A hybrid life cycle assessment model for comparison with conventional methodologies in Australia

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Abstract

Background, aim, and scope One barrier to the further implementation of LCA as a quantitative decision-support tool is the uncertainty created by the diversity of available analytical approaches. This paper compares conventional ('process analysis') and alternative ('input–output analysis') approaches to LCA, and presents a hybrid LCA model for Australia that overcomes the methodological limitations of process and input–output analysis and enables a comparison between the results achieved using each method. A case study from the water industry illustrates this comparison.

Materials and methods We have developed a tiered hybrid model for calculating the life cycle impacts of a system. In so doing, we have developed a novel way of overcoming a key methodological issue associated with this method: avoiding double counting. We calculate 'system incompleteness factors' and use these to delete the lower-order burdens in the input–output inventory according to the depth of production taken into account in the process inventory. We apply this method to a case study of Sydney Water Corporation. The functional unit is *the provision of water and sewerage services to residential, industrial, and commercial customers in the city of Sydney in the year 2002/03*.

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Results and discussion We analysed the case study using three methods: process analysis, input–output analysis, and hybrid analysis. In each case, we obtained results for eight impact categories: water use; primary energy use; global warming potential; carcinogenic and non-carcinogenic human toxicity potentials; and terrestrial, marine and freshwater ecotoxicity potentials. Although the process analysis has a relatively shallow investigative depth, it shows good system coverage (i.e. a small truncation error) for most indicators. The truncation errors for all of the indicators except marine aquatic ecotoxicity potential compare favourably with predicted truncation errors for the relevant industry sector. This suggests that the truncation error of a particular process analysis cannot be accurately predicted using generic system completeness curves, and implies that the truncation error of a typical process analysis may be less severe than is commonly generalised by the proponents of input–output analysis.

Conclusions The case study supports the largely theoretical claims in the literature about the relative merits and drawbacks of process and input–output analysis. Each method has the potential to highlight different aspects of the system. By estimating the truncation error of the process analysis independently of the relationship between the results obtained using the other methods, our hybrid model enhances the ability to investigate the differences between results and thus adds considerable value to such a study.

Recommendations and perspectives Input–output LCA has become more popular as computational tools have become more accessible. We directly compare input–output, process and hybrid LCA and recommend that, from an environmental analysis perspective, it would be beneficial to consider the three methodologies in parallel. We highlight the potential for misinterpretation of differences between methods that rely on different reporting frameworks, and

recommend that LCA method and practice continue to emphasise the role of careful interpretation.

1 Background, aim, and scope

For LCA to be a successful decision-support tool, especially for small- and medium-sized enterprises and developing countries, it should be both technically rigorous and practical to apply (UNEP/SETAC 2005). We therefore need robust, easy-to-use tools for LCA, combined with informed interpretation of the results. This paper presents a tiered hybrid analysis method that overcomes the limitations of stand-alone process and input–output analyses. A case study from the water industry illustrates the comparison between the three methods.

1.1 Process analysis

Process analysis is the conventional method of life cycle inventory (LCI) compilation. It involves detailed study of resource uses and environmental releases from on-site production, and contributions from suppliers of inputs considered significant by the analyst (Suh et al. 2004). There are two generally accepted approaches to process analysis (Suh and Huppes 2005): the commonly-used process flow diagram approach; and the matrix inversion approach introduced by Heijungs (1994).

Process analysis requires the analyst to draw a system boundary, beyond which the upstream impacts are considered to be negligible (Suh et al. 2004). However, "in modern economies all industry sectors are dependent on all other sectors, and this process of industrial interdependence proceeds infinitely in an upstream direction, through the whole life cycle of all products, like the branches of an infinite tree" (Lenzen and Wood 2003, p 20). This is illustrated in Fig. 1. We refer to the environmental burdens
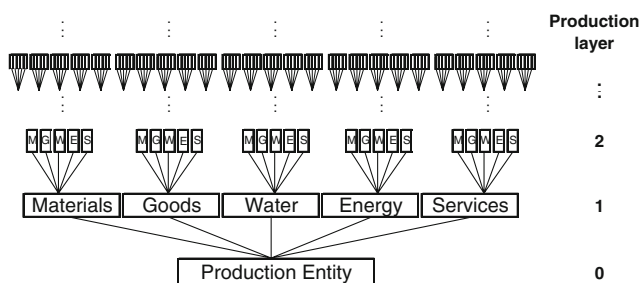
resulting directly from the operations of the production entity under study as *zero-order* burdens, while the environmental burdens resulting from other processes on which that production entity depends are *higher-order* burdens.[1] In this paper, unless otherwise specified, these terms refer to the system as a whole, with the production entity under study located on the *zero-order* layer. It has previously been suggested that the truncation error caused by drawing the system boundary in a process flow diagram approach could be up to 50%, depending on the industry sector (Lenzen 2000).

