Environmental Economics

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1. Introduction

Environmental economics studies the impact of economic activity on the environment. This concerns both (1) how scarce natural resources are used (and potentially over-used) to facilitate economic activity, and (2) how negative 'externalities', such as pollution, can sub-optimally affect environmental quality unless they are somehow 'internalised' in the decision-making of producers and consumers. In addressing these problems, environmental economics tackles a number of key issues. These range from the valuation of natural resources, and more generally 'putting a price on' inputs from, and outputs to, the environment, to the development and analysis of the operation of environmental policy instruments such as standards, taxes and markets for emissions rights. A wide array of techniques and modelling approaches are used in these studies. Increasingly environmental economists and policy analysts have employed multi-sector general equilibrium models of which Input Output (IO) could be regarded as a limiting, but transparent, rigorous and familiar, example.

The nature and magnitude of economic-environmental interaction varies dramatically across different types of economic activity. Specifically, individual production industries and final consumption sectors differ significantly in the intensity and pattern of resource use and pollution generation. This motivates the application of multi-industry general equilibrium

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models to environmental problems, where the first requirement is that the level of industrial disaggregation should appropriately reflect the different resource-use and pollution characteristics. This allows investigation of the relationship between both the size and composition of economic activity and its environmental impact.

A second crucial factor is that the interdependence among industries will be important in determining the environmental impact of aggregate economic activity. For example, modelling and accounting for the pattern of energy use has been a major focus of economic-environmental analyses since the OPEC oil crisis in the 1970s. Simple measures of resource extraction and pollution generation can be made for the economy as a whole, or for any one industry, but this does not help in terms of targeting and tailoring policy action to reduce environmental damage. This is because from the point of extraction of an energy resource, such as oil, there is a complex chain of input and output use by, and between, different industries within the economy. Therefore any industry's output level, and therefore its resource requirements and pollution production, are dependent on demand in other industries in the economy. So the type of model that is required to address key problems arising from economic-environmental interaction must not only be multi-industrial but also be able to capture interdependencies and feedback effects between all producers and (intermediate and final) consumers in the economy. In this respect, the strengths of IO are its ease and transparency of implementation and its potentially very high levels of industrial disaggregation. This disaggregation allows key interactions and interdependencies to be easily traced and identified.

This chapter proceeds as follows. Section 2 details the way in which environmental elements have been incorporated into standard demand-driven IO analysis. Section 3 focusses on pollution generation in production and consumption in single and multi-regional/national IO modelling. This includes attribution using the production and consumption accounting principles and the incorporation of cleansing sectors. Section 4 gives further developments of the IO method to investigate water use and waste generation, and its use in the identification of intermediate use of energy in efficiency rebound studies. Section 5 discusses practical problems of implementing these models and their policy relevance.

2. Extension of the conventional demand driven quantity IO model to the economy-environment nexus

The most basic demand-driven IO environmental models retain at their core Leontief's initial (1936) theoretical focus on final demands (**y**) driving output (**x**) – and, crucially, any related variables - through the familiar matrix identity:

$$[1]\ \mathbf{x} = [\mathbf{I} - \mathbf{A}]^{-1}\mathbf{y} = \mathbf{L}\mathbf{y}$$

where bold font upper case variables denote matrices and bold font lower case variables vectors. In equation (1), **A** is a matrix of input coefficients where the individual elements, a_{ij}, show the value of input from industry i per monetary unit of industrial output j. The Leontief inverse matrix of output multipliers, $\mathbf{L} = [\mathbf{I} - \mathbf{A}]^{-1}$, then identifies the required input from each industry i to meet one unit of final demand for the output of each sector j. This matrix forms the transmission mechanism in the simple Leontief demand-driven general equilibrium model. In the conventional account, such a model embodies fixed technical (and where relevant consumption) coefficients and a perfectly elastic supply of non-produced inputs.

Related physical variables, such as the level of employment or, in the current context, tonnes of a given pollutant, cubic metres of water etc. are introduced through industry output coefficients. These link the output of each sector i in the Leontief inverse to a physical quantity that is directly required for, or generated in, the production of that output. In this way, the theoretical premise of a demand-driven economic system is extended to variables other than output. The central IO accounting identity can then be applied so that the total quantities of physical variables used or generated in the economy can be determined by the level and sectoral composition of final demands. This is a natural and intuitive development given that human consumption decisions are commonly regarded as the ultimate source of most environmental problems. For this reason, environmental IO models tend to focus on application of what is commonly referred to as 'Type I' multiplier analysis where household consumption is treated as an element of the exogenous final demands that are driving the system.

It is important to note that IO analysts have made developments that more directly capture key causal relationships than is possible with the simple output coefficient extension. For example, Lange (1998), Gale (1995), Hyami *et al* (1997) Lenzen (1998) and Weir (1998) all relate the generation of pollution directly to (mainly energy) input use rather than to industry outputs. In a conventional IO setting the distinction between input- and output-pollution coefficients is irrelevant. This is because with the conventional assumption of universal fixed (Leontief) production technologies there is a constant proportional relationship between inputs and outputs in all sectors. With the input mix fixed, output-pollution coefficients will be equivalent to input-pollution coefficients in terms of impact.Nevertheless there are situations where linking pollution to resource inputs in an IO analysis allows more careful tracking and reporting of causal relationships. An example is Wier's, (1998) structural decomposition analysis using IO data for several accounting periods to identify the evolving relationship between energy use and pollution generation. A second is Lenzen's (1998) focus on primary energy use and greenhouse gas (GHG) emissions to distinguish key sources of particular types of pollution. However, as will be discussed later in this chapter, where there is any change in these relationships, particularly where this involves any change in prices and/or supply behaviour, the applicability of the conventional demand-driven environmental IO model is more limited.

