complement conventional process analysis. While chemical or process engineering is normally

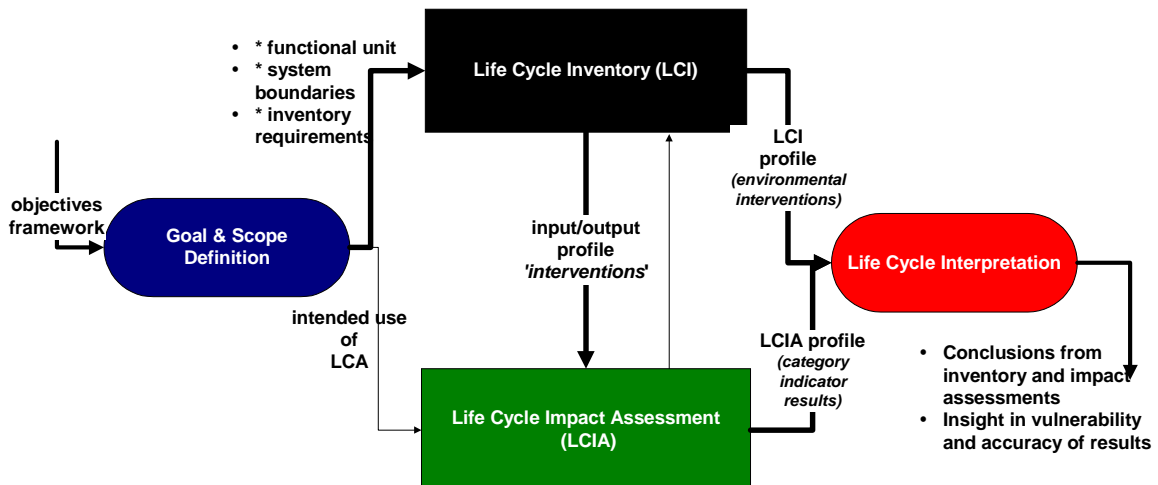
Figure 1: System boundaries for process and life cycle assessments (modified from Azapagic et al, 1999 a).



concerned with the operations within the process system boundary, LCA considers the whole material and energy supply chains, so that the system of concern becomes everything within the lifecycle system boundary. An inventory is made of the material and energy flows that enter, exist in or leave the system. It includes material and energy resources and emissions to air, water and land. LCA is then based on rigorous mass and energy balances calculated by modelling and/or measuring the material and energy flows of the various processes in the system. These balances are used to evaluate the resource consumption and waste generation inventories of the product or process. Next these inventories are linked to recognised environmental impacts in an objective manner.

The LCA methodology is still under development. At present, the methodological framework comprises four stages (ISO, 1997 a): goal and scope definition; life cycle inventory assessment; life cycle impact assessment; and life cycle interpretation (see figure 2). The *Goal and Scope Definition* establishes the functional unit, system boundaries, and quality criteria for inventory data. The *Life Cycle Inventory Analysis* deals with the collection and synthesis of information on physical material and energy in- and outputs in the various stages of the product life cycle. The input and output data collected for individual process steps are then summarised by input and output category, to compile the environmental input output profile, or Life Cycle Inventory LCI- profile for the product life cycle being studied. In the *Life Cycle Impact Assessment* these environmental input and outputs are assigned to environmental impact categories, and characterisation models used to calculate the contribution of each of these inputs and outputs to category indicators. This leads to a Life Cycle Impact Assessment Profile of category indicator scores for all environmental impact categories. Finally, the *Life Cycle Interpretation* deals with the interpretation of the results from both the Life Cycle Inventory Analysis and Life Cycle Impact Assessment. It includes the identification of significant issues and the evaluation of results.

Figure 2: Structure of Life Cycle Assessment (modified from ISO 1997 a).