The matrix inversion approach to process analysis also suffers from a truncation error as it fails to account for additional upstream inputs, although it can account for infinite orders of relations between those upstream processes that are already included within the system boundary (Suh et al. 2004).

Both approaches to process analysis generally also have high labour and resource requirements, due to the large amount of process-specific primary data on material and energy requirements that must be collected and analysed.

1.2 Input–output analysis

Input–output analysis (IOA) is an economic modelling technique that "uses sectoral monetary transactions data to account for the complex interdependencies of industries in modern economies" (Lenzen 2002). It was introduced by Leontief (1941), who later explained how it could be used as a method of analysing 'externalities' such as environmental impacts (Leontief 1970). The method is briefly explained here.

When used in LCI compilation, IOA separately considers *zero-order* and *higher-order* environmental burdens.

Zero-order environmental burdens are calculated by obtaining a matrix of 'direct requirements'[2] $\mathbf{Q}$ (in 'impact units'/\$) in which each entry q_{ij} represents the magnitude of burden i as a result of the *zero-order* operations of industry j, per dollar of output from that industry. Using a pure input–output approach, the vector $_z\mathbf{g}$, of length N, which represents the environmental burdens (in 'impact units', on N environmental indicators) of a production entity's *zero-order* operations, can then be estimated by multiplying the value of the output (\$) of that entity by the entries ('impact

Fig. 1 Industrial interdependence in the economy (after Lenzen and Wood 2003)

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[1] Alternative nomenclature for zero-order and higher-order burdens includes the use of 'on-site' and 'off-site', but these terms have special meaning in process analysis and often imply a physical 'site' or plant location so are avoided here to minimise confusion. Zero-order and higher-order burdens are elsewhere also referred to as 'direct' and 'indirect' burdens.

[2] Note that $\mathbf{Q}$ is also known as the 'environmental interventions' matrix (Suh and Huppes 2005).

unit'/$) in the column of the industry sector to which it belongs.[3]

In calculating the *higher-order* environmental burdens, IOA uses a 'transactions matrix' **Z** in which each entry z_{ij} represents the financial flow of commodities from industry i to industry j. This matrix can be expressed as **A**, in which the entries of **Z** are divided by the total output of industry j. The matrix **A** then represents, in proportionate monetary terms, the input of commodities that each industry requires from all the other industries (and itself), in order to produce one unit of output (i.e. units of [$/$]).

To produce the 'inputs' required by other industries, each industry also requires further inputs from its own suppliers, and the suppliers of their suppliers, and so on. This creates an infinitely extensive supply chain.

It has been shown (e.g. Leontief 1970) that this infinitely extensive supply chain can be expressed in matrix form. Furthermore, the sum of this infinite series can be calculated using what is now known as the 'Leontief inverse'. This allows calculation of a 'multiplier matrix' **M**, in which each entry m_{ij} ('impact unit'/$) represents the *total* environmental burden i caused by one dollar of output from industry j. The system's *higher-order* burdens $_h\mathbf{g}$ can thus be calculated using the system's expenditure **y** across each industry sector, according to the equation:

$$_h\mathbf{g} = \mathbf{My} = \mathbf{QLy} = \mathbf{Q(I-A)}^{-1}\mathbf{y} \qquad (1)$$

where **I** is an identity matrix with the same dimensions as **A**. The multiplier matrix needs only to be updated at appropriate intervals, rather than for each application of the model. This gives IOA a practical advantage over process analysis in that, once the multiplier matrix has been constructed, the higher order burdens can be computed using only the expenditure data, which is common business information.