There are two general advantages of IO. The first is that the the typically relatively high levels of industrial disaggregation facilitates analysis of interdependencies and interactions within the economy and between the economy and the environment. Even so, this disaggregation is often argued as being insufficient, as is discussed later. The second is the clear causal ordering of both direct and indirect resource use and pollution generation that arises from the theoretical premise and accounting identities first proposed by Leontief (1936).

It is the latter in particular that is the basis of the huge growth in application of the demand-driven environmental IO model over the last couple of decades, with the main direction being towards the use of IO techniques to calculate environmental 'footprints'. This in turn has motivated the extension of the demand-driven environmental IO framework in an inter-regional/inter-country context: to account for resource-use and pollution embodied in trade so that it may be attributed to the country where it ultimately supports final demand. This is particularly important in the context of GHGs that contribute to climate change and is one of the extensions considered in the next section.

3. Pollution generation and the demand-driven environmental IO model

In this section we formalise and extend discussion of the most straightforward and commonly used environmental IO accounting and modelling framework. This focuses on pollution generation and is based on the conventional demand-driven IO system introduced in Section 2. We begin with the simplest single region/country framework and place this in the context of the production and consumption accounting principles (Munksgaard and Pedersen, 2001). These underlie the territorial emissions vs. carbon footprint debate that is the most common context in which environmental IO techniques have been applied in recent years. We then go on to consider two key extensions. The first is the most commonly applied inter-regional method that facilitates full consumption or carbon footprint accounting. The second is the the rarely used approach proposed by Leontief (1970) to consider the implications of internalising the resource costs of emissions generation. Finally we detail the early attempts to simultaneously model the economy and the environment through utilising the material balance principle.

3.1. The single region/country framework

Let us consider k = 1, …, K different pollutants directly generated in production and consumption activity within a given economy.[1] The vector of total direct pollutant emissions generated within a region's/country's borders as a result of economic activity, $\mathbf{p^T}$, can be separated into those determined in production, $\mathbf{p^X}$, and consumption, $\mathbf{p^C}$:

[2] $\mathbf{p^T} = \mathbf{p^X} + \mathbf{p^C}$

where $\mathbf{p^X}$, $\mathbf{p^C}$ and $\mathbf{p^T}$ are K×1 vectors, whose elements p_k^X, p_k^C and p_k^T are the total physical amount of pollutant *k* directly generated in the economy through production, consumption and in total respectively.

In the single region/country environmental IO framework, these pollutant vectors can be shown to be determined by exogenous final demand as follows:

[1] We ignore emissions not associated with economic activity, for example from volcanic activity.

[3] $\mathbf{p^T} = \mathbf{P^X}\left[\mathbf{I} - \mathbf{A}\right]^{-1}\mathbf{y} + \mathrm{P^C}\mathbf{c} = \mathrm{P^X}\mathbf{Ly} + \mathrm{P^C}\mathbf{c}$

There are *i, j* = 1, …, N industries (where *i* denotes industry and *j* its output), **y** is the N×1 vector of total final demands for each industrial output *j* and **c** is the N×1 household consumption vector, **c**. $\mathbf{P^X}$ is a K×N matix of output-pollution coefficients, or direct pollution intensities. Each of the elements ρ_{ki}^{X} is the physical amount of pollutant *k* generated in the production of one monetary unit of output in industry *i*. $\mathbf{P^X L}$ is then the K×N matrix of output-pollution multipliers, with elements d_{kj} which give the physical amount of pollutant *k* generated in all sectors per monetary unit of final demand for industrial output *j*..

Total pollution generated on the production side of the economy, identified as the K×1 vector $\mathbf{p^X}$,is therefore determined by the product of the K×N matrix of output-pollution multipliers $\mathbf{P^X L}$ and the N×1 vector of exogenous final demands, **y**. This generates the same result as would be found by multiplying the K×N matrix of industry output-pollution coefficients, $\mathbf{P^X}$ and the N × 1 vector of sectoral outputs **x**. This extends the basic IO accounting identity that all industrial inputs and outputs can be attributed to final demand to pollution, in so far as pollution is related to the production of output.

Where final household consumption directly generates emissions, we populate a K×N matrix of final expenditure-pollution coefficients, $\mathbf{P^C}$. Each element, ρ_{ki}^{C}, is the physical amount of pollutant *k* generated per monetary unit of total expenditure on domestically consumed outputs of commodity *i*. This is then multiplied by the N × 1 vector, **c**, of total household consumption to give the K×1 vector $\mathbf{p^Y}$ of emissions directly generated in final consumption.[2]

[2] We assume here that household consumption is the only final demand sector that directly generates pollutants.

Thus [3] relates total pollution emissions generated as a by-product of economic activity (on both the production and final consumption sides of the economy) to the exogenous final demands. These are the pollution emissions which have been, or will be, accounted for under international agreements and negotiations such as the Kyoto Protocol, Copenhagen Accord and the Paris 2015 UNFCCC discussions. Note that final demands include both internal and external demand for the nation/regions output, so that export demands are included as one of the final demand categories that drive economic activity and emissions production.

As will be discussed in Section 3.4, the basic environmental IO approach to considering physical inputs of natural resources is the same as that outlined above for physical outputs of pollution. In the case of natural resources, matrices of physical IO coefficients for the resources in question replace $\mathbf{P}^{\mathbf{X}}$ and $\mathbf{P}^{\mathbf{C}}$ in [3].