Goal and Scope Definition

The goal and scope of a LCA study are to be clearly defined (ISO, 1997a). The goal should unambiguously state the intended application, the reasons for carrying out the study and the intended audience. The use of LCA for the comparison of life cycle environmental impacts from different product systems is often perceived as the ultimate,

but also most widely debated, goal for conducting LCAs. The ISO 14040 series defines a number of features that LCAs have to meet to be used for such '*comparative assertions*'. These include the need to include the Life Cycle Impact Assessment with a sufficiently comprehensive set of category indicators, inclusion of a critical review, and elimination of weighting and other normative steps.

Scoping the LCA study is an iterative process that results in the definition of the functional unit, the establishment of system boundaries and data quality requirements. The functional unit is a measure of the performance of the functional outputs of the product system. The functional unit has then to be related back physical units to perform the LCA. Once the functional unit and goals have been set, the system boundaries and data quality requirements can be deducted.

Strictly speaking for a minerals system the preferred functional unit would follow a particular application of the mineral. In the case of copper, for instance, its use in copper wire, with the functional unit defined in terms of a number of years that a particular electric current can be transferred at a specified performance level (e.g. related to power losses). This interpretation is not practical due to the vast diversity of the possible applications of most minerals. In a minerals environment it has therefore become common practice to conduct '*cradle-to-gate*' LCA that includes resource extraction and minerals beneficiation as the primary production processes. The functional unit can then be formulated in physical units (tons) of metal at a particular quality level available from the factory gate of the beneficiation plant.

The importance of a proper functional unit and its implications are regularly overlooked. For example, the functional unit for the recent LCA study for gas on behalf of the Australian Gas Association was defined as 'kilograms carbondioxide equivalent per gigajoule of gross energy contained in the fuel as delivered to the end user (kg CO₂-e/GJ)' (AGA, 2000). Firstly, the functional unit should not have been linked to any – set of – specific environmental impact categories of concern. Therefore, the functional unit should probably have been 'gigajoule of gross energy contained in the fuel as delivered to the end user', with 'kilogram carbondioxide-equivalent' the '(lifecycle impact) category indicator' for the climate change 'impact category'. Secondly, this functional unit defines delivery of the fuel as the key function of the product, and therefore in principle does not include fuel combustion and subsequent release of the energy contained in the fuel. In these terms, the functional unit appears to be more appropriate for an LCA of the distribution network, in which case the functional unit would have to be complemented with performance criteria, for instance in terms of fuel flows, accepted fuel losses and service lifetime. In fact the LCA did include the combustion emissions, and a functional unit in terms of releasing energy for power generation and residential, industrial and commercial heating would have been more appropriate.

Life Cycle Inventory Analysis

The Life Cycle Inventory Analysis (LCI) is the LCA phase involving the compilation and quantification of inputs and outputs for a given product system throughout its life cycle. It is sometimes also referred to as the 'resource and waste inventory'. The definition of goal and scope of the LCA provides the initial plan for the LCA, and the LCI is concerned with the data collection and calculation procedures.

LCI relies on being able to link unit processes within a product system by simple material or energy flows. However, most industrial processes yield more than one product, and intermediates or by-products are recycled. Therefore the materials and energy flows as well as associated environmental releases are to be allocated to the different products according to clearly stated allocation procedures (ISO 1997, b). The preferred practice is to minimise such allocation by expanding the system boundary (and include the entire product system for the recycled material or co-product in the LCA) or further subdivision of multi-output processes to arrive at single-output processes. Where allocation cannot be avoided, it should be done in a way that reflects the physical relationships between the process outputs, or if such is not feasible, in a way that reflects other (e.g. monetary) relationships between them.

