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Ecosystem Services in Life Cycle Assessment - Part 1: A computational framework

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Abstract

Explicit inclusion of the role of ecosystems in life cycle assessment (LCA) is needed to prevent the selection of alternatives that depend on or degrade scarce ecosystem services (ES), and to help identify opportunities for enhancing sustainability by not just reducing impact but also protecting and restoring ecosystems and the diverse goods and services they supply. Various approaches have been suggested for including ES in LCA but a general computational framework is not yet available. This paper extends the framework of conventional process LCA to assess and encourage techno-ecological synergies in life cycle assessment (TES-LCA). It includes ecosystem modules along with process modules in LCA. Analogous to the technology matrix in conventional LCA, TES-LCA defines a "techno-ecological" matrix. It consists of four components: a technology matrix defined by economic flows, an intervention matrix interpreted as the ES demanded by technological activities, an ecosystem matrix interpreted as the capacity of ecosystems to supply these services, and a management matrix to capture the interaction between technological and ecological systems. This work demonstrates the computational structure through a toy example and discuss the major challenges of TES-LCA in terms of data availability for an exhaustive array of ES. This work suggests that such data need to be made available and included in future versions of life cycle inventory databases. The computational structure of TES-LCA is able to capture the interactions between and within technological and ecological systems. It enables including of the role and capacity of ecosystems in a life cycle. The framework can encourage development of data and models to enable practical use of TES-LCA, which can provide unique insights into absolute environmental sustainability by quantifying overshoots for specific ES, and help identify improvement strategies based on improving technological efficiency and restoring ecosystems.

Keywords: Computational structure, Life cycle assessment, Ecosystem service, Environmental sustainability

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1. Introduction

Currently, life cycle assessment (LCA) is among the most widely used approaches for assessing and designing sustainable systems (Liu et al., 2018). LCA prevents the shifting of impacts by considering a broad system boundary (Rebitzer et al., 2004). The methodology has been standardized, with various computational structures available for constructing process, input-output (Heijungs and Suh, 2002) and hybrid life cycle networks (Suh, 2004). Fundamental process models have also been integrated with hydrid LCA by the process-to-planet framework to enable multiscale sustainable process design (Hanes and Bakshi, 2015; Ghosh and Bakshi, 2017).

Nonetheless, LCA makes decisions by comparing alternatives in terms of their resource use and emissions, thus providing only relative sustainability metrics (Bjørn et al., 2015). This relative approach prefers options that have smaller environmental impacts. However, it does not consider nature's carrying capacity (CC) to handle these impacts and might even result in perverse decisions that increase reliance on goods and services from vulnerable or degraded ecosystems (Urban and Bakshi, 2009). Therefore, in order to obtain absolute environmental sustainability metrics, the role of ecosystem services (ES) needs to be referenced explicitly and ecological CC must be respected (Bakshi et al., 2015).

Existing efforts to incorporate ES into sustainable decision making include connecting ES to LCA, allocating planetary boundary (PB) to smaller scales and utilizing CC information to define characterization factors (CF). Isard proposed that a fully integrated economic-ecological model should be developed by creating and linking an ecosystem submatrix with economic input-output model (Isard, 1972). However, this model has never been fully implemented (Miller and Blair, 2009). Othoniel et al. suggested that an impact category regarding ES should be included in LCA with related indicators (Othoniel et al., 2016). This requires a cause-effect chain to be completed to characterize the impacts of environmental intervention flows on ES provisioning. So far, the main focus is on land-use driven impacts (Koellner et al., 2013). However, this approach ignores ecological CC and thus provides information only about relative sustainability (Bjørn, 2015). Zhang et al. (2010) developed the Eco-LCA tool that quantifies the direct and indirect dependence of activities in different economic sectors on ES, based on the monetary throughput associated with the activities. This approach adopted a "top-down" approach, which disabled multi-scale decision making. Moreover, it did not specifically propose the concept of absolute environmental sustainability.

To account for absolute environmental sustainability, the impacts from human activities need to be compared to nature's remedial capacities. Attempts have been made to allocate PB to support decisions at product and corporate levels with LCA methodology. On one hand, LCA impact categories were linked to PB to develop normalization references at midpoint level (Bjørn and Hauschild, 2015) and set impact reduction targets (Sandin et al., 2015). On the other hand, novel CF are developed to incorporate carrying capacity information. Doka (2015) modified conventional CF to characterize how much a persons annual allowance of PB would be appropriated for a unit of emission created. However, as has been pointed out by its developers, the PB framework is not designed to be down-scaled to support decisions at product level (Steffen et al., 2015). Furthermore, the utilization of allocated values from the global scale ignores the concept of serviceshed, which defines the area from where users can receive ES (Tallis et al., 2013). Accounting for the serviceshed of an ES is essential for understanding its absolute sustainability.

Bjørn et al. (2016) modified the CF in life cycle impact assessment (LCIA) to incorporate information of grid-specific CC. The CF were expressed as CC occupation, with unit of ha·year/kg emission, normalized by the CC of the receiving units. The authors suggested that the developed CF can be linked to life cycle inventory to characterize the impacts of emissions within each unit. For an individual system and a specific time frame of its intervention, the midpoint indicators calculated the land area that supplies the demanded ES. If the system occupied more land area than what is allocated to it, then absolute environmental sustainability cannot be claimed (Bjørn et al., 2016). However, this method does not take into account interactions between ecosystem components, rendering it insufficient to consider ES trade-offs and synergies. Moreover, the authors regarded ecological CC as a static value calculated from a steady-state threshold, ignoring the existing environmental interventions from the ecosystem itself. This approach also does not consider the serviceshed of ES and the fact that absolute sustainability can only be defined at this scale. Opportunities such as ecosystem restoration and protection were also not considered.

The Techno-Ecological Synergies (TES) framework has the advantage of explicitly incorporating ES in the assessment boundary by developing synergies between technological activities and their surrounding ecosystems (Bakshi et al., 2015). Applications of TES has been mainly focused on assessing and designing localized systems, such as a residential house (Urban and Bakshi, 2013), a biodiesel manufacturing site (Gopalakrishnan et al., 2016), and a local energy production system (Martinez-Hernandez et al., 2016). To prevent the impact shifting issue, Liu and Bakshi (2018) have developed a methodology that combines TES and LCA. The resulting Techno-Ecological Synergy in Life Cycle Assessment framework, hereafter referred to as TES-LCA, can potentially capture trade-offs and synergies between ES, account for absolute environmental sustainability, and identify novel improvement opportunities through ecosystem restoration and ES trading, while incorporating life cycle impacts. To realize its wider applications, a computational structure needs to be developed with the ability to account for local and absolute environmental sustainability, and regional variation.

This paper develops a computational structure of TES-LCA by advancing the framework of process LCA to explicitly include ecological components and account for regional variation. To our knowledge, this paper is the first effort to explicitly capture the interactions between and within technological and ecological components in an integrated matrix form. This novel computational structure is able to provide insights about absolute environmental sustainability, while incorporating life cycle considerations. Part 1 of this paper series presents the basic computational structure and Part 2 presents how the basic structure can be adapted to account for regional and serviceshed information.