The system's *total* environmental burdens can then be computed simply by adding the *zero-order* and *higher-order* burdens on each environmental indicator, i.e.:

$$_{IOA}\mathbf{g} = {}_z\mathbf{g} + {}_h\mathbf{g} \qquad (2)$$

Because IOA can take account of the entire production chain (i.e. *zero-order* and all *higher-order* burdens) it is able to solve the upstream 'system boundary problem' or

truncation error associated with process analysis. However, input–output inventories are also subject to limitations, including (Heijungs and Suh 2002; Suh and Nakamura 2007):

- Aggregation error introduced by a coarse industry classification scheme[4];
- The assumption that environmental burdens arising abroad from the production of imports are identical to those generated by equivalent domestic production, which may be a significant source of error for a severely import-dependent national economy, or one that trades with economies with very different production characteristics (e.g. energy sources);
- The varying completeness of the national datasets used (input–output tables and direct requirements data); and
- The assumption of proportionality between the financial flows and the physical flows of commodities in the input–output table (i.e. the uniformity of prices for a particular commodity within the national economy).

In addition, national input–output tables are often published with a delay of a few years or more.[5] This may have a significant effect on the analysis of highly dynamic sectors, such as the sector in which mobile communications technology is manufactured. It may also have an effect where commodity prices have changed dramatically, such as in the natural resource extraction sector.

1.3 Hybrid analysis

To draw on the main advantages of both process and IOA—specificity and completeness respectively—hybrid approaches have been proposed and developed (e.g. Bullard et al. 1978). Using a hybrid approach, the truncation error in process analysis can be reduced (Lenzen 2000) whilst maintaining sufficient detail to meaningfully compare two very similar products or systems (Heijungs and Suh 2002).

Three types of hybrid analysis are identified in the literature: tiered hybrid analysis, input–output based hybrid analysis, and integrated hybrid analysis. Only the former is discussed here. The other approaches are discussed by Suh and Huppes (2005) and Heijungs and Suh (2002), and summarised in Table 1.

In a tiered hybrid analysis, process based data are gathered for the use and disposal phases, and some important upstream processes. The remaining upstream processes are then modelled using IOA, and the two datasets are simply added together (Suh and Huppes

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[3] This is a 'pure' input–output approach. In practice, we commonly use process data to calculate the zero-order environmental burdens where those data are available. This is a simple form of tiered hybrid analysis.

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[4] The industry classification scheme used in Australian input–output tables is limited to 106 sectors, so a single sector can contain many dissimilar industries. This introduces an error due to the use of averages throughout the analysis.

[5] This varies between countries, but is true for Australia.

Table 1 Application of the hybrid approaches to the components of the life cycle

	Higher-order processes	Zero-order processes	Downstream processes	Method of connection
Tiered hybrid analysis	Some process analysis, some IOA	Process analysis	Process analysis	Manual addition of results
Input–output based hybrid analysis	Partially disaggregated IOA	Partially disaggregated IOA	Process analysis	Manual addition of results
Integrated hybrid analysis	Some process analysis, some IOA	Process analysis	Some process analysis, some IOA	Matrices are fully inter-linked

2005). This approach utilises detailed process analysis in the conventional way where possible, supplementing what is 'missing' with IOA (Heijungs and Suh 2002).

A number of methodological issues have been identified with respect to tiered analysis (Suh and Huppes 2005, p 691), including the need to carefully select the interface between the models; the potential for double-counting of flows included in both the process and input–output analyses; and the inability to model interaction between the systems. The distinguishing feature of our tiered hybrid analysis method is the way in which it addresses double counting.

1.4 Aim and scope

Our research aims were to:

- Develop a hybrid LCA model for Australia that combines the advantages of process and input–output models and overcomes the methodological limitations associated with combining them;
- Enable a comparison between the results achieved process analysis, IOA and hybrid analysis by applying each method to a single case study; and
- Using the case study, investigate the validity of the largely theoretical claims about the relative merits and drawbacks of process and IOA.