As noted in Section 2, another key feature that is common to most IO environmental attribution analyses focussed on the basic system outlined in equation [3] is that they tend to limit attention to analysis of direct and indirect effects in a Type I framework. Typically, environmental attribution analyses do not endogenise household (wage) income, and household consumption expenditures associated with that expenditure, as occurs in a Leontief Type II approach.[3] There might be clear cases where a Type II analysis is more appropriate. These would typically involve local (rather than global) pollutants and geographic areas where the environmental damage produced is of particular importance. For example, concerns over the total pollution impact of, say, the decision to exploit resources in a wilderness area. In these cases the induced

[3] There are slightly different approaches to the calculation of Type II multipliers. This is discussed in Emonts-Holley et al., (2015).

local consumption could be an important policy issue and a Type II multiplier would be appropriate.

A key strength of the demand-driven Type I analysis represented by [3] is that it focuses attention on human consumption behaviour as the main driver of environmental problems. That is, the demand-driven IO approach is consistent with the perspective that, while the technologies employed in production activities may dictate the nature of emissions, producers only produce because there is a (direct or indirect) human consumption demand for their outputs. But where [3] is applied in the case of an open economy, part of the final consumption vector is export demand, and this is likely to be composed of both intermediate and final demands in other countries/regions. Moreover, production activity serving final consumption demand in the region/country being accounted for or modelled in [3] is unlikely to be limited to domestic producers. Therefore the most common extension of the basic environmental IO framework in recent years has been to move to an inter-regional variant (or multi-region where there is a paucity of data on inter-regional/inter-national trade flows).

3.2. Extension 1: the inter-regional input output framework and consumption or 'footprint' accounting

The IO framework can be extended to consider the issue of pollution spill-overs between regions/countries using the conventional inter-regional IO framework. (For convenience we refer to the geographic areas subsequently as regions, but this designation could cover countries or groups of countries). This extension, facilitated by an inter-regional IO, allows consideration of the emissions embedded in trade between regions that give rise to a difference between a region's territorial emissions of a pollutant and its total 'footprint' in terms of that pollutant. A particular region's footprint with respect to a specific pollutant or resource is the amount generated or used, directly or indirectly, to support the consumption of that region.[4] This typically involves the use of resources or generation of pollutants in other regions which supply imports of goods and services for intermediate inputs, investment, private and public consumption and other elements of final demand.

For simplicity, we analyse a two region system and begin with emissions generated in production. In this case, instead of creating a single K×1 vector of total pollutants emitted by a single Nx1 economy, the analysis produces a 2K×2 matrix of pollutants generated by a 2N×2 matrix of final demands. In equation [4], $\mathbf{p}^{X}_{r,s}$ is therefore a 1×K vector representing the amount of each pollutant k generated in production activities in region r to support final demand in region s. For example, the kth element of the vector $\mathbf{p}^{X}_{21}$ is the amount of pollutant k that is generated in production activities in region 2 to support final demand in region 1. Similarly the vector $\mathbf{y}_{rs}$ is the final demand in region r spent directly on goods and services produced in region s. Therefore $\mathbf{y}_{12}$ is the vector of imports from region 2 going directly to final demand in region 1.

$$[4]\begin{bmatrix} \mathbf{p}^{X}_{11} & \mathbf{p}^{X}_{12} \\ \mathbf{p}^{X}_{21} & \mathbf{p}^{X}_{22} \end{bmatrix} = \begin{bmatrix} \mathbf{P}^{X}_{1} & \mathbf{0} \\ \mathbf{0} & \mathbf{P}^{X}_{2} \end{bmatrix} \begin{bmatrix} \mathbf{1-A}_{11} & \mathbf{-A}_{12} \\ \mathbf{-A}_{21} & \mathbf{1-A}_{22} \end{bmatrix}^{-1} \begin{bmatrix} \mathbf{y}_{11} & \mathbf{y}_{12} \\ \mathbf{y}_{21} & \mathbf{y}_{22} \end{bmatrix}$$

$$= \begin{bmatrix} \mathbf{P}^{X}_{1}\mathbf{L}_{11} & \mathbf{P}^{X}_{1}\mathbf{L}_{12} \\ \mathbf{P}^{X}_{2}\mathbf{L}_{21} & \mathbf{P}^{X}_{2}\mathbf{L}_{22} \end{bmatrix} \begin{bmatrix} \mathbf{y}_{11} & \mathbf{y}_{12} \\ \mathbf{y}_{21} & \mathbf{y}_{22} \end{bmatrix} = \begin{bmatrix} \mathbf{P}^{X}_{1}(\mathbf{L}_{11}\mathbf{y}_{11}+\mathbf{L}_{12}\mathbf{y}_{21}) & \mathbf{P}^{X}_{1}(\mathbf{L}_{11}\mathbf{y}_{12}+\mathbf{L}_{12}\mathbf{y}_{22}) \\ \mathbf{P}^{X}_{2}(\mathbf{L}_{21}\mathbf{y}_{11}+\mathbf{L}_{22}\mathbf{y}_{21}) & \mathbf{P}^{X}_{2}(\mathbf{L}_{21}\mathbf{y}_{12}+\mathbf{L}_{22}\mathbf{y}_{22}) \end{bmatrix}$$

Note that any exogenous final demand elements that take the form of exports in the single region framework represented in [3] will be incorporated in the $\mathbf{A}_{rs}$ matrices and $\mathbf{y}_{rs}$ vectors,

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[4] The footprint calculation is usually given for a particular geographical territory but it can be calculated for any level of disaggregation See the World Wildlife Website to calculate your personal environmental footprint at `http://footprint.wwf.org.uk/

where r≠s, in [4]. For the Type I multiplier, the exports from one region that go directly to household consumption in the other region are still treated as exogenous, but the emissions that they generate in production can now be explicitly allocated to consumption in the importing region. The exports of goods and services going to intermediate use in the other region, by being incorporated in the relevant $\mathbf{A}_{rs}$ matrix becomes endogenous and dependent on changes in demand in the second region. Specifically, $\mathbf{L}_{rs}$ is that sub-matrix of the partitioned Leontief inverse that gives the total impact on the output in the producing sectors on region *r* per unit of final demand for output in region *s*.