The rest of Part 1 of this paper series is organized as follows. Section 2 provides a brief introduction to the computational structure of conventional LCA and the TES-LCA methodology. Section 3 introduced the basic TES-LCA computational structure and its variation under different ES ownership scenarios. To facilitate the understanding of matrix-based no-

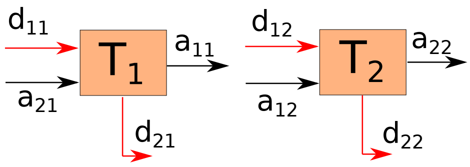

Figure 1: Process LCA

tations, the case study has been introduced in Section 2 and utilized throughout the paper to demonstrate each relevant methodology. The purpose is to provide a more direct comparison and a clearer depiction of similarities and differences between various methods. The advantages of explicitly including ecological systems in the analysis are conveyed through comparisons with the CF approach used by Bjørn et al. (2016) and the conventional LCA approach (Heijungs and Suh, 2002). The paper closes with a discussion of future research needs.

2. Background

2.1. Computational Structure of Conventional LCA

The process LCA model is adapted from Heijungs and Suh (2002) and written as:

$$\begin{aligned} \mathbf{Am} &= \mathbf{f} \\ \mathbf{Dm} &= \mathbf{r} \end{aligned} \tag{1}$$

In the context of Figure 1:

$\mathbf{A} = \begin{bmatrix} a_{11} & -a_{12} \\ -a_{21} & a_{22} \end{bmatrix}$ is the technology matrix, which shows the economic product flows between technological modules. If multiple functional processes are present, allocation is needed to split the inputs and outputs between co-products. Hence, $\mathbf{A}$ stays a square matrix, which can be inverted (Hanes et al., 2015).

$\mathbf{D} = \begin{bmatrix} -d_{11} & -d_{12} \\ d_{21} & d_{22} \end{bmatrix}$ is the environmental intervention matrix, which indicates resource use ("-") and emissions ("+") associated with a unit of product. In TES terminology, these flows are referred to as demands for ES, thus are denoted by d. Each item d_{kn} represents the demand for the k-th ES from the n-th activity. To be more specific, resource use demands corresponding provisioning ES while emissions demand for corresponding regulating ES.

$\mathbf{m} = \begin{bmatrix} m_1 \\ m_2 \end{bmatrix}$ is a vector of scaling factors or multipliers for technological modules, which is determined by solving balance equations.

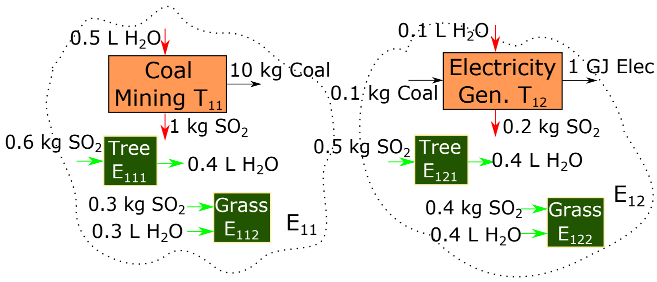

Figure 2: System Considered in the Toy Example

$\mathbf{f} = \begin{bmatrix} f_1 \\ f_2 \end{bmatrix}$ is a vector of final demands for technological modules, which is determined by societal needs. Once $\mathbf{f}$ is defined, scaling factors vector $\mathbf{m}$ can be calculated by rearranging eq (1) to: $\mathbf{m} = \mathbf{A}^{-1}\mathbf{f}$.

$\mathbf{r} = \begin{bmatrix} -r_1 \\ r_2 \end{bmatrix}$ is the inventory, which contains resource use ("-") and emissions ("+") information for the specified process network. Each item r_k can be interpreted as the demands for the k-th ES from the product's life cycle. Life cycle impact is calculated by:

$$\mathbf{g} = \mathbf{Q}\mathbf{r} = \mathbf{Q}\mathbf{D}\mathbf{A}^{-1}\mathbf{f} \tag{2}$$

in which $\mathbf{Q}$ is the characterization matrix. Each row in $\mathbf{Q}$ represents an impact category, e.g. acidification potential or water depletion potential; and each column contains the CF associated with each environmental invention in corresponding impact categories (i.e. rows). The vector $\mathbf{g}$ contains midpoint indicators for each impact category, which can be further aggregated to obtain endpoint indicators or even a single score.

Figure 2 shows the system for the case study. Two technological processes are considered, namely, coal mining and electricity generation. Both processes consume water and emit SO_2. Within the surrounding ecosystems, two ecological components are considered, namely tree and grass covers. It is assumed that both ecological components sequester SO_2; while trees recharge groundwater that can be utilized for grass growth. The ecological modules are interactive due to flow of H_2O between them. On the other hand, SO_2 is a non-interactive flow.

In the context of the conventional LCA, the focus is only on the technological systems T_{11} and T_{12}. The coal mining activity produces 10 kg of coal while using 0.5 L of H_2O and emitting 1 kg of SO_2; and the electricity generation activity consumes 0.1 kg coal to produce 1 GJ of electricity while using 0.1 L H_2O and emitting 0.2 kg SO_2. The acidification potential and respiratory effects are quantified for SO_2 flow, the CF of which are 1.0 kg SO_2 eq/kg and 0.061 kg $PM_{2.5}$ eq/kg, respectively (Bare, 2011). The water depletion potential is quantified for H_2O flow, the CF of which is 1 m^3/m^3 (Huijbregts et al., 2016). If a final demand of 0 kg of coal and 100 GJ of electricity is required by the outside consumers, by applying eq (1) and (2):

$$
\mathbf{m} = \mathbf{A}^{-1}\mathbf{f} = \begin{bmatrix} 10 & -0.1 \\ 0 & 1 \end{bmatrix}^{-1} \begin{bmatrix} 0 \\ 100 \end{bmatrix} = \begin{bmatrix} 1 \\ 100 \end{bmatrix}
$$

$$
\mathbf{Dm} = \begin{bmatrix} 1 & 0 \\ -0.5 & 0 \\ 0 & 0.2 \\ 0 & -0.1 \end{bmatrix} \begin{bmatrix} 1 \\ 100 \end{bmatrix} = \begin{bmatrix} 1 \\ -0.5 \\ 20 \\ -10 \end{bmatrix} = \mathbf{r} \tag{3}
$$

$$
\mathbf{g} = \mathbf{Qr} = \begin{bmatrix} 0 & 1 & 0 & 0 \\ 1 & 0 & 0 & 0 \\ 0.061 & 0 & 0 & 0 \\ 0 & 0 & 0 & 1 \\ 0 & 0 & 1 & 0 \\ 0 & 0 & 0.061 & 0 \end{bmatrix} \begin{bmatrix} 1 \\ -0.5 \\ 20 \\ -10 \end{bmatrix} = \begin{bmatrix} -0.5 \\ 1 \\ 0.061 \\ -10 \\ 20 \\ 1.22 \end{bmatrix}
$$

The vector $\mathbf{g}$ contains the environmental impacts of each activity separately, in terms of water depletion, acidification and respiratory effects. Note that in $\mathbf{D}$ of eq (3), the same environmental intervention flow, e.g. SO_2, for activities T_{11} and T_{12} need to be written in two separate rows, if the impacts from each activity in the life cycle are to be quantified separately. This representation is useful when the regionalized impacts need to be calculated (Yang and Heijungs, 2017). If the overall impacts from the entire life cycle are to be quantified, then the interventions from T_{11} and T_{12} need to be summed up before applying eq (2), which can be easily implemented by putting their intervention flows in the same row within $\mathbf{D}$ matrix, as follows:

$$
\mathbf{Dm} = \begin{bmatrix} 1 & 0.2 \\ -0.5 & -0.1 \end{bmatrix} \begin{bmatrix} 1 \\ 100 \end{bmatrix} = \begin{bmatrix} 21 \\ -10.5 \end{bmatrix} = \mathbf{r} \tag{4}
$$

$$
\mathbf{g} = \mathbf{Qr} = \begin{bmatrix} 0 & 1 \\ 1 & 0 \\ 0.061 & 0 \end{bmatrix} \begin{bmatrix} 21 \\ -10.5 \end{bmatrix} = \begin{bmatrix} -10.5 \\ 21 \\ 1.28 \end{bmatrix}
$$

The vector $\mathbf{g}$ contains the life cycle environmental impacts, in terms of water depletion, acidification and respiratory effects.