We do not wish to recommend any one of the three methods as being most appropriate to any particular situation. Rather, we seek to enhance comprehension of the alternative methods, thereby enabling informed analysis and decision making by LCA practitioners, policy makers and others, particularly in the Australian context.

2 Materials and methods

2.1 Developing a hybrid LCA model

The environmental burdens $_{hybrid}\mathbf{g}$ of a system, calculated using a tiered hybrid analysis, can be simply expressed as (Heijungs and Suh 2002):

$$_{hybrid}\mathbf{g} = {_{process}}\mathbf{g} + {_{IOA}}\mathbf{g} = {_{process}}\mathbf{g} + \mathbf{Q}(\mathbf{I} - \mathbf{A})^{-1}\mathbf{y} \qquad (3)$$

where $\mathbf{Q}$, $\mathbf{I}$, $\mathbf{A}$ and $\mathbf{y}$ are familiar notation from previous discussion.

Selecting an appropriate system interface location is relatively straightforward for the LCA practitioner, because it is methodologically identical to the selection of a system boundary location in conventional process analysis. As with the latter case, the compromise between system completeness and resource availability must be resolved and associated decisions documented.[6]

A more significant issue is avoiding double counting. Simply adding the results of a process LCA and an IOA of the same system will erroneously include the system components modelled by processes analysis twice. For example, the environmental burdens caused by electricity use may be determined by process analysis, and also appear as expenditure allocated to the electricity supply sector in the IOA.

One rather obvious method of avoiding double counting is to convert the process data into equivalent dollar values using commodity prices, subtract these dollar values from the expenditure vector $\mathbf{y}$, and carry out the IOA as usual, with some elements of $\mathbf{y}$ reduced. However, there are drawbacks to this approach. For one, using commodity prices would be problematic because it would introduce an additional dependency on approximations. Furthermore, commodity prices for each specific element of the process analysis are not always available due to reporting limitations or confidentiality requirements.

We have therefore devised an alternative approach to avoiding double counting, which overcomes these drawbacks. We calculate 'system incompleteness factors' and use these to delete the lower-order burdens in the input–output based inventory according to the estimated depth of production that has been taken into account in the process based inventory. The input–output inventory then represents only the 'missing' or truncated inventory and is suitable to add to the process based inventory.

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[6] If anything, the correct selection of a system interface in hybrid analysis is less crucial than the selection of a system boundary, since in the former case the truncated processes are included in the IOA.

2.1.1 Deriving system incompleteness factors

We have seen that IOA gives a system's *higher-order* burden $_h\mathbf{g}$ across all environmental indicators (in 'impact units') as:

$$_h\mathbf{g} = \mathbf{QLy} \tag{4}$$

which can be written as:

$$_h\mathbf{g} = \mathbf{Q}(\mathbf{I} - \mathbf{A})^{-1}\mathbf{y} \tag{5}$$

and ultimately expanded to the infinite series:

$$_h\mathbf{g} = \mathbf{Qy} + \mathbf{QAy} + \mathbf{QA}^2\mathbf{y} + \mathbf{QA}^3\mathbf{y} + ... \tag{6}$$

where the first term in the series represents the burdens of the first production order in the system (i.e. *zero-order* burdens are not included).

The expanded series, given in Eq. 6, can be used to obtain generic 'system completeness' curves for each industry sector's performance on each environmental indicator (cf Fig. 1 in Lenzen 2000). These curves are obtained by comparing the accumulation of successive terms of the series to the sum of the infinite series as calculated using the Leontief inverse in Eq. 5.

The fraction of $_h\mathbf{g}$ that is computed in an analysis that includes k production orders is commonly referred to the degree of 'system completeness' at production order k. We now also refer to the fraction of $_h\mathbf{g}$ that is excluded from such an analysis as a 'system incompleteness factor' (SIF). The unique SIF for industry sector j and environmental indicator n at production order k can be expressed as:

$$SIF_{j,n,k} = \frac{g_{j,n,\infty} - g_{j,n,k}}{g_{j,n,\infty}} = 1 - \frac{g_{j,n,k}}{g_{j,n,\infty}} \tag{7}$$

We have derived a SIF for each of the 106 Australian industry sectors, on each environmental indicator, for each production order up to $k=20$ (by which point, system completeness is within 5×10^{-5} of 1).