Again, where final consumers also directly generate emissions, these are incorporated with a K×N matrix of final expenditure-pollution coefficients, $\mathbf{P}_r^C$, for each region r. Pollution directly generated by final household consumption expenditure vectors, $\mathbf{c}_{rs}$, which identify the household consumption in region r of goods and services produced in region s, is then given as[5]:

$$[5]\begin{bmatrix} \mathbf{p}_1^C & \mathbf{0} \\ \mathbf{0} & \mathbf{p}_2^C \end{bmatrix} = \begin{bmatrix} \mathbf{p}_{11}^C + \mathbf{p}_{12}^C & \mathbf{0} \\ \mathbf{0} & \mathbf{p}_{21}^C + \mathbf{p}_{22}^C \end{bmatrix}$$

where

$$[6]\begin{bmatrix} \mathbf{p}_{11}^C & \mathbf{p}_{21}^C \\ \mathbf{p}_{12}^C & \mathbf{p}_{22}^C \end{bmatrix} = \begin{bmatrix} \mathbf{c}_{11} & \mathbf{c}_{21} \\ \mathbf{c}_{12} & \mathbf{c}_{22} \end{bmatrix} \begin{bmatrix} \mathrm{P}_1^C & \mathbf{0} \\ \mathbf{0} & \mathbf{P}_2^C \end{bmatrix}$$

Summing the partitioned matrices in Equations [4] and [5] identifies all emissions in regions 1 and 2 that are attributable to the final consumption demand in each region for the outputs of

[5] If $r \neq s$, these are goods and services produced in s but imported into r, whose consumption in r generates additional emissions in r. An example would be imported petroleum.

both regions. The vectors of total emissions of each pollutant k generated in regions 1 and 2 ($\mathbf{p}_1^{\mathbf{T}},\mathbf{p}_2^{\mathbf{T}}$) are given by:

$$[7]\begin{bmatrix}\mathbf{p}_1^{\mathbf{T}}\\ \mathbf{p}_2^{\mathbf{T}}\end{bmatrix}=\left[\begin{bmatrix}\mathbf{p}_{11}^{\mathbf{X}} & \mathbf{p}_{12}^{\mathbf{X}}\\ \mathbf{p}_{21}^{\mathbf{X}} & \mathbf{p}_{22}^{\mathbf{X}}\end{bmatrix}+\begin{bmatrix}\mathbf{p}_1^{\mathbf{C}} & \mathbf{0}\\ \mathbf{0} & \mathbf{p}_2^{\mathbf{C}}\end{bmatrix}\right]\mathbf{e}$$

where e is a (2×1) unit vector.

Thus the use of expressions [4] and [5] to generate [7] again reflects the central IO accounting identity that exogenous final demands drive all activity so that $\mathbf{p}_{\mathbf{r}}^{\mathbf{T}}$ equates to total territorial emissions that would be measured for region r under the 'production accounting principle' (Munksgaard and Pedersen, 2001). Again, this is what has/will be inherent in international agreements/negotiations such as the Kyoto Protocol, Copenhagen Accord and the Paris 2015 UNFCCC discussions.

Extension to the IO accounting framework also allows consideration of the pollution emissions embedded in trade so that the total 'footprint' of regions 1 and 2 in terms of emissions of each pollutant *k* can be calculated. This is done by examining the vector of total emissions ($\mathbf{p}_{\mathbf{r}}^{\mathbf{F}}$) in both regions that are attributable to final consumption demand in region r. These can be calculated in the IO framework by transposing [4] and adding to [5] and then summing across the rows, as in [8]:

$$[8]\begin{bmatrix}\mathbf{p}_1^{\mathbf{F}}\\ \mathbf{p}_2^{\mathbf{F}}\end{bmatrix}=\mathbf{P}^F\mathbf{e}$$

where. $\mathbf{P}^F=\left[\begin{bmatrix}\mathbf{p}_{11}^{\mathbf{X}} & \mathbf{p}_{12}^{\mathbf{X}}\\ \mathbf{p}_{21}^{\mathbf{X}} & \mathbf{p}_{22}^{\mathbf{X}}\end{bmatrix}^{\mathbf{T}}+\begin{bmatrix}\mathbf{p}_1^{\mathbf{C}} & \mathbf{0}\\ \mathbf{0} & \mathbf{p}_2^{\mathbf{C}}\end{bmatrix}\right]$

. Munksgaard and Petersen (2001) call this procedure the 'consumption accounting principle'. Again, under the accounting identities inherent in the inter-regional IO framework, this could also be calculated by taking the territorial emissions of region 1, measured using the production accounting principle, adding emissions embedded in imports (here from region 2) and subtracting emissions embedded in exports from region 1 (here to region 2).

This framework is proposed by Turner et al. (2007) and has been applied by numerous authors for consumption-based accounting particularly of the main greenhouse gas pollutant, CO_2. Comprehensive reviews/surveys of applications in the literature are provided by Minx et al. (2009), Wiedmann et al. (2007) and Wiedmann (2009a). Attempts, mainly in Europe, to create environmentally extended inter-country IO databases such as EXIOBASE (Tukker et al., 2009, 2013) and WIOD (Dietzenbacher et al., 2013) have increased the accuracy of carbon footprint accounting and facilitated the analysis of carbon embedded in international trade flows. For more details see the special issue of Economic Systems Research in 2009 introduced by Wiedmann (2009b).