2.2. Techno-Ecological Synergy in Life Cycle Assessment (TES-LCA)

TES quantifies and compares demands and supplies of ES at multiple spatial scales. demand can be interpreted as resource use and emissions; while supply as the ecological capacity to provide the resources and absorb the emissions. In other words, demand is determined by technological processes while supply is determined by the considered ES and its serviceshed. TES sustainability metric v_k has been defined as (Bakshi et al., 2015):

$$
v_k = \frac{s_k - d_k}{d_k} \tag{5}
$$

Table 1: Procedures for Process LCA and TES-LCA (Liu and Bakshi, 2018)

Step	**LCA**	**TES-LCA**
Goal & scope definition	Define the scope for technological systems	Define the scope for both techno- and ecosystems
Life cycle inventory	Quantify resource use and emissions	Quantify ES demand and supply
Impact assessment	Calculate indicator scores for diffrenent impact categories	Measure the extent of sustainability
Improvement analysis	Discover hotspots and improve process efficiency	Encourage activities that do less bad and more good

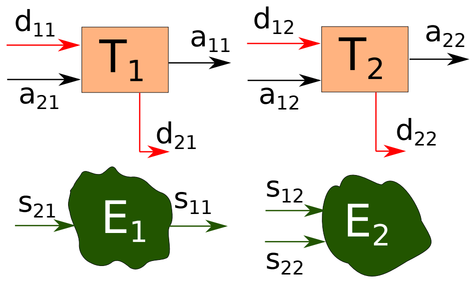

Figure 3: Technologies and Local Ecosystems

in which, d_k and s_k represent the demand and supply for the k-th ES, respectively. $v_k \geq 0$ implies local sustainability (or "island of sustainability") at the selected scale. These local sustainability metrics are not meant to indicate absolute environmental sustainability, but used to guide local improvements of sustainability performance. Since ES flows are largely confined to their servicesheds, the absolute environmental sustainability for the k-th ES is defined at the scale of serviceshed (indicated by *) as $v_k^* \geq 0$. This suggests that to claim absolute environmental sustainability, demand of the k-th ES should not exceed its supply at the serviceshed scale.

The procedure for conducting TES-LCA can be modified from that of process LCA, as summarized in Table 1. The process LCA analysis boundary is expanded to include the interacting ecosystems (Liu and Bakshi, 2018). Therefore, the computational structure of TES-LCA will also be based on that of process LCA, which was summarized in Section 2.1.

3. Computational Structure of TES-LCA

3.1. Basic Computational Structure

TES-LCA considers both technological and ecological systems in an integrated manner. In the context of Figure 3, all four modules need to be included in the analysis, where E_1 and E_2 are the surrounding ecosystems of technological activities T_1 and T_2, respectively.

Ecological systems can be treated as modules that input wastes discharged from technological activities and output ecosystem goods and services utilized by technological activities.

Therefore, they can be incorporated into the computational framework just like the technological modules. The computational structure of TES-LCA can be adapted from that of process LCA by including ecological modules and written as follows:

$$\left[\begin{array}{c|c} \mathbf{A} & \mathbf{C} \\ \hline \mathbf{D} & \mathbf{S} \end{array}\right] \left[\begin{array}{c} \mathbf{m} \\ \hline \mathbf{m}_e \end{array}\right] = \left[\begin{array}{c} \mathbf{f} \\ \hline \mathbf{f}_e \end{array}\right] \tag{6}$$

The notations used will be explained in the context of Figure 3. The two matrices, $\mathbf{A}$ and $\mathbf{D}$ are defined in the same manner as with process LCA in eq (1). To be more specific, $\mathbf{A}$ is the technology matrix, which shows the economic product flows between technological modules; while $\mathbf{D}$ is the environmental intervention matrix, which indicates resource use and emissions associated with a unit of product. Two additional submatrices are created to incorporate ecological modules explicitly:

$\mathbf{S} = \begin{bmatrix} s_{11} & -s_{12} \\ -s_{21} & -s_{22} \end{bmatrix}$ is the "ecosystem matrix", representing the flows between ecological modules E_1 and E_2. In TES terminology, the ES supply to technological activities (denoted by s) are generated from the interactions between ecological systems. For example, E_1 sequesters s_{21} units of a flow such as SO_2, while synergistically recharging s_{11} units of a resource such as groundwater, which can then be utilized by ecosystem E_2 at the rate of s_{12} for activities such as crop growth (Klimas et al., 2016). E_2 may also simultaneously sequester s_{22}. These interactions between ecological components can be captured in $\mathbf{S}$.

$\mathbf{C}$ is the "technological intervention" or "management" matrix, representing the economic product flows between technologies and ecosystems. For example, man-made ecosystems such as a lawn, may need to be managed with fertilizers and pesticides, to provide additional ES, which may not have direct economic values. These flows can be captured in $\mathbf{C}$.

$\mathbf{m}$ and $\mathbf{m}_e$ are vectors of scaling factors, while $\mathbf{f}$ and $\mathbf{f}_e$ are vectors of final demands from technologies and ecosystems, respectively. $\mathbf{f}$ can be interpreted as economic products from the network that satisfy human needs while $\mathbf{f}_e$ can be interpreted as environmental flows from the network that actually impact the ecosystems.

Eq (6) can be rewritten in a compact form as:

$$\mathbf{A}_{te}\mathbf{m}_{te} = \mathbf{f}_{te} \tag{7}$$

in which $\mathbf{A}_{te} = \left[\begin{array}{c|c} \mathbf{A} & \mathbf{C} \\ \hline \mathbf{D} & \mathbf{S} \end{array}\right]$, $\mathbf{m}_{te} = \left[\begin{array}{c} \mathbf{m} \\ \hline \mathbf{m}_e \end{array}\right]$ and $\mathbf{f}_{te} = \left[\begin{array}{c} \mathbf{f} \\ \hline \mathbf{f}_e \end{array}\right]$; the subscript te suggests that information about both technological and ecological systems is included. Therefore, $\mathbf{A}_{te}$ is referred to as "techno-ecological" matrix. It should not be surprising that eq (7) has a similar formulation as eq (1), since both technological and ecological modules can be incorporated into the production network in the same manner.