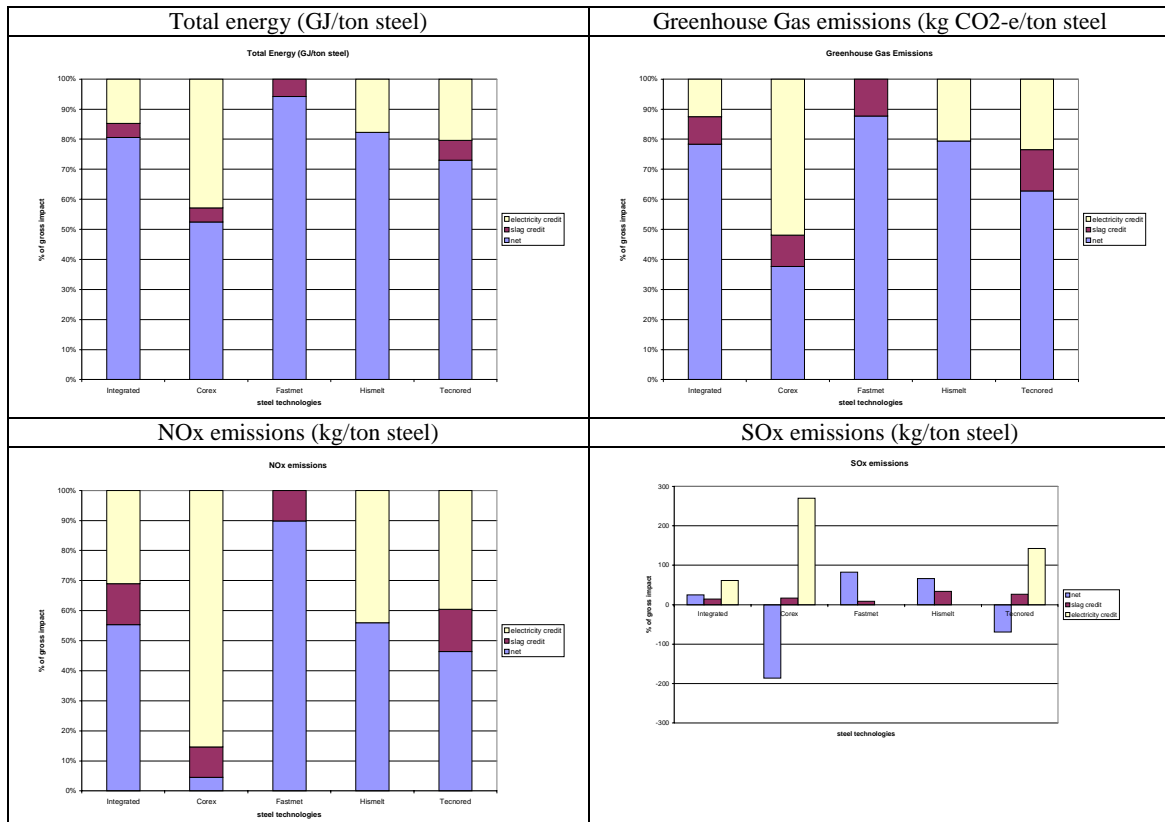
In the minerals sector, allocation will generally be needed for multi output process steps, i.e. extraction of a combined deposit, followed by distinct beneficiation processes for the different ores, and for allocation of the environmental benefits from recycling of process waste streams outside the minerals lifecycle. The former allocation can generally be done on the basis of physical or chemical relationships. The later is known as '*open loop recycling*', the waste from one process becomes an input in another process. This allocation problem is rather complex, and no single approach has been prescribed in ISO 14041. The environmental implications of open loop recycling relate to:

1. *Waste Management*: avoiding the management of the wastes from the first production system, and thereby avoiding the environmental impacts caused by waste management
2. *Virgin Material Production*: avoiding the production of virgin materials for use in the second production system, and thereby avoiding the environmental impacts caused by virgin material production
3. *Recycling Process*: introducing a recycling process (plus most often transport) for converting the waste from the first production system into a useful input to the second production system, and thereby creating the environmental impacts associated with the recycling process.