Summing across emissions generated in different countries to produce a consumption-based footprint measure, as in [8], might be appropriate for a pollutant like CO_2. For CO_2, the main impact of concern is climate change. This is not thought to be dependent on the spatial location of emissions, hence attempts to determine a single 'price of carbon' through mechanisms such as the European Emissions Trading Scheme, ETS. However, this is not the case across all pollutants and resource uses. That is, full global implementation of the inter-regional approach in [8] might, in principle, be used to compute 'footprints' for a wide range of pollutants and resources with spatially-independent impacts. But these figures might offer limited and/or misleading insights where the social costs of direct emissions or resource extraction differ across space.

One solution to this problem is that rather than present the figures in terms of a scalar footprint, a footprint vector can be reported that identifies not only the global total of particular emissions/resource extraction directly or indirectly supporting consumption in one location, but also its distribution across different localities. These data are provided in the $\mathbf{P}^{F}$ matrix defined in [8].

The (2K×2) $\mathbf{P}^{F}$ matrix can be partitioned into 2 (K×2) matrices, so that: $\mathbf{P}^{F} = \begin{bmatrix} \mathbf{P}_{1}^{F} \\ \mathbf{P}_{2}^{F} \end{bmatrix}$ where the 2 elements of the kth row of the matrix $\mathbf{P}_{r}^{F}$ would give the level of pollutant/resource k produced/extracted in locations 1 and 2 to support the consumption in location*r*. In the standard footprint these elements are simple summed, as shown in [8], implying that the impact does not vary across locations. In situations where this is not the case, the location of the impact can be appropriately weighted. These weights are likely to differ across pollutants. This is given as:

$$[9]\begin{bmatrix} \mathbf{w}_{1}^{F} \\ \mathbf{w}_{2}^{F} \end{bmatrix} = \left[\begin{bmatrix} \mathbf{w}_{11}^{X} & \mathbf{w}_{12}^{X} \\ \mathbf{w}_{21}^{X} & \mathbf{w}_{22}^{X} \end{bmatrix}^{T} + \begin{bmatrix} \mathbf{w}_{1}^{C} & \mathbf{0} \\ \mathbf{0} & \mathbf{w}_{2}^{C} \end{bmatrix} \right] \mathbf{e}$$

where $\left[\begin{bmatrix} \mathbf{w}_{11}^{X} & \mathbf{w}_{12}^{X} \\ \mathbf{w}_{21}^{X} & \mathbf{w}_{22}^{X} \end{bmatrix}^{T} + \begin{bmatrix} \mathbf{w}_{1}^{C} & \mathbf{0} \\ \mathbf{0} & \mathbf{w}_{2}^{C} \end{bmatrix} \right] = \mathbf{\Omega} \left[\begin{bmatrix} \mathbf{p}_{11}^{X} & \mathbf{p}_{12}^{X} \\ \mathbf{p}_{21}^{X} & \mathbf{p}_{22}^{X} \end{bmatrix} + \begin{bmatrix} \mathbf{p}_{1}^{C} & \mathbf{0} \\ \mathbf{0} & \mathbf{p}_{2}^{C} \end{bmatrix} \right]$ and $\mathbf{\Omega}$ is a (2K×2K) diagonal matrix where the diagonal entries 1 to K are the weights attributed to the corresponding K environmental pollutants or resource extractions in location 1 and the entries K+1 to 2K are the corresponding weights for location 2. All the off-diagonal enteris are zero. The weights reflect policy priorities and could be linked to the marginal damage of pollutants or the shadow prices of resource use in different geographic locations. Further, these weights could be endogenised in a general equilibrium setting.

The inter-regional IO approach is initially considered for ecological footprints in Turner et al. (2007). However, ecological footprints tend to focus on land use and the scarcity of land varies across different countries and regions of the world. Similarly, a 'water footprint' based on un-weighted summation across water uses in different countries/regions across the world will not reflect differences in scarcity and thereby social costs of water extraction at source. Therefore inter-regional accounting applied to water has tended to focus on water embedded in inter-regional trade, where the results identify whether water scarce/abundant regions are net importers/exporters of water, rather than quantify the regional 'water footprints'. An example is the Chinese study by Guan and Hubacek (2007).[6]

More generally, non-GHG pollutants, other waste products and physical/natural resource generated or used as a result of production and consumption activities pose an alternative economic problem. This is the need to determine social and/or resource costs of supplying common resources, such as a 'clean environment', at a local level. We now turn our attention to consider this issue.

3.3 Extension 2: considering the resource costs of internalising polluting externalities through incorporation of 'cleansing' sectors

Most of the empirical work using the environmentally extended IO framework does not consider how the externality might be internalised or, more generally, how the initial impact of the economy on the environment might feedback in the form of economic activity generated

[6] However, see the special issue of the journal *Economic Systems Research* on 'Input-Output and Water', introduced by Duarte and Yang (2011). This includes a review paper by Daniels et al., 2011, where the concept of water footprints is discussed.

in environmental cleansing. In his seminal paper on environmental IO analysis, Leontief (1970) extends the standard IO accounting and modelling framework in two ways. These were: to incorporate pollution as an additional commodity ("bad") that accompanies production and consumption activities; and to separately identify sectors that clean up or prevent these unwanted outputs. These sectors are generically referred to here as "cleansing sectors".