TES-LCA aims to create an integrated network involving both technologies and ecosystems, as shown in Figure 4. Ecosystems can absorb emissions and wastes from technologies, and in turn provide raw materials and services required by technologies. The curly arrows represent the surplus flows from the integrated network. The red ones indicate the actual impacts from the technologies after accounting for the capacities of ecosystems; while the

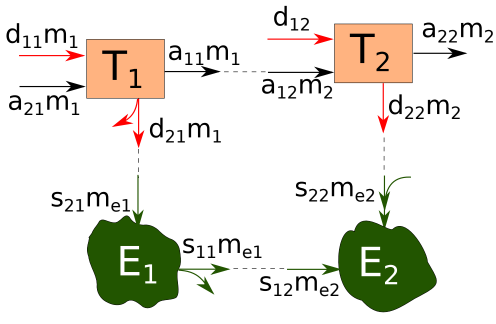

Figure 4: TES-LCA Network

green ones suggest the additional ES that can be utilized by other technologies not involved in the current network.

The formulation in eq (6) can be used to calculate v_k. If ecological modules are incorporated with their *maximum* current capacity into $\mathbf{A}_{te}$ and specify $\mathbf{f}_e$ to be $\mathbf{0}$, then a scaling factor of $m_{ek} = 1$ implies that the demand and supply of the k-th ES are equal. Also, when $m_{ek} > 1$, it indicates ecological overshoot. In fact, $m_{ek} = d_k/s_k$ since m_{ek} can be interpreted as the fraction of occupation of maximum current supply by the demand and therefore,

$$v_k = m_{ek}^{-1} - 1 \tag{8}$$

However, for some cases, since there exist complex interactions between ecological components, entries in $\mathbf{f}_e$ cannot achieve value 0 simultaneously. If $\mathbf{f}_e$ is forced to be $\mathbf{0}$, then the issue of negative scaling factors for ecosystems may appear. This means that ecosystems are operating in a reverse manner. For example, if a tree ecosystem module generally sequesters carbon, a negative scaling factor for this module would suggest that the tree will emit carbon instead, which does not make practical sense. Moreover, in the context of TES-LCA, determining the net interventions ($\mathbf{f}_e$) of technological systems after accounting for ecosystem's maximum mitigation capacity is of primary interest. Therefore, instead of considering $\mathbf{f}$ and $\mathbf{f}_e$ to be the known variables, $\mathbf{f}$ and $\mathbf{m}_e$ are considered to be the known variables. This is because after including flows from all available ecosystems in the selected region, $\mathbf{m}_e$ can be specified to be $\mathbf{1}$. Therefore, eq (6) is rearranged to be:

$$\begin{bmatrix} \mathbf{A} & \mathbf{0} \\ \mathbf{D} & -\mathbf{I} \end{bmatrix} \begin{bmatrix} \mathbf{m} \\ \mathbf{f}_e \end{bmatrix} = \begin{bmatrix} \mathbf{f} \\ \mathbf{0} \end{bmatrix} - \begin{bmatrix} \mathbf{C} \\ \mathbf{S} \end{bmatrix} \mathbf{m}_e \tag{9}$$

in which $\mathbf{I}$ is an identity matrix. With $\mathbf{m}_e = \mathbf{1}$, once the final demand $\mathbf{f}$ from the technological systems is specified, eq (9) can be used to calculate $\mathbf{m}$ and $\mathbf{f}_e$. The first matrix on the left-hand side can be adjusted to a square matrix, which is invertible. This is because the dimensions of the identity ($\mathbf{I}$) and zero ($\mathbf{0}$) matrices can be tuned accordingly, as long as $\mathbf{A}$ matrix is square (which can be achieved through proper allocation between co-products (Hanes et al., 2015)). Once $\mathbf{f}_e$ is known, v_k can be calculated based on f_{ek}:

$$v_k = -\frac{f_{ek}}{\mathbf{d}_{k\cdot} \cdot \mathbf{m}} \tag{10}$$

in which $\mathbf{d}_{k\cdot}$ is related to r_k through equation $\mathbf{d}_{k\cdot} \cdot \mathbf{m} = r_k$, where $\mathbf{m}$ is a vector of scaling factors of technological activities. Such a framework can be used to assess whether the ecological CC has been trespassed and if so, to which extent. The equivalence of eq (10) and eq (5) has been proved as follows:

$$\mathbf{Dm} + \mathbf{Sm}_e = \mathbf{f}_e \tag{11}$$

$\mathbf{d}_{k\cdot}$ and $\mathbf{s}_{k\cdot}$ are used to represent the k-th row in matrices $\mathbf{D}$ and $\mathbf{S}$, respectively, which can be interpreted as the demand and supply of the k-th ES by activities in the life cycle. Therefore, with regard to the k-th ES:

$$\mathbf{d}_{k\cdot} \cdot \mathbf{m} + \mathbf{s}_{k\cdot} \cdot \mathbf{m}_e = f_{ek} \tag{12}$$

For resource use flows, $\mathbf{d}_{k\cdot}$ has a "-" sign while $\mathbf{s}_{k\cdot}$ has a "+" sign. Since both s_k and d_k in eq (1) are positive values, therefore:

$$v_k = \frac{s_k - d_k}{d_k} = \frac{\mathbf{s}_{k\cdot} \cdot \mathbf{m}_e + \mathbf{d}_{k\cdot} \cdot \mathbf{m}}{-\,\mathbf{d}_{k\cdot} \cdot \mathbf{m}} = -\frac{f_{ek}}{\mathbf{d}_{k\cdot} \cdot \mathbf{m}} \tag{13}$$

For emission flows, $\mathbf{d}_{k\cdot}$ has a "+" sign while $\mathbf{s}_{k\cdot}$ has a "-" sign, therefore:

$$v_k = \frac{s_k - d_k}{d_k} = \frac{-\mathbf{s}_{k\cdot} \cdot \mathbf{m}_e - \mathbf{d}_{k\cdot} \cdot \mathbf{m}}{\mathbf{d}_{k\cdot} \cdot \mathbf{m}} = -\frac{f_{ek}}{\mathbf{d}_{k\cdot} \cdot \mathbf{m}} \tag{14}$$

The impacts resulting from the exceedance of ecological CC can be potentially quantified by combining TES-LCA with conventional LCIA (Posch et al., 2008), as follows:

$$\mathbf{g} = \mathbf{Qf}_e \tag{15}$$

This modification to eq (2) advances existing LCIA methods by making their results closer to the actual damages (Bare, 2011).

In the context of the case study (Figure 2), the ES supply from sites E_{11} and E_{12} are considered. Both sites have two ecological components, namely tree and grass covers. On E_{11}, the tree module sequesters 0.6 kg of SO_2 and recharges 0.4 L of H_2O; while the grass module sequesters 0.3 kg SO_2 and consumes 0.3 L of H_2O. On E_{12}, the tree module sequesters 0.5 kg of SO_2 and recharges 0.4 L of H_2O; while the grass module sequesters 0.4 kg SO_2 and consumes 0.4 L of H_2O. Assuming the final demands for coal and electricity are still 0 kg and 100 GJ, respectively, eq (9) can be applied to obtain the following:

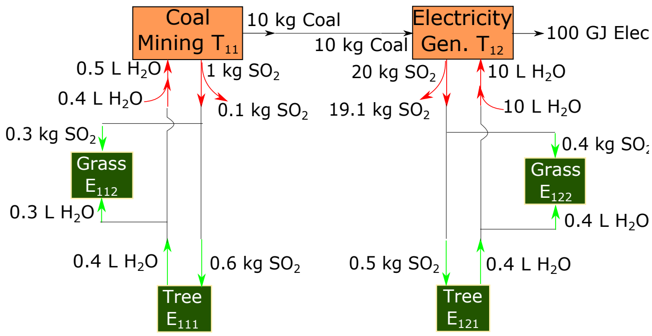

Figure 5: Network: Basic Computational Structure

$$
\begin{gathered}
\left[\begin{array}{c:c} \mathbf{A} & \mathbf{0} \\ \hdashline \mathbf{D} & -\mathbf{I} \end{array}\right]\left[\begin{array}{c} \mathbf{m} \\ \hdashline \mathbf{f}_e \end{array}\right] = \left[\begin{array}{cc:cccc} 10 & -0.1 & 0 & 0 & 0 & 0 \\ 0 & 1 & 0 & 0 & 0 & 0 \\ \hdashline 1 & 0 & -1 & 0 & 0 & 0 \\ -0.5 & 0 & 0 & -1 & 0 & 0 \\ 0 & 0.2 & 0 & 0 & -1 & 0 \\ 0 & -0.1 & 0 & 0 & 0 & -1 \end{array}\right]\left[\begin{array}{c} m_1 \\ m_2 \\ \hdashline f_{eS1} \\ f_{eH1} \\ f_{eS2} \\ f_{eH2} \end{array}\right] = \\
\left[\begin{array}{c} \mathbf{f} \\ \hdashline \mathbf{0} \end{array}\right] - \left[\begin{array}{c} \mathbf{C} \\ \hdashline \mathbf{S} \end{array}\right]\mathbf{m}_e = \left[\begin{array}{c} 0 \\ 100 \\ \hdashline 0 \\ 0 \\ 0 \\ 0 \end{array}\right] - \left[\begin{array}{cccc} 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 \\ \hdashline -0.6 & -0.3 & 0 & 0 \\ 0.4 & -0.3 & 0 & 0 \\ 0 & 0 & -0.5 & -0.4 \\ 0 & 0 & 0.4 & -0.4 \end{array}\right]\left[\begin{array}{c} 1 \\ 1 \\ 1 \\ 1 \end{array}\right]
\end{gathered} \tag{16}
$$

Eq (9) is applied instead of eq (6) to prevent the potential issue of negative scaling factors for ecosystems. In fact, if the water provisioning capacity of both tree modules are 0.2 L instead of 0.4 L, applying eq (6) directly can result in negative scaling factors for grass modules.

Figure 5 shows a substantiation of the TES-LCA network, in terms of the case study. From Figure 5, it can be inferred that both technological modules require additional water provisioning services to fully meet water demand and air quality regulation services to fully mitigate SO_2 emission, as indicated by the red curly arrows. These results are in line with those shown in Table 2. From eq (16), $\mathbf{f}_e$ can be calculated:

$$
\left[\begin{array}{c} \mathbf{m} \\ \hdashline \mathbf{f}_e \end{array}\right] = \left[\begin{array}{c} m_1 \\ m_2 \\ \hdashline f_{eS1} \\ f_{eH1} \\ f_{eS2} \\ f_{eH2} \end{array}\right] = \left[\begin{array}{c:c} \mathbf{A} & \mathbf{0} \\ \hdashline \mathbf{D} & -\mathbf{I} \end{array}\right]^{-1}\left\{\left[\begin{array}{c} \mathbf{f} \\ \hdashline \mathbf{0} \end{array}\right] - \left[\begin{array}{c} \mathbf{C} \\ \hdashline \mathbf{S} \end{array}\right]\mathbf{m}_e\right\} = \left[\begin{array}{c} 1 \\ 100 \\ \hdashline 0.1 \\ -0.4 \\ 19.1 \\ -10 \end{array}\right] \tag{17}
$$

Sustainability metrics can then be calculated using eq (10) for each activity in the life cycle and the life cycle itself. The impacts of excessive flows that are not mitigated by ecosystems can then be quantified following eq (15) for each activity separately. And for the same reason, the same emission flow (e.g. SO_2) from the two activities are treated as different environmental interventions (e.g. SO_2 emissions from activities T_{11} and T_{12}). By applying eq (15):

$$\mathbf{g} = \mathbf{Q}\mathbf{f}_e = \begin{bmatrix} 0 & 1 & 0 & 0 \\ 1 & 0 & 0 & 0 \\ 0.061 & 0 & 0 & 0 \\ 0 & 0 & 0 & 1 \\ 0 & 0 & 1 & 0 \\ 0 & 0 & 0.061 & 0 \end{bmatrix} \begin{bmatrix} 0.1 \\ -0.4 \\ 19.1 \\ -10.0 \end{bmatrix} = \begin{bmatrix} -0.4 \\ 0.1 \\ 0.0061 \\ -10.0 \\ 19.1 \\ 1.165 \end{bmatrix} \tag{18}$$

If the impacts from the life cycle are to be quantified, then the actual interventions from T_{11} and T_{12} need to be summed up before applying eq (15), which can be easily implemented by putting their intervention flows in the same row within $\mathbf{D}$ matrix, as shown in eq (3) and (4).

With the CF approach, the land area needed to provide the services is calculated using the CC and demand information. Following assumptions are made: 1) all emissions within a serviceshed do not flow across serviceshed boundaries (fate factor equals to 1); 2) the two activities are nested within the same air quality regulation and water provisioning servicesheds, the capacities of which are assumed to be 9 kg SO_2/ha and 4 L H_2O/ha, respectively; 3) each hectare of land nested within the serviceshed has the same CC of that particular ES. Then starting from eq (3) in Bjørn et al. Bjørn et al. (2016), CF can be calculated for each environmental intervention flow as follows:

$$CF_k = \frac{1}{CC_k} \tag{19}$$

Note that up to the calculation of inventory $\mathbf{r}$, the CF approach is exactly the same as the conventional one, as shown in eq (3). Then by applying eq (2) with the updated CF incorporating CC information:

$$\mathbf{g} = \mathbf{Q}\mathbf{r} = \begin{bmatrix} \frac{1}{9} & 0 & 0 & 0 \\ 0 & -\frac{1}{4} & 0 & 0 \\ 0 & 0 & \frac{1}{9} & 0 \\ 0 & 0 & 0 & -\frac{1}{4} \end{bmatrix} \begin{bmatrix} 1 \\ -0.5 \\ 20 \\ -10 \end{bmatrix} = \begin{bmatrix} 0.111 \\ 0.125 \\ 2.22 \\ 2.5 \end{bmatrix} \tag{20}$$

The impacts of the CF approach are calculated in terms of land area that is needed to mitigate the interventions. The available land area is assumed to be 0.1 ha for each activity. Comparing the area needed to that available, in this case, $[0.1, 0.1, 0.1, 0.1]^T$, with eq (10), sustainability metrics can be obtained.

For conventional LCA approach, the inventory is built for technological modules through eq (3). The above results are summarized in Table 2. No supply and absolute environmental sustainability metric information is available from conventional LCA approach and the corresponding blanks are thus marked with "n/a".