The recent Australian LCA study for the use of coal for power generation and steel production disclosed only information on the second category of environmental benefits (referred to as '*replacement*' or '*displacement*' benefits) for the use of slag and fly-ash in cement production and for use of off gases for power production (Wibberley et al, 2000). These displacement credits were all credited to the coal consuming process (i.e. steel production and electricity generation). Although this still conforms with the ISO 14041 standard, it is generally regarded as too optimistic for the first - coal consuming - process. A more accepted practice is to allocate the avoided environmental impact from not having to dispose and manage the first process waste streams to the first production system (i.e. category 1). The second production system that utilises the recycled materials is then credited with the avoided environmental impact from not producing virgin materials (i.e. category 2) and debited with the environmental impact caused by the recycling process (i.e. category 3) (e.g. Maillefer, 1996). Figure 3 shows the importance of allocation of the displacement credits for four environmental impact categories for five coal based steel production technologies. In the case of greenhouse

gas emissions, the electricity displacement credit ranges between 0 and 52% of the gross greenhouse gas emissions from steel production, and the slag displacement credit between 0-7 %. The impacts are largest for SOx emissions, where the displacement credits for electricity generation range between 0 and 270% of the gross SOx emissions and the slag displacement credit between 0 and 14%.

Figure 3: ‘Displacement’ credits for open loop recycling from coal based steel technologies (data from Wibberley et al, 2000).



Note:

Integrated = integrated steelworks (baseline technology)

Corex = technology for converting iron ore pellets to carbon saturated steel, using lump coal instead of cokes

FASTMET = process for producing direct reduced iron from cold bonded ore-coal pellets via a rotary hearth furnace

HISMELT = combination of basic oxygen steelmaking with a smelt reduction vessel

Tecnored = modification to the basic oxygen steelmaking process.

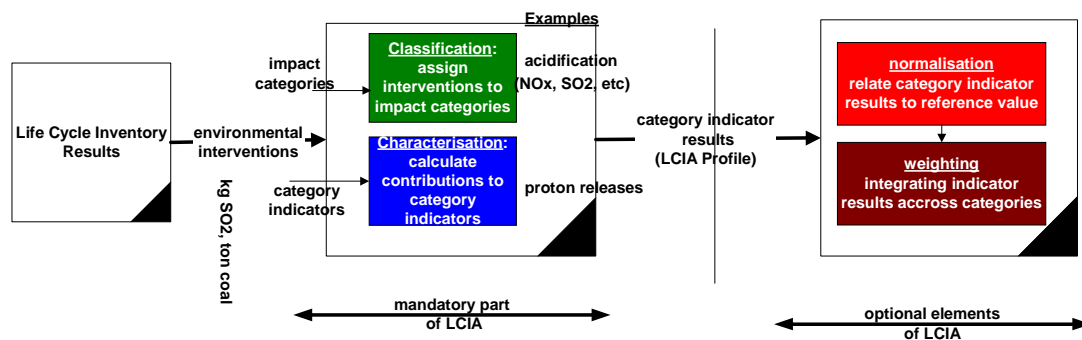
An important constraint in conducting LCI is the lack of sufficiently detailed physical data on the unit processes that make up the life cycle. In general a division is made between the ‘foreground’ and ‘background’ processes. The foreground processes are those directly linked to the production, distribution and use of the product, and it is generally aimed to use specific data for the foreground processes, either from process modelling (e.g. Wibberley et al, 2000) or from measurement of existing processes (e.g. Stewart et al, 2000; Norgate et al, 2000). By contrast, the background processes are not directly related to the product system being studied. These involve for instance the provision of energy, transport and waste management services to the production system, and the production of input materials and capital goods for the production system.

Life Cycle Impact Assessment

The Life Cycle Impact Assessment (LCIA) aims to examine the product system from an environmental perspective, using impact categories and category indicators connected with the LCI results (ISO 14042). The LCIA links the physical interventions included in the LCI with recognised environmental impact categories (*'classification'*). It then continues to estimate the relative contribution of these physical interventions to the environmental impacts of concern (*'characterisation'*). This results in 'category indicator results' and is done with characterisation models that provide a link between a physical input or output to the natural environment and the estimated incremental increase in the environmental impact.