2.1.2 Applying the system incompleteness factors

We wish to delete the lower-order burdens in the input–output inventory, according to the depth of production that has been taken into account in the process based inventory. For a process based inventory that includes $k = K$ orders of production with reference to the entire system, we will thus obtain an estimate $_{total}\hat{\mathbf{g}}$ of the total effects of industry sector j across N environmental indicators where:

$$_{total}\widehat{g}_{j,n} = \sum_{k=0}^{K} {}_{process}\, g_{j,n,k} + {}_h\, g_{j,n} \times SIF_{j,n,K} \tag{8}$$

Note the inherent assumption that the *zero-order* environmental burdens $_z\mathbf{g}_j$ are accounted for in the process analysis.

This method may be explained using an example. Let us assume that in a given process analysis, the required input from 'Sheep farming' is included up to the first production layer.[7] Let us also assume that the system completeness for the sector 'Sheep farming' has been calculated to be 75% at a relative production layer of $k=0$ (i.e. $k=1$ with reference to the entire system). We can then multiply the *higher-order* input–output result for this sector by $100-75 = 25\%$ (i.e. the SIF), and add the result of the process analysis.

Applying this hybrid model requires the practitioner to know or estimate the 'depth' (in production orders) to which each industry sector was studied in the process analysis. This is done by examining the process LCI documentation. It should be noted that these 'depths' will be a 'best fit' estimate only, as in practice they cannot be as simply defined as they are here.

After completing the hybrid analysis, the overall truncation error of the process analysis can be estimated by simply subtracting the process based result from the hybrid result for each environmental indicator (see Table 2).

2.2 Case study: Sydney Water Corporation

We have applied the above method to a case study of Sydney Water Corporation (SWC), with the functional unit defined as *the provision of water and sewerage services to residential, industrial, and commercial customers in the city of Sydney in the year 2002/03*. A process LCA with the same functional unit but different reference year was undertaken by Lundie et al. (2004). Those authors obtained reliable, system-specific LCI data through process monitoring on site at SWC. We scaled those LCA results to our reference year using the ratio of total water supplied in each case. We also redefined 'water use' to be consistent with the definition of water use in the input–output data (i.e. losses, leakage and use within SWC operations, rather than total water supplied to customers).

We constructed an input–output model of the Australian economy using data from the Australian Bureau of Statistics (ABS 2004). The input–output model was extended for hybrid analysis according to the method described above.

Environmental emissions data for 2002/03 were obtained from the National Pollutant Inventory (NPI) (DEH 2004) and were allocated to industry sectors and emission compartments.[8] Primary energy usage and greenhouse gas emissions data have been allocated to industry sectors in previous work

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[7] Recall that this refers to the entire system, i.e. it includes orders $k=0$ and $k=1$ where $k=0$ represents *zero-order* burdens.

[8] The term 'compartment' refers to the initial receiving environment of an emission, in this case distinguishing between rural and urban air, fresh and sea water, and agricultural and industrial soil.

Table 2 Summary of case study results, with the estimated and predicted truncation error of the process analysis, according to the generic system completeness for the industry sector '*Water supply; sewerage and drainage services*'