Table 2: Comparison of Results Obtained from Different Methods

Flows	**Methods**	**Demand**		**Supply**		**Unit**	**Metric v_k**		**Metric v_k^***	
		T_{11}	T_{12}	T_{11}	T_{12}		T_{11}	T_{12}	T_{11}	T_{12}
SO_2	Conventional	1	20	n/a	n/a	kg	n/a	n/a	n/a	n/a
	CF	0.111	2.22	0.1	0.1	ha	-0.1	-0.96	n/a	n/a
	TES-LCA	1	20	0.9	0.9	kg	-0.1	-0.96	2.9	-0.51
H_2O	Conventional	0.5	10	n/a	n/a	L	n/a	n/a	n/a	n/a
	CF	0.125	2.5	0.1	0.1	ha	-0.2	-0.96	n/a	n/a
	TES-LCA	0.5	10	0.1	0	L	-0.8	-1	0.2	-0.9

It can be inferred from Table 2 that both TES-LCA and CF approaches attempt to modify conventional LCA approach to incorporate ecological CC information, but at different points in the impact pathway. TES-LCA method compiles life cycle inventories for both technological and ecological systems, enabling the direct incorporation of the role of ES and their interactions. CF approach compiles inventories only for technological system. Ecological CC is only included in an implicit manner by acting as the normalization factor in CF. Also note that these two methods measure demand and supply in different units. The CF method calculates the land area that supplies the demanded ES, which resembles ecological footprint method (Bjørn et al., 2016); while TES-LCA measures the demands and supplies in physical units (Liu and Bakshi, 2018). If the metrics calculated by applying eq (5) for these two methods are compared further, it can be inferred that for non-interactive flows such as SO_2, the sustainability metrics calculated with both methods are the same. However, the sustainability metrics are different for interactive flows, such as H_2O. It can be observed that for interactive flows, the metrics calculated using the CF approach in general have larger values. The reason is that their way of including CC cannot account for the current interventions that result from the interactions between ecological components. This depicts the advantage of capturing the interactions within and between techno-ecological systems in the matrix form. Also note that in this case, both approaches measure environmental sustainability only at a local scale. But the decisions on absolute sustainability need to be made at the serviceshed scale, which TES-LCA includes.

3.2. Accounting for Allocation of ES between Multiple Users

ES supply from a serviceshed needs to be partitioned between multiple users nested within it. Two potential solution strategies are available, namely proportional allocation and avoided allocation (Bakshi et al., 2015). The avoided allocation approach provides information on absolute environmental sustainability by considering total demand and supply of ES within the serviceshed.

Proportional allocation approach allocates ES supply in the serviceshed to multiple users according to selected properties, such as population, area, or demand. The allocated ES can be interpreted as the "right to use". Two approaches for determining this use right include the private and public ownership of ES (Liu and Bakshi, 2018). Private ownership implies the situation where the land owners can claim the ownership of ES produced from their privately owned land and a fraction of ES allocated from the publicly owned land in the serviceshed. Public ownership, on the other hand, implies that ES produced within the serviceshed belongs to every activity that demands this service, regardless of the ownership of land from where it is produced (Liu and Bakshi, 2018).

The proposed computational structure can be modified to account for these cases. E_1, ..., E_n are assumed to represent local ecosystems privately owned by activities T_1,..., T_n. The ES supply available from these sites are s_{k1},..., s_{kn} in terms of the k-th ES. $E_{kn\beta}$ represents the publicly owned ecosystem (denoted by β) in the serviceshed of the k-th ES where the n-th activity is nested within, whose service supply, $s_{kn\beta}$, should be allocated to all activities in that serviceshed. This allocation process can be envisioned as splitting $s_{kn\beta}$ between multiple users based on the selected quantities. Therefore, a partitioned ES supply flow can be created by combining s_{kn} and the allocated supply from $E_{kn\beta}$ due to the fact that they are dealing with common environmental flows by providing the same ES indexed by k. Mathematically, the allocation under private ownership of k-th ES can be generalized as:

$$\widetilde{\mathbf{S}} = \mathbf{S} + \mathbf{S}_\beta \circ \mathbf{W} \tag{21}$$

in which $\widetilde{\mathbf{S}} = \begin{bmatrix} \widetilde{s}_{11} & \widetilde{s}_{12} & \dots & \widetilde{s}_{1n} \\ \widetilde{s}_{21} & \widetilde{s}_{22} & \dots & \widetilde{s}_{2n} \\ \vdots & \vdots & \ddots & \vdots \\ \widetilde{s}_{k1} & \widetilde{s}_{k2} & \dots & \widetilde{s}_{kn} \end{bmatrix}$, $\mathbf{S} = \begin{bmatrix} s_{11} & s_{12} & \dots & s_{1n} \\ s_{21} & s_{22} & \dots & s_{2n} \\ \vdots & \vdots & \ddots & \vdots \\ s_{k1} & s_{k2} & \dots & s_{kn} \end{bmatrix}$, $\mathbf{S}_\beta = \begin{bmatrix} s_{11\beta} & s_{12\beta} & \dots & s_{1n\beta} \\ s_{21\beta} & s_{22\beta} & \dots & s_{2n\beta} \\ \vdots & \vdots & \ddots & \vdots \\ s_{k1\beta} & s_{k2\beta} & \dots & s_{kn\beta} \end{bmatrix}$, $\mathbf{W} = \begin{bmatrix} w_{11} & w_{12} & \dots & w_{1n} \\ w_{21} & w_{22} & \dots & w_{2n} \\ \vdots & \vdots & \ddots & \vdots \\ w_{k1} & w_{k2} & \dots & w_{kn} \end{bmatrix}$. The "$\circ$" suggests for element-wise matrix multiplication. If the special case where these activities are nested within the same serviceshed of the k-th ES is considered, then $s_{k1\beta} = s_{k2\beta} = \ldots = s_{kn\beta}$. With the additional assumption that they are the only consumers of the k-th ES in that serviceshed, $w_{kn} = \dfrac{\pi_n}{\sum \pi_n}$ calculates weighting factors used when allocating ES between technological activities based on the selected property π. $\widetilde{s}_{kn}$ can be interpreted as the total amount of the k-th ES that can be claimed by activity n. If there exist multiple ecological modules in the local ecosystem that supply the same ES, the allocated supply should be further partitioned between these modules, where eq (21) can also be applied.

Under the public ownership scenario, the serviceshed ecosystem is treated as a whole. Thus, the total supply of the k-th ES from the serviceshed needs to be known. A partitioned ES supply flow can be created by allocating the serviceshed supply between users. The allocated value should be directly incorporated as a module in $\widetilde{\mathbf{S}}$, since the ecological components in the locality and thus their interactions are not the primary focus of the public ownership scenario. Mathematically, the allocation under public ownership of k-th ES can be generalized as:

$$\widetilde{\mathbf{S}} = \mathbf{S}^* \circ \mathbf{W} \tag{22}$$

in which $\mathbf{S}^* = \begin{bmatrix} s^*_{11} & s^*_{12} & \cdots & s^*_{1n} \\ s^*_{21} & s^*_{22} & \cdots & s^*_{2n} \\ \vdots & \vdots & \ddots & \vdots \\ s^*_{k1} & s^*_{k2} & \cdots & s^*_{kn} \end{bmatrix}$, whose elements s^*_{kn} stands for the total supply of the k-th ES from the serviceshed where the n-th activity is nested. $\widetilde{\mathbf{S}}$ and $\mathbf{W}$ are defined as with eq (21).

Eq (10) can then be applied to calculate sustainability metrics for both scenarios. These sustainability metrics are calculated at the scale of the serviceshed and define whether the activity can claim its use of ES to be sustainable in the context of servicesheds.