Figure 4 provides an overview of the LCIA. Mandatory elements in the LCIA according to ISO 14042 are the selection of impact categories, the assignment of LCI results to impact categories and the calculation of category indicator results. Moreover, ISO 14042 recognises that there are optional elements and information, which can be used depending on the goal and specific requirements. Optional are: calculating the magnitude of the category indicator result relative to reference information (*'normalisation'*), sorting and possibly ranking of the impact categories (*'grouping'*), converting and possibly aggregating indicator results across impact categories using numerical factors based on value-choices (*'weighting'*) and data quality analysis. LCIA's which support comparative assertions should employ a sufficiently comprehensive set of category indicators, should be done category by category, and need to be complemented with qualitative information to address inherent limitations of the LCIA (ISO 14042). Moreover, weighting is not allowed in case of comparative assertions.

Figure 4: Structure of the Life Cycle Impact Assessment (ISO 14042)



The selection of impact categories is in principle free for the LCA practitioner. However, in general terms the selection of impact categories should reflect a comprehensive set of environmental issues related to the product system being studied, taking the goal and scope into consideration (ISO 14042). In practical terms this might be achieved through the identification of key environmental interventions from the foreground processes, and subsequent analysis of all important environmental impact categories to which those contribute. In the case of LCA in the minerals sector with resource extraction and mineral beneficiation in the foreground processes, in general terms, the key environmental interventions are use of energy, land and water, and generation of mining and minerals processing wastes (including tailings). Accordingly, the set of environmental impact categories should cover the width of environmental

impacts possibly caused by these key interventions; for instance: energy use, water use, land use (Lindeijer, 2000), ecotoxicity and human toxicity.

A division in '*stress*' and '*effect*' impact categories is still most common (Lafleur et al, 1998; Barnthouse et al, 1997). Stress categories relate to a material and energy flow that in turn is considered to be indicative for the severity of several environmental problems. Common stress categories are total energy (in MJ), total waste (in kg), land use (in m².s) and water use (in kl). On the other hand, effect categories deal with specific environmental problems, related to either resources (exhaustion of biotic and abiotic resources), damage (to ecosystem, landscape or human health) or pollution (e.g. climate change, ozone layer depletion, acidification, eutrophication, photo oxidation formation, human toxicity, ecotoxicity, heat, odour and noise).

The application of traditional LCIA practice in the minerals sector has severe limitations, as for instance revealed by a case study on a copper hydrometallurgical process (Giurco et al, 2000). With regard to human and eco-toxicity the default approach in LCA assumes that all metals within a solid waste will mobilise and impact on the environment. This default LCIA approach will therefore fail to reward process innovations that reduce the metal mobility in the process waste stream. A possible way forward is to base the ecotoxicity characterisation on the *mobile* instead of *total* metal content in the process waste stream. The metal mobility can be assessed with sequential leach tests. Sequential leach tests were performed on the copper hydrometallurgical waste, and the results showed that a significant amount of material did not mobilise from the solid residue. While iron and sulphur make up over half of the solid residue, only 10% of these metals mobilise after stage two of the sequential leach. In practical terms, it is unlikely that more than 10% of iron and sulphur will mobilise from the solid waste residue in the copper case study, even in an uncapped deposit exposed to significant rainfall. Feeding the revised estimates of metal mobility back into the LCIA had a significant impact on potential ecotoxicity and human toxicity scores which are influenced by the metals in the solid waste residue. Potential eco-toxicity decreased by a factor of 5.5 and human toxicity decreased by a factor of 3.

The practice for LCIA is rapidly developing. It is now increasingly becoming accepted that LCIA is a combination of science and informed, but value based, judgement, that the choice of impact categories to be considered is dependent on goals and scope for the LCA, and that the LCIA results in indicator results that are relative rather than absolute impact scores. In fact, there is growing reluctance to distil LCIA into a single score. No scientific process to arrive at any single score is feasible, where fundamentally different and independent processes are aggregated. Different values will lead to intrinsically different scores; the perceptions and priorities leading to the score of one party are not any more valid or important than another's (Barnthouse et al, 1997).