For simplicity we follow Leontief's example by restricting the analysis to one pollutant (air pollution) and a corresponding cleansing sector, although adjusting the model to incorporate many pollutants and cleansing sectors is conceptually straightforward. The Leontief approach involves expanding and partitioning the standard IO accounts as follows. First an extra row is inserted to record the air pollution generated as an additional output in each production and final demand sector. The elements of this row are calculated using the physical output-pollution coefficients, the ρ_{ki}^{X} and ρ_{ki}^{C} from equation [3]. Second, an additional column is created, showing the inputs committed to air pollution elimination or prevention.

The Leontief (1970) framework as initially stated poses a number of problems. One is the introduction of pollution as an additional (unwanted) output. Subsequent authors argued that the analysis is more straightforward if the pollution generation row and pollution elimination column are reinterpreted as the activity of a single sector that, in this case, produces clean air (Qayum, 1991; Arrous, 1994; and Luptacik & Böhm, 1999). Allan et al. (2007) proposed that clean air may be regarded as a "common pool" resource so that the row entries show the demand for replenishing the clean air implied by each sector's production activity. This corresponds to the amount of pollution generated in each sector (what is accounted for in the basic framework in Section 3.1). The additional column shows the actual inputs used to supply these cleansing activities.

More formally, we can generate an expression for the output, x_k, of a sector which cleans the pollutant *k*. This is presented in the standard IO form:

$$\textbf{[10]}\ x_k = \frac{\mathbf{p_k^X x} + \mathbf{p_k^C c} - \Delta p_k}{\lambda_k}$$

Here $\boldsymbol{\rho}_\mathbf{k}^\mathbf{X}$ and $\boldsymbol{\rho}_\mathbf{k}^\mathbf{C}$ are (1×N) vectors of production and consumption coefficents for pollutant k, Δp_k is the pollutant released into the environment and λ_k is physical quantity of pollutant k treated per unit of expenditure in cleansing sector k.

Equation [10] can be reformulated as:

$$\textbf{[11]}\ \Delta p_k = \mathbf{p_k^X x} + \mathbf{p_k^C c} - \lambda_k x_k$$

That is, the difference between the cleansing requirements generated by production and final demand consumption and the physical output of the cleansing sector, $\lambda_k x_k$, gives the amount of pollutant released into the environment. If Δp_k is negative, this implies net cleaning of the environment in order to repair earlier environmental damage or to deal with concurrent pollution by sources other than domestic production and final demand expenditure. This could include the environmental impacts of economic activity in other jurisdictions, for example through air pollution up wind.On the other hand, if Δp_k is positive, this indicates that there is pollution being emitted to the environment. This does not necessarily mean that the environment suffers, although of course it might: the environment may have some ‘carrying capacity’ or facility to naturally treat pollutants. Finally, if Δp_k is zero, this implies that the output of the cleansing sector is just enough to clean the pollutants generated in current domestic production and final demand consumption.

The system as presented so far does not in itself indicate the responsibility for financing the cleansing sector, nor the institutional arrangements for deciding the actual level of cleaning activity. A central problem in the consideration of externalities like pollution is that the level of pollutant production, and any internalisation through cleansing, is not generally determined through standard market mechanisms. In particular, the fact that production sector *i*, for example, generated pollutants that would require $a_{ki}x_i$ output of the cleansing sector to treat does not mean that such treatment will necessarily take place, or that if it does, that the cost is borne by the polluting sector.

In order to consider the use of inputs in the cleansing sector, Leontief (1970) separately identifies the column representing the fixed coefficient technology in the air pollution cleaning industry. In his pedagogic approach, Leontief proceeds as though the cleansing sector were newly created; that is, as though a cleansing sector were introduced into a system that previously generated untreated pollution. However, cleansing activity will typically already occur in the economy, whether these cleansing industries are separately identified as IO industries or not (Allan et al., 2007). Separating out the inputs used in the cleansing is then primarily a practical issue, but not one that is easily resolved, as evidenced by Leontief's own (solitary) co-authored attempt to practically apply his system (Leontief and Ford, 1972).

Allan et al. (2007) focuses on the generation of physical waste rather than air pollution. This is primarily for the practical reason that the corresponding cleaning sector (waste collection, treatment and disposal) can easily be identified as a separate sector using the Standard

Industrial Classification. This practical application draws attention to the fact that often a polluting industry's payments to the cleansing industry does not equate to the demand implied by their generation of the pollutant in question. Therefore, in developing Leontief's (1970) system, this study revisits equation [10] to identify the row of the cleansing sector, *k*, which is already specified in the IO accounts as one of the N production sectors. This is shown in equation (12). However, note that this equation determines the values of output and final demands measured in monetary units rather than the physical units in which Leontief (1970) formulates the system.

[12]

[12] $x_k = \mathbf{a_k^X x} + a_k^C c + a_k^f f$

In equation (12) $\mathbf{a_k^X}$ is the row vector of unit inputs of the cleaning industry k in the production of the N industries, a_k^C and a_k^f are the household consumption and other final demand expenditure coefficients for the output of the cleaning industry and c and f are the total household consumption and other final demand expenditures. The critical point to make is that only under exceptional circumstances will the elements of the row identifying existing sales of the output of the cleaning sector, shown in equation [12], straightforwardly map to those in the row of implied demands identified in equation [10].

Allan et al. (2007) construct a set of accounts where the expenditures by industry, consumers and government on cleaning (in their case waste treatment) match the associated physical amount of waste that is treated. In order to quantify the implied demands to populate such an adjusted set of accounts, a unit cost for the cleansing activity is calculated. This is the total value of cleansing sector output divided by the physical amount of pollution/waste that is actually cleansed. This figure is then used to value the physical amount of pollution actually generated in each production activity and in final consumption. Then by comparing the output and price multiplier results that are generated from IO systems where the cleansing sector is represented by equations [10] and [12] respectively, sectors can be identified which would ultimately (directly or indirectly) benefit/suffer if the cost of cleansing activities were to be fully internalised by the direct polluters.