In the context of the case study (Figure 2), it is assumed that T_{11} is allocated 3 kg SO_2 and 0.5 L H_2O while T_{12} is allocated 9 kg SO_2 and 1 L H_2O from the corresponding servicesheds for air quality regulation and water provisioning. In practice, this allocation can be performed according to any selected property by applying eq (21). For both sites E_{11} and E_{12}, water provisioning service is only provided by trees. Therefore, the allocated water supply is combined with the tree modules, E_{111} and E_{121}, respectively. Air quality regulation services are provided by both tree and grass cover. Therefore, in this demonstration, the allocated air quality regulation service supply is further partitioned between tree and grass modules based on their current mediating capacity. The partitioned ecosystem matrix $\widetilde{\mathbf{S}}$ can be obtained following eq (21):

$$\begin{aligned} \widetilde{\mathbf{S}} &= \mathbf{S} + \mathbf{S}_\beta \circ \mathbf{W} \\ &= \begin{bmatrix} -0.6 & -0.3 & 0 & 0 \\ 0.4 & -0.3 & 0 & 0 \\ 0 & 0 & -0.5 & -0.4 \\ 0 & 0 & 0.4 & -0.4 \end{bmatrix} + \begin{bmatrix} -3 & -3 & 0 & 0 \\ 0.5 & 0.5 & 0 & 0 \\ 0 & 0 & -9 & -9 \\ 0 & 0 & 1 & 1 \end{bmatrix} \circ \begin{bmatrix} 0.67 & 0.33 & 0 & 0 \\ 1 & 0 & 0 & 0 \\ 0 & 0 & 0.56 & 0.44 \\ 0 & 0 & 1 & 0 \end{bmatrix} \\ &= \begin{bmatrix} -2.6 & -1.3 & 0 & 0 \\ 0.9 & -0.3 & 0 & 0 \\ 0 & 0 & -5.5 & -4.4 \\ 0 & 0 & 1.4 & -0.4 \end{bmatrix} \end{aligned} \tag{23}$$

Eq (10) can then be applied to calculate sustainability metrics at the serviceshed scale, which are provided in Table 2 in columns Metric v^*_k. It can be inferred that due to the ownership of the additional ES supply allocated from the serviceshed, activity T_{11} may claim its use of ES to be sustainable in terms of both air quality regulation and water provisioning services, in the context of servicesheds. Since the CF approach does not account for the serviceshed concept, the corresponding blanks are thus marked with “n/a”.

Figure 6 shows a substantiation of the TES-LCA network, in terms of the case study, involving the allocation of ES between multiple users in the corresponding servicesheds under the private ownership scenario. Note that the serviceshed for water provisioning ES does

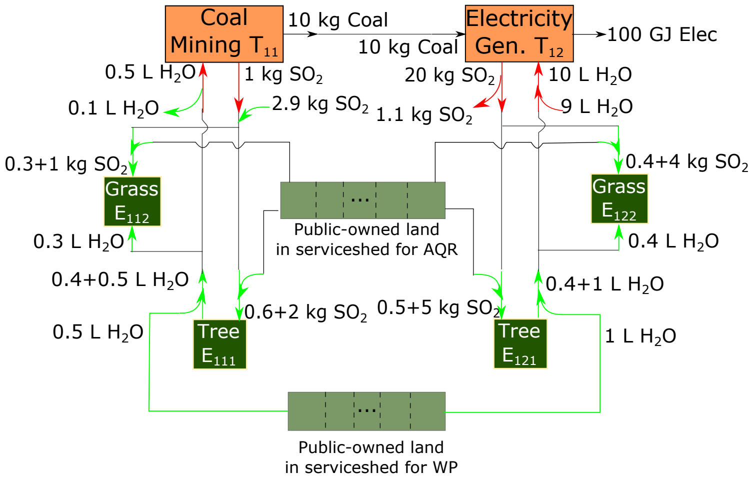

Figure 6: Network: Accounting for Allocation of ES between Multiple Users (AQR: air quality regulation; WP: water provision)

not coincide with that for air quality regulation ES. This is the reason why the public-owned lands in the corresponding servicesheds are represented separately in Figure 6. Based on the partitioned ES supply flows, it can be inferred that T_{11} can claim its activity to be sustainable and provide surplus ES. On the other hand, T_{12} still requires additional ES inputs, thus cannot claim sustainability.

4. Conclusions and Discussion

The TES-LCA computational framework developed in this work explicitly captures the interactions between technological and ecological components in an integrated matrix form while accounting for the product's life cycle and absolute environmental sustainability. The framework can help in meeting the requirements for environmental sustainability assessment of preventing unintended harm and respecting nature's limits. Since the role of ES is included directly in TES-LCA, the actual impacts on ecosystems that cannot be quantified by the conventional LCA approaches can be obtained, together with the potential identification of novel ecological solutions. Nonetheless, several challenges need to be addressed to implement the proposed TES-LCA computational structure, mainly on the availability of data about ES and their servicesheds.

Information about ES supply can be obtained from detailed ecological models and remote sensing. Such data need to be made available in future versions of LCI databases. In addition, accounting for absolute environmental sustainability requires the consideration of an exhaustive array of ES, which necessitates the development of consistent methods to quantify their demand and supply. ES classification schemes, such as CICES, can be applied to fulfill this need (Haines-Young and Potschin, 2013).

Regarding the serviceshed delineation issue, accepted "standard" needs to be developed for each and every ES, based on advanced understanding on ES delivery. More rigorous methods, such as setting cutting-off criteria, may also be applied (Bjørn et al., 2016).

Moreover, a systematic methodology is needed to identify all activities that demand an ES in the serviceshed to avoid biased use right in the allocation step. Ecosystems require certain amount of ES for proper functioning, which prevents this portion from being available for human use. Take water provisioning service as an example, it is suggested that for maintaining a fair condition, 20% to 50% of the mean annual river flow is required by ecosystems (Smakhtin et al., 2004). Likewise, the amount of service that should be kept aside needs to be quantified for each ES.

Once available, such data can be included in future versions of life cycle inventory databases. To further gain popularity and facilitate usage, a software implementing this computational structure needs to be developed, as with conventional LCA.

Furthermore, the dynamic aspects are ignored in the current study. In fact, by treating ecological components as modules, their annual average performances are intuitively used. However, ecological CC may change on an hourly base, which suggests that ecosystems need to be analyzed under a much higher time resolution.

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Appendix A.

Table A.3: Description of notations used

Symbols	Description and Units
$\mathbf{A}$	Technological matrix
$\mathbf{C}$	Management matrix
$\mathbf{D}$	Environmental intervention matrix
$\mathbf{S}$	Ecosystem matrix
$\mathbf{Q}$	Characterization factor matrix
$\mathbf{f}$	Final demand vector, technological systems
$\mathbf{f}_e$	Final demand vector, ecological systems
$\mathbf{m}$	scaling factor vector for technological modules
$\mathbf{m}_e$	scaling factor vector for ecological modules
$\mathbf{r}$	Life cycle inventory vector
$\mathbf{g}$	Midpoint indicator, environmental impacts vector
d_k	Demand for k-th ecosystem service (physical units)
s_k	Supply for k-th ecosystem service (physical units)
v_k	Sustainability metric (dimensionless)
k	Ecosystem service numbering
n	Activity numbering
T	Technological modules
E	Ecological modules

CF	Characterization factor
ES	Ecosystem service
CC	Carrying capacity
β	Public owned ecosystem

References

Bakshi, B. R., Ziv, G., Lepech, M. D., 2015. Techno-ecological synergy: A framework for sustainable engineering. Environ Sci Technol 49 (3), 1752–1760.