In recognition of such changing perceptions on scientific rigor in LCIA, consensus is growing around a new approach to LCIA, with different types of indicators that offer the LCA practitioner a menu of impact indicators to choose from (Udo de Haes et al, 1999 a and b). Conceptually this starts with defining relevant areas of protection, most often: human health, natural environment, natural resources and man-made environment. Next distinct steps in the cause effect relation between environmental interventions and those areas of protection are defined, and these are known as either midpoints or endpoints. For example greenhouse gas emissions will increase radiative

forcing, which in turn can lead to a temperature rise, a sea level rise and changes in climate patterns; all are regarded as category midpoints, and can contribute to several category endpoints, closely linked to the areas of protection. For climate change, such endpoints are e.g. loss of fish catch, loss of crops and wood and damage of wildlife and plants. The modelling of the relationships between interventions, and category midpoints and endpoints is critical, and so far such relationships have only been broadly accepted for a few midpoints (such as for instance greenhouse gas emissions, acidification and eutrophication).

Life Cycle Interpretation

This final element of an LCA deals with structuring the results from the LCI and LCIA phases in order to determine significant issues, in accordance with the goal and scope definition, and interactively with an evaluation element. The purpose of such interaction is to include the implications of the methods used, assumptions made, etc. in the preceding phases, such as allocation rules, cut-off decisions, selection of impact categories, category indicators, characterisation models, etc. (ISO 14043).

The search for significant issues in the LCI and LCIA is in principle not bound by limitations, as long as potentially significant issues are carefully reviewed and checked on completeness, sensitivity and consistency. Universal approaches for identifying significant issues are division of LCI and LCIA results by inventory data category (e.g. energy, waste), by impact category (e.g. greenhouse gas emissions, ecotoxicity) or by life cycle stages (e.g. contribution of transport to total life cycle environmental interventions and impacts).

A simple starting point for the life cycle interpretation is the break down of the LCIA and/or LCI results between '*direct*' and '*indirect*' effects. The direct effects arise from the foreground processes themselves and the indirect effects are those arising from the background processes, in principle the manufacture of inputs to the foreground processes, and the provision of transport, energy and waste management services for the foreground processes. This division has major ramifications for environmental improvement efforts. Indirect effects can only be reduced by means of reduction of the resource requirements (i.e. material, energy and other inputs) to the foreground processes. For direct effects, there is a potential to reduce environmental impacts through improvement and innovation in the foreground processes. For example in the copper hydrometallurgical case study discussed before, it was found that contributions to acidification, greenhouse effect, ozone depletion, eutrophication and smog are '*indirect*' effects, mainly resulting from the burning of coal to generate electricity used in the process. The impact categories of ecotoxicity and human toxicity are predominantly '*direct*' effects of the copper process. The company has most control over these effects, and can manage its process to reduce those potential impacts, for example by running the process in a manner that creates a more stable waste. The process modelling revealed that such is possible by optimising nominal current density and feed concentrations (Giurco et al, 2000).

APPLICATIONS

LCA has the ability to provide transparent and comprehensive information on the life cycle environmental impacts of a product system. LCA has come to fruition in

situations where such life cycle information could support decision-making, both in the public and private sectors. In very broad terms a distinction is possible between applications at the level of materials and at the level of processes. Both are in first instance based on cradle-to-gate LCA for mining and minerals beneficiation.

Comparison and selection of materials

As argued before, the minerals sector appears to be most suited for cradle-to-gate LCA. The functional unit is then defined in terms of mass of metal meeting specific standards at the exit of the refinery gate. The resulting LCA information will not be suited for comparative assertions between different minerals and metals, since material properties differ, leaving the materials as not interchangeable. Comparative assertions are only possible at the product level where alternative products based on other minerals and metals can achieve the same functionality.