The central point is that with equation [12] the price and conventional output multipliers are derived from the actual set of accounts. However, payments for cleansing services by each industry and final demand sectors do not necessarily reflect the demand implied by their actual direct pollution generation. With equation [10], on the other hand, the up- and down-stream impacts reflect the outcome (dependent on standard IO assumptions regarding linearity etc.) if sectoral payments did truly reflect this implied demand.

Application of the full Leontief model as presented in [10]-[12] has proved problematic even at the single region level, largely due to paucity of data (see Allan et al., 2007). Schäfer & Stahmer (1989) is the only IO study (for the then Federal Republic of Germany) of which we are aware where a distinct 'sector' that carries out pollution abatement services is separately identified. However, their analysis focuses entirely on the economic implications of environmental protection activities, and does not relate these to physical pollution or waste generation at the aggregate or sectoral level. That is to say, they do not proceed to an application of the full Leontief (1970) model. The Schäfer & Stahmer (1989) IO database was also later used in the Computable General Equilibrium (CGE) work of Nestor and Pasurka (1995) in order to 'externalise' any pollution abatement activity that is carried out within firms as an 'end of pipe' technology that does not affect production activities in the economy.

In principle, as demonstrated in Allan et al. (2007), there is value in attempting to develop the full Leontief (1970) framework in future IO research to consider both sectoral output and cost/price implications of internalising the costs of common resource provision. Moreover, the approach could be developed at an inter-regional level to consider the implications of spatial differences in resource scarcity and, therefore, social costs at a spatial level in order help overcome the limitation of simple un-weighted additive footprint measures for resources such as water or pollutants with localised impacts.

3.4 **IO and the materials balance principle: a dead end?**

In the previous sub-section we argue that there is value in thinking about the environment as a common pool resource with a degree of carrying capacity. In the late 1960s and early 1970s there was significant research interest in IO modelling of the economic-environmental system adopting a more holistic approach that attempted to capture the implications of the 'materials balance principle' (MBP) (Daly, 1968; Isard, 1969). The MBP arises from the 1st Law of Thermodynamics, which states that matter can neither be created nor destroyed by human activity; rather it can merely be transformed from one state to another. In the current context, this means that any waste products/residuals discharged into the environment as a result of economic activity are the necessary corollary of the extraction of material resources from it. This is because the system in which all economic activity takes place, earth's natural environment, is a thermodynamically closed system in that it exchanges energy but not matter with its environment. Incoming solar radiation is balanced against outgoing energy flows. Therefore the mass of materials extracted from the environment in any period must necessarily subsequently be returned to it. This leads to the interpretation of the MBP as implying an equilibrium relationship between the economic and environmental systems.

Early developments in environmental IO analysis seem to have been motivated by the objective of attempting to capture and analyse the implications of the MBP. This point is confirmed by Victor (1972) framing his review of five key early models - Cumberland (1966), Daly (1968), Ayres & Kneese (1969), Isard (1969), Leontief (1970) - in terms of their attention to the MBP.

Some of these early models attempt to capture flows within and between the sectors of the economic and environmental systems in an IO framework along the lines of a two region model, with the economy as one region and the environment as the other. The idea is that this type of framework tracks the flow of materials around the thermodynamically closed system implied by the MBP. No matter is destroyed but merely transformed from one state to another. For example, resources flowing from one of the environmental 'sectors' are transformed into goods and services by one or more of the economic sectors, with any residual material flowing back into one or more of the environmental 'sectors'.

The central conclusion of Victor's (1972) study of this early research is that it has proved impractical, if not impossible, to properly model the full system of economic-environmental interactions implied by the MBP. The key problems lie in a lack of knowledge about how the microstructure of the environmental system operates (at least in a way that can be translated to a multi-sector IO framework) and a lack of practical information on non-marketed flows of what Daly (1968) refers to as free goods, such as the products of the atmosphere, hydrosphere etc. or the energy provided by the sun. Isard (1968) was unable to construct a complete set of IO accounts for the system. Insurmountable issues include the fact that a materials balance accounting system would appear to require measurement at the atomic level. This renders even the notion of IO accounting of the full economic-environmental system completely impractical.

Forssell & Polenske (1998) make the point that in practice it is difficult to account for environmental sectors because the appropriate data cannot generally be obtained for them. One issue is that it would seem difficult to apportion environmental resources/pollution flows to specific individual sectors. The data availability problem primarily stems from the fact that the inputs from environmental sectors are likely to be non-marketed. Moreover, even where property rights are assigned, this implies further interactions between economic actors rather with environmental sectors.

However, there appears to be another fundamental issue that rules out the development of fully integrated economic-ecologic models. Interactions implied by the MBP take place within a closed system consisting of the economy and the environment. But the economic-environmental system will be closed in this way only at the global level; at any sub-global level, there will be economic and environmental interaction with the wider system. Whilst inter-country global IO frameworks are increasingly becoming a reality, these focus on interactions between economies, which tend to be defined in terms of political boundaries. However, the environment cannot be defined by economic or political boundaries in the same way.

Even if there is delineation of regions according to environmental boundaries, rather than political/economic ones, it is still highly unlikely that this would represent an environmentally closed system. [7] Only the global environment is a thermodynamically closed system, so that any sub-global region would, by definition, have to be environmentally open. While accounting for system openness is relatively straightforward in the case of an economic system, the same cannot be said for an environmental system. Victor (1972) ultimately goes on to propose balancing accounting identities that must hold at aggregated rather than sectoral level. These are formalised in Victor (1972, pp.61-63) but basically require that the mass of natural material inflows to economic activity equals the mass of outputs discharged back to the environment as waste products/pollution plus any material accumulations within the economy (i.e. accumulation of capital and consumer durables).