Environmental indicator[a]	Process LCA only	Input–output LCA only			Hybrid LCA	Predicted truncation error for a process LCA with k included production orders			Estimated truncation errors[b]	
		Zero-order	Higher-order	Total		k=0	k=1	k=2	Process LCA	IOA
Primary energy use (MJ)	8.4×10^9	$1. \times 10^9$	6.8×10^9	8.4×10^9	1.1×10^{10}	97%	43%	21%	21%	21%
Water use (GL)	5.5×10^1	5.5×10^1	5.3×10^0	6.0×10^1	5.8×10^1	7%	1%	1%	6%	−3%
GWP (kg CO_2-eq)	7.4×10^8	3.9×10^8	6.0×10^8	1.0×10^9	9.5×10^8	97%	47%	26%	23%	−5%
HTP (carcinogenic)	5.7×10^1	2.5×10^{-2}	2.3×10^0	2.3×10^0	5.9×10^1	94%	69%	40%	3%	96%
HTP (non-carcinogenic)	4.6×10^2	7.8×10^1	1.4×10^1	9.2×10^1	4.7×10^2	41%	25%	15%	2%	80%
Freshwater aquatic ETP	2.6×10^{-7}	5.5×10^{-8}	8.5×10^{-8}	1.4×10^{-7}	3.0×10^{-7}	12%	3%	2%	12%	53%
Marine aquatic ETP	4.7×10^{-5}	1.6×10^{-5}	1.4×10^{-4}	1.6×10^{-4}	1.1×10^{-4}	12%	4%	2%	55%	−51%
Terrestrial ETP	7.0×10^{-7}	1.5×10^{-10}	1.0×10^{-7}	1.0×10^{-7}	7.7×10^{-7}	13%	5%	3%	10%	87%

[a] *GWP* global warming potential; *HTP* human toxicity potential; *ETP* ecotoxicity potential

[b] Difference between the hybrid LCA results and the process LCA or IOA results, as a percentage of the former; see Section 2.1.2 for details

(Lenzen and Lundie 2002), as have water use data (Foran et al. 2005). SWC expenditure data for 2001/02 were derived from Lenzen et al. (2003). Although there is some disparity in the reference years of the datasets, it was beyond the scope of this work to attempt to reconcile them.

Preliminary analysis demonstrated that the results of a pure IOA were considerably less reliable than the process based results (Peters et al. 2006). This was especially true for the estimation of *zero-order* burdens. Therefore, we calculated the input–output based results using a form of tiered hybrid analysis, with the interface set between the *zero-order* and *higher-order* processes. We estimated the *zero-order* environmental burdens from SWC publications (SWC 2003) and data reported to the NPI (DEH 2004).

For all methods, characterisation for human toxicity and ecotoxicity was carried out using Australian characterisation factors published by Lundie et al. (2007).

3 Results and discussion

The results obtained for the case study of SWC using each of the methodologies are summarised in Table 2. They are also presented in Fig. 2 as percentages of the process based results.
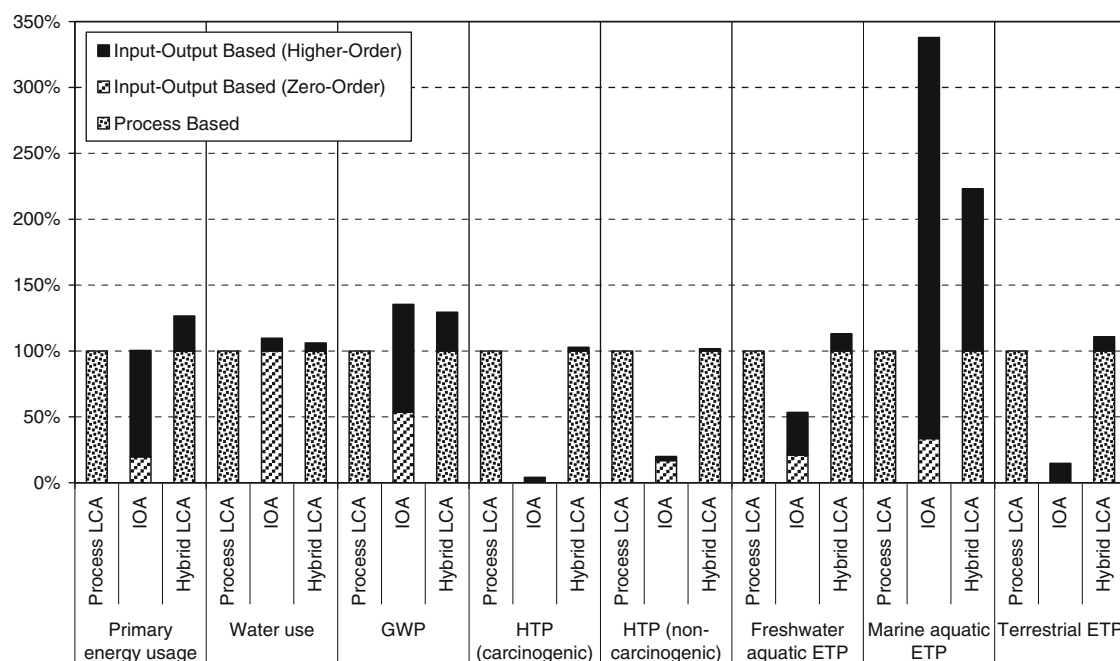
3.1 Environmental indicators