Despite this, there is substantial merit in conducting cradle-to-gate LCA for assisting downstream producers in conducting LCAs for their products. The cradle-to-gate life cycle information can be converted into an exchangeable format to assist with the transparent and efficient transfer of data (such as the format promoted by the Society for the Promotion of LCA Development (www.spold.org)). Once baseline LCA data are available at the materials levels, downstream producers can include those data as background processes for LCA of their products and concentrate on obtaining and improving data for their own manufacturing and distribution processes, and the use and disposal of their products by their customers. In principle, the producer would then be able to perform the LCA in a manner that is supportive for comparative assertions.

A good example of this approach is the Worldwide Life Cycle Inventory Study for Steel Products by the International Iron and Steel Institute (IISI) (www.worldsteel.org). The objective of the initiative is to provide steel customers with an accurate and up-to-date profile of the environmental performance of steel, to enable them to conduct LCAs. IISI collected information from 55 iron and steel production sites around the world. Many of those sites, collected additional life cycle information from their suppliers include resource extraction, ore beneficiation and transport into the LCA for steel. Results have been assembled at the level of environmental interventions in order to allow different users to utilise different combinations of impact categories and characterisation models for their LCAs. Similar sector wide life cycle inventory initiatives, have been completed for example for the plastics industry and are well underway for other minerals.

Selection and improvement of processes

The LCA framework was initially developed to facilitate the achievement of reductions in the net environmental impacts of product systems. This was reflected in the early demarcation of the final stage of the LCA as '*Life Cycle Improvement Analysis*' (see e.g. Fava et al, 1991; Consoli et al, 1993). The explicit focus on improvement has however faded away over the last decade, as a result of a number of factors. Most importantly, the identification of improvement opportunities is essentially a creative and very process-, product- and site-specific affair (e.g. Van Berkel et al, 2000), and therefore does not lend itself well for rigorous generalisation in an ISO standard. Moreover, it can be argued that the Environmental Management Systems (EMS)

Standard (ISO 14001) should cover the improvement stage of the LCA, given the fact that the EMS requires continuous improvement of the environmental performance of the organisation.

Cradle-to-gate LCA is however very suitable for directing improvement efforts in the minerals sector. A distinction can be made between application in the project development stage and at operating sites. In the case of project development, the LCA should support the selection of the most environmentally preferred process ('process selection'). As no operating data are available, the process selection need to be guided by LCA based on the results of process modelling for alternative processes (i.e. alternative processing routes, process chemistries and unit operations). For operating processes, the LCA should support environmental process improvements. Only the existing process is being assessed, but the LCA needs to be more specific and detailed to identify significant environmental issues that can be addressed through process improvements.

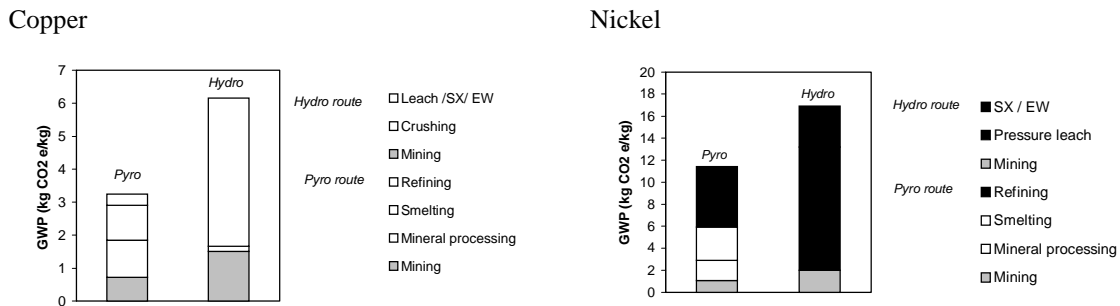
Norgate and Rankin (2000) produced a good illustration of how LCA can in principle be used for process selection. They compiled background data for a hydrometallurgical and pyrometallurgical route for both copper and nickel production, and conducted a cradle-to-gate LCA per kg of metal at the refinery gate. The LCIA was limited to two impact categories, i.e. climate change (with greenhouse gas emissions in kg CO₂-equivalent as indicator) and acidification (with acidifying emissions in kg SO₂-equivalent as indicator). The results of the study are summarised in table 1 in terms of total (or full cycle) energy consumption, Global Warming Potential and Acid Potential. These results show that nickel production is 3 to 4 -fold more energy intensive than copper and that the hydrometallurgical processes involving solvent extraction and electrowinning have higher energy consumptions and GWPs than the pyrometallurgical processes for both metals. The contributions of the various process stages to the overall GWP are shown in figure 5. The APs of the copper pyrometallurgical and hydrometallurgical processes are similar, while the AP of the nickel hydrometallurgical process is about half that of the pyrometallurgical process because of the oxide ore being treated.