Bare, J., 2011. Traci 2.0: the tool for the reduction and assessment of chemical and other environmental impacts 2.0. Clean Technol Envir 13 (5), 687–696.

Bjørn, A., 2015. Better, but good enough? indicators for absolute environmental sustainability in a life cycle perspective. Ph.D. thesis, DTU Management Engineering.

Bjørn, A., Diamond, M., Owsianiak, M., Verzat, B., Hauschild, M. Z., 2015. Strengthening the link between life cycle assessment and indicators for absolute sustainability to support development within planetary boundaries. Environ Sci Technol 49 (11), 6370–6371.

Bjørn, A., Hauschild, M. Z., 2015. Introducing carrying capacity-based normalisation in lca: framework and development of references at midpoint level. Int J Life Cycle Ass 20 (7), 1005–1018.

Bjørn, A., Margni, M., Roy, P.-O., Bulle, C., Hauschild, M. Z., 2016. A proposal to measure absolute environmental sustainability in life cycle assessment. Ecol Indic 63, 1–13.

Doka, G., 2015. Combining life cycle inventory results with planetary boundaries: the planetary boundary allowance impact assessment method PBA'05. Doka Life Cycle Assessment, Zürich, `http://www.doka.ch/Doka2ndDraftPBAMethod.pdf`. Accessed 12/21/2016.

Ghosh, T., Bakshi, B. R., 2017. Process to planet approach to sustainable process design: Multiple objectives and byproducts. Theoretical Foundations of Chemical Engineering 51 (6), 936–948.

Gopalakrishnan, V., Bakshi, B. R., Ziv, G., 2016. Assessing the capacity of local ecosystems to meet industrial demand for ecosystem services. AIChE J 62 (9), 3319–3333.

Haines-Young, R., Potschin, M., 2013. Revised report prepared following consultation on cices version 4, august-december 2012. eea framework contract no eea/iea/09/003.

Hanes, R. J., Bakshi, B. R., 2015. Process to planet: A multiscale modeling framework toward sustainable engineering. AIChE J 61 (10), 3332–3352.

Hanes, R. J., Cruze, N. B., Goel, P. K., Bakshi, B. R., 2015. Allocation games: addressing the ill-posed nature of allocation in life-cycle inventories. Environ Sci Technol 49 (13), 7996–8003.

Heijungs, R., Suh, S., 2002. The computational structure of life cycle assessment. Kluwer Academic Publishers.

Huijbregts, M., Steinmann, Z., Elshout, P., Stam, G., Verones, F., Vieira, M., Van Zelm, R., 2016. ReCiPe2016. A harmonized life cycle impact assessment method at midpoint and endpoint level. Report I: characterization. RIVM Report 2016–0104. National Institute for Human Health and the Environment, Bilthoven, `http://www.rivm.nl/en/Documents_and_publications/Scientific/Reports/2016/december/ReCiPe_2016_A_harmonized_life_cycle_impact_assessment_method_at_midpoint_and_endpoint_level_Report_I_Characterization`. Accessed 01/05/2017.

Isard, W., 1972. Ecologic-economic analysis for regional development. Free Press.

Klimas, C., Williams, A., Hoff, M., Lawrence, B., Thompson, J., Montgomery, J., 2016. Valuing ecosystem services and disservices across heterogeneous green spaces. Sustainability-Basel 8 (9), 853.

Koellner, T., de Baan, L., Beck, T., Brandão, M., Civit, B., Margni, M., i Canals, L. M., Saad, R., de Souza, D. M., Müller-Wenk, R., 2013. Unep-setac guideline on global land use impact assessment on biodiversity and ecosystem services in lca. Int J Life Cycle Ass 18 (6), 1188–1202.

Liu, X., Bakshi, B. R., 2018. Ecosystem services in life cycle assessment while encouraging techno-ecological synergies. Journal of Industrial Ecology. DOI: 10.1111/jiec.12755.

Liu, X., Singh, S., Gibbemeyer, E. L., Tam, B. E., Urban, R. A., Bakshi, B. R., 2018. The carbon-nitrogen nexus of transportation fuels. Journal of Cleaner Production 180, 790–803.

Martinez-Hernandez, E., Leung Pah Hang, M. Y., Leach, M., Yang, A., 2016. A framework for modeling local production systems with techno-ecological interactions. Journal of Industrial Ecology.

Miller, R. E., Blair, P. D., 2009. Input-output analysis: foundations and extensions. Cambridge University Press.

Othoniel, B., Rugani, B., Heijungs, R., Benetto, E., Withagen, C., 2016. Assessment of life cycle impacts on ecosystem services: Promise, problems, and prospects. Environ Sci Technol 50 (3), 1077–1092.

Posch, M., Seppälä, J., Hettelingh, J.-P., Johansson, M., Margni, M., Jolliet, O., 2008. The role of atmospheric dispersion models and ecosystem sensitivity in the determination of characterisation factors for acidifying and eutrophying emissions in lcia. Int J Life Cycle Ass 13 (6), 477–486.

Rebitzer, G., Ekvall, T., Frischknecht, R., Hunkeler, D., Norris, G., Rydberg, T., Schmidt, W.-P., Suh, S., Weidema, B. P., Pennington, D. W., 2004. Life cycle assessment: Part 1: Framework, goal and scope definition, inventory analysis, and applications. Environ Int 30 (5), 701–720.

Sandin, G., Peters, G. M., Svanström, M., 2015. Using the planetary boundaries framework for setting impact-reduction targets in lca contexts. Int J Life Cycle Ass 20 (12), 1684–1700.

Smakhtin, V., Revenga, C., Döll, P., Tharme, R., Nackoney, J., Kura, Y., 2004. Taking into account environmental water requirements in global-scale water resources assessments. Comprehensive Assessment of Water Management in Agriculture.

Steffen, W., Richardson, K., Rockström, J., Cornell, S. E., Fetzer, I., Bennett, E. M., Biggs, R., Carpenter, S. R., de Vries, W., de Wit, C. A., Folke, C., Gerten, D., Heinke, J., Mace, G. M., Persson, L. M., Ramanathan, V., Reyers, B., Sörlin, S., 2015. Planetary boundaries: Guiding human development on a changing planet. Science 347 (6223), 1–10.

Suh, S., 2004. Functions, commodities and environmental impacts in an ecological–economic model. Ecol Econ 48 (4), 451–467.

Tallis, H., Wolny, S., Lozano, J. S., Benitez, S., Saenz, S., Ramos, A., 2013. "Servicesheds" Enable Mitigation of Development Impacts on Ecosystem Services. Natural Capital Project, `https://www.naturalcapitalproject.org/pubs/Servicesheds_Working_Paper_2013.pdf`. Accessed 12/21/2016.

Urban, R. A., Bakshi, B. R., 2009. 1, 3-propanediol from fossils versus biomass: a life cycle evaluation of emissions and ecological resources. Ind Eng Chem Res 48 (17), 8068–8082.

Urban, R. A., Bakshi, B. R., 2013. Techno-ecological synergy as a path toward sustainability of a north american residential system. Environ Sci Technol 47 (4), 1985–1993.

Yang, Y., Heijungs, R., 2017. A generalized computational structure for regional life-cycle assessment. Int J Life Cycle Ass 22, 213–221.

Zhang, Y., Singh, S., Bakshi, B. R., 2010. Accounting for ecosystem services in life cycle assessment, part i: a critical review. Environ Sci Technol 44 (7), 2232–2242.