The input–output based result for freshwater aquatic ecotoxicity potential (ETP) was smaller than the process based result. Inspection of the data revealed that the

process based result was dominated by copper emitted to agricultural soil during application of biosolids (48%), and freshwater nickel emissions by inland wastewater treatment plants (22%). By contrast, most of the *zero-order* result in the input–output result was caused by chlorine emissions to freshwater. The IOA did not capture the more significant downstream emissions. The hybrid analysis was able to compensate for the 12% truncation error of the process analysis while retaining the downstream burdens to give a more complete picture of the total burden.

The results for marine aquatic ETP were interesting because in the initial analysis, the *zero-order* burden in the IOA was calculated to be approximately 120 times the magnitude of the process based result. Inspection of the datasets revealed that this was primarily attributable to fluoride compounds emitted to water. Fluoride compounds were not reported in the process analysis but appear in the NPI substance list. Analyses based on these two different reporting frameworks were thus not meaningfully comparable. Further, there is no way of interpreting which result is more accurate. This demonstrates that LCA is a tool better suited to comparative analyses (between systems or products) than to drawing conclusions based upon absolute impacts.

To overcome the problem in this analysis, the reporting frameworks were standardised by removing emissions of fluoride compounds to water by the water and wastewater industry sector from the NPI dataset. The revised results reveal a large contribution from *higher-order* processes in the input–output based result. We hypothesise that this is probably due to a higher system truncation error than the

GWP = Global warming potential; HTP = Human toxicity potential; ETP = Ecotoxicity potential

Fig. 2 Comparative results for the SWC case study, relative to the process LCA result for each environmental indicator. GWP = Global warming potential; HTP = Human toxicity potential; ETP = Ecotoxicity potential

55% error indicated by the hybrid analysis, or the presence of one or more further anomalies between the reporting frameworks. However, more detailed modelling would be necessary to investigate this further.

The input–output result for terrestrial ETP is much smaller than the process result. This is primarily due to the emission of heavy metals to agricultural soils during biosolids application, which is not captured by the IOA because it is a 'downstream' activity. In particular, chromium (VI) and copper contribute 82% of the total process result (43% and 39% respectively). In this case, the hybrid analysis is again able to compensate for the system truncation error whilst retaining the burdens caused by downstream activity (identified in the process analysis), to give a more complete result than either of the other methods.

The process analyses of human toxicity potential (HTP) (carcinogenic) and HTP (non-carcinogenic) have small truncation errors of approximately 3% and 2% respectively. In both cases, the process result is much larger than the input–output based result, primarily due to a large contribution from *zero-order* and 'downstream' processes that are not captured by the IOA. The process analysis of HTP (carcinogenic) is dominated by selenium emitted to seawater (60%) and cadmium to agricultural soil (36%). HTP (non-carcinogenic) is dominated by seawater emissions of selenium (82%) and arsenic (12%).

The process analyses of global warming potential, primary energy usage and water use have estimated truncation errors of approximately 23%, 21% and 6%

respectively. The trend in results for these indicators is more consistent with the expected trend as the results for each method are similar in magnitude.

In the case of primary energy usage, the use of the hybrid analysis alongside the other methods provides particular insight. Had the process and input–output analyses been carried out independently of the hybrid, the analyst would most likely have concluded that since the results were almost identical, they both gave a reasonably 'correct' answer. However, the hybrid analysis reveals that this similarity is coincidental, and that a more accurate result is more than 20% greater than either of the other results.

3.2 Comparison of estimated and predicted truncation errors

As described above, SIFs are calculated for each industry sector, on each environmental indicator, for each production order (a total of 17,808 SIFs). At the zeroth production order (i.e. $k=0$), system completeness ranges between 0 and 97%. It increases with successive production orders, converging to 1. The shape of the convergence curve depends on how the higher-order environmental burdens of each industry sector are distributed through the supply chain.