Table 1. Total energy, GWP and AP for nickel and copper production (data from Norgate et al, 2000).

Metal	Process	Total energy (MJ/kg)	GWP (kg CO ₂ -e/kg)	AP (kg SO ₂ -e/kg)
Nickel	• Flash furnace smelting and Sherritt-Gordon refining (sulphide ore @ 2.3% Ni)	114	11.4 (48 %)*	0.13
	• Pressure acid leaching, solvent extraction and electrowinning (laterite ore @ 1.0% Ni)	194	16.1 (52 %)	0.07
Copper	• Smelting/converting and electro-refining (sulphide ore @ 3.0 % Cu)	33	3.3 (71 %)	0.04
	• Heap acid leaching, solvent extraction and electrowinning (sulphide ore @ 2 % Cu)	64	6.2 (93 %)	0.05

Figures in brackets are % contributions of electricity generation and supply to the overall process global warming potential.

Figure 5: Contribution of process stages to total Global Warming Potential
(data from Norgate et al, 2000).



CONCLUDING REMARKS

Even though the roots of Life Cycle Assessment go back to total energy analysis developed in the 1970's, LCA in its present form can still be regarded as an emerging environmental management tool with significant potential for supporting environmental management decisions in the public and private sectors. The strength of LCA is that it provides a framework for analysing and evaluating the environmental impacts in the different stages of the life cycle of a product system. The potential of LCA as a decision support tool is however still constrained by a number of barriers. Broad categories of barriers to the wider use of LCA are: absence of the perceived need; lack of LCA expertise and know how; lack of funding for LCA; and lack of appropriate data and methodologies (e.g. UNEP, 1999).

Even though the execution of LCA is now guided by a comprehensive set of ISO 14040 standards, there still remains a large degree of flexibility for conducting LCA to serve different decision making settings. The flexibility relates in particular to the execution of the Life Cycle Impact Assessment (i.e. impact categories, category indicators and characterisation models are not prescribed, but their choice should be motivated with reference to the goal and scope of the particular LCA), and to a lesser degree the Life Cycle Inventory Analysis (e.g. use of cut-off rules and allocation procedures). It is therefore probably better to think of LCA as a set of tools, rather than a particular tool. Generally speaking, LCAs that combine about equal levels of detail and sophistication in the LCI and the LCIA appear to make most sense.

The applicability of LCA in the minerals sector is constrained by the fact that the outputs of the sector (metals and other minerals) are generally not exchangeable, due to large differences in material specifications and performance. LCAs in support of comparative assertions for environmentally preferred materials are therefore not possible. Cradle-to-gate LCA for the resource extraction and minerals beneficiation stages can serve various applications. First of all, life cycle information generated by the minerals sector will enable downstream processors and manufacturers to conduct proper LCAs for their products and services. Secondly, LCA can be used for process improvement in the minerals sector, both in the project development stage (for process selection) as well as during the operational stages of the project (for process optimisation).

This paper revealed a number of weaknesses in the application of LCA methodology in recent LCA studies for primary resource extraction industries, such as inappropriate definition of functional unit, the generous allocation of ‘recycling credits’ for open loop recycling and the exclusion of obvious direct impact categories. To increase public trust in cradle-to-gate LCAs in the minerals sector, it is important to ensure that best LCA practice is being applied for conducting LCAs in the minerals sector.

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