In the hybrid model, the appropriate SIF is applied to each industry sector for each indicator, according to the production layer included in the process analysis. By

observing the overall effect on the total input–output result, the application of these generic[9] SIFs thereby enables an estimation of the overall truncation error of the specific process analysis under study, for each environmental indicator. These results are presented in Table 2, alongside the predicted truncation errors for a process analysis in the '*Water supply; sewerage and drainage services*' industry sector with an overall completeness of zero, one, or two production orders, according to the generic system completeness curves.

It was only during the calculation of the hybrid analysis that the truncation errors for each indicator were estimated. This is because they are specific to the particular process analysis. In contrast, most other work on system completeness (e.g. Lenzen 2000) is founded on generic system completeness curves within each industry sector.

The investigative depth of the process analysis was relatively shallow. It included *zero-order* processes, 3 industry sectors in the *first order*, and 2 industry sectors in the *second order*. These depths were estimated by inspecting the process data and model. Sensitivity testing determined that the estimates were acceptably robust, with an increase or decrease of one order generally producing less than a 5% variation in the results.

Using these specific investigative depths, the overall investigative depth may be estimated at somewhere between the *zero-order* and *first order* layers of production. We expect this to be reflected in Table 2.

In general, the estimated truncation error of the process based result is similar to or better than expected. For example, the terrestrial ETP result of 10% lies between the zero-order and first order predictions of 13% and 5%. Meanwhile, the results of 23% for global warming potential and 21% for primary energy use are equal to or better than their second order predictions of 26% and 21% respectively. This may reflect the deeper investigation of electricity supply and transport in the process analysis.

In extreme cases, the estimated truncation error is much smaller than even the second order prediction. This applies to HTP (carcinogenic) and HTP (non-carcinogenic), with estimated truncation errors of just 3% and 2% respectively.

Marine aquatic ETP is an exception to this overall pattern, with its estimated truncation error of 55% far exceeding even the zero-order prediction of 12%.

These results suggest that the truncation error of a particular process analysis cannot be accurately predicted using a generalised approach and generic system completeness curves. Since this case study may be considered relatively shallow in its investigative depth, our results imply that the truncation error of a typical process analysis may be less severe than is commonly generalised by the proponents of IOA.

4 Conclusions

The case study results support the largely theoretical claims in the literature about the relative merits and drawbacks of process LCA and IOA. Each method has the potential to highlight different aspects of the system. This is particularly useful in the water services industry where there has been much debate about whether the greatest environmental burden is caused by nutrient discharge (downstream) or electricity use (upstream).

This study has also enabled the calculation of a hybrid result that is virtually independent of the relationship between the results obtained using the other methods. This helps to identify the cause of differences between the process based and input–output based results and reduces the likelihood of drawing false conclusions about the relationship between the other results (e.g. the coincidental similarity of the primary energy use results that are shown by the hybrid analysis to cover different aspects of the system). Thus, the hybrid analysis added great analytical value to the study.

5 Recommendations and perspectives

Although the data used in our work are specific to Australia, the underlying method we describe could be applied in any country or region where input–output tables are compiled. Although IOA reduces the time and resource requirements of LCA, it has its own limitations, including the influence of rapidly-fluctuating commodity prices. The hybrid model addresses some key weaknesses of each method.

From an environmental analysis perspective, it would be beneficial for LCA practitioners to consider the three methodologies in parallel. In particular, this study has also highlighted the potential for differences in reporting frameworks to cause massive anomalies between methods if they are applied without detailed interpretation. This casts some doubt on the appropriateness of wide-scale use of commercially available input–output tools by non-specialists.[10] We recommend that LCA method and practice continue to emphasise the role of careful interpretation.

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[9] They are 'generic' in the sense that may be applied to any system and are not specific to the system under study.

[10] We specify input-output models here because process analysis tools by their nature require some level of professional knowledge and experience, whereas the very attraction of marketed input–output tools is the simplicity of their use.

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