Modelling the economy and the environment simultaneously as a balanced IO system based around MBP principles seems an intriguing prospect but perhaps a step too far. In practice such attempts have proved impossible because of data difficulties. Moreover, the notion that the environment operates with linear processes and on time scales in any way comparable to the economic system seems misplaced.

4. Further examples and developments

In recent years the main application of environmental IO methods has been in identifying the carbon embedded in international trade flows. IO-based carbon accounting and 'responsibility' studies have been the dominant focus of papers published in the literature and are too numerous to list here; again we refer the reader to survey papers such as Minx et al. (2009), Wiedmann

[7] For example, Kamat *et al*'s (1999) CGE model of the US Susquehanna River Basin, which is delineated on the basis of environmental homogeneity in terms of climate and the impact of climate change.

et al. (2007) and Wiedmann (2009a).There have also been several other areas in which environmental IO methods have been applied and we review these in this section.

4.1. Water and IO

First, there has been increasing focus on developing the demand-driven IO methods detailed above for the analysis of water uses. The early contributions of Daly (1968) and Isard (1969), which have been discussed in Section 3.4, proposed the idea of incorporating water use information in an Input-Output framework. Official water satellite accounts are now constructed for some (mainly water scarce) countries, such as Spain and Australia. This has facilitated increased work with IO to analyse issues such as trade in 'virtual water' (Dietzenbacher and Velázquez, 2007) and to consider whether water scarce/abundant regions are likely to be net importers/exporters of water. IO studies commonly find that inter-regional trade violates Heckscher-Ohlin predictions in this respect. Examples are the Dietzenbacher and Velázquez (2007) Spanish study and the Guan and Huback (2007) Chinese study. As noted earlier, there have also been IO-based attempts to consider water footprints though recall that in Section 3.2 we express caution over extending the footprint concept beyond carbon or other global pollutants. Again, we do not give a more detailed account of these or other water IO studies here as comprehensive reviews are already offered the special issue of the journal *Economic Systems Research* on 'Input-Output and Water', introduced by Duarte and Yang (2011).

4.2. Physical waste and IO

Input-output methods have been specifically developed to consider waste management issues. Nakamura and Kondo (2009) reviews and proposes methodological developments in the extended use of waste input-output models, for example in terms of analysis of sustainable consumption, life cycle and materials flows analysis. This builds on their earlier contributions to the development of a waste Input-Output model that integrates waste creation and management options so that waste can be tracked through the economic system from origin to destination (Nakamura, 1999; Nakamura and Kondo, 2002). Choi et al. (2011) focuses attention on e-waste in particular and the role of recycling. More generally, as discussed in Section3.3, in terms of incorporating waste treatment, Allan et al. (2007) attempt to use the full Leontief (1970) model with a single (public) waste disposal sector taking the role of a cleaning sector.

In its simplest form (i.e. the single or inter-regional demand driven IO systems detailed in Sections 3.1 and3.2), environmental IO methods have been used to attribute waste generation to final demand for different types of sectoral outputs and different types of consumers (e.g. Jensen et al., 2013). On the other hand, others have argued in favour of revisiting the development of physical IO systems to properly assess waste flows through the economy, for example Xu and Zhang (2009).

4.3 Energy rebound effects

In recent years, the issue of whether "rebound" effects in energy use may partially, or even wholly, offset anticipated energy savings from increases in energy efficiency has become a source of considerable concern and debate in both academic and policy circles. Sorrell (2007) and Turner (2013) review this literature including detailed consideration of general equilibrium studies. Rebound, R, is a measure of the difference between the actual and "expected" reduction in energy use after the introduction of an improvement in energy efficiency. It is expressed as a percentage. Where the proportionate efficiency improvement and change in energy use are γ and $\dot{L}$ respectively, the rebound value is formally defined as:

[13] $R = \left[1 + \frac{\dot{L}}{\gamma}\right] \cdot 100$

Therefore after the introduction of a 10% improvement in energy efficiency, if the actual proportionate change in energy use is -6%, then the rebound value is 40%. It can be measured at direct, indirect, economy and world-wide levels. As the scope of the measure widens it takes into account of how energy use changes not only for the user directly affected by the efficiency improvement, but also other producers and consumers. These impacts occur through changing patterns of intermediate and final consumption demands which are affected by the changes in the level and composition of income, output and prices.

The primary driver of the rebound effect is the reduction in the price of energy measured in efficiency units. Given that in its conventional interpretation IO is a fixed price modelling technique, the effect of this price change (and related price changes in other commodities and factor incomes) cannot be incorporated endogenously. Demand-driven quantity IO models therefore have great difficulty in dealing with efficiency improvements that take place in production, which constitutes a change in supply conditions, so that energy efficiency/rebound studies have primarily been more the domain of CGE models. Nevertheless, , as outlined in Lecca et al. (2014), an interesting area of rebound research has recently developed in using the demand-driven IO method outlined in Section 3.1. [8] This work relates to an element of rebound

[8] Lecca et al (2014) use a full CGE analysis to incorporate endogenous income and price effects in their study of increased efficiency in UK household energy use.

that is associated with changes in energy efficiency in household consumption which uses IO Type I modelling techniques as shown in equation [3]. Essentially IO can be used to identify changes in the intermediate demands for energy use that accompany the primary changes in household final demand if nominal income and prices are held fixed.

When energy efficiency improvements are made in household consumption, this involves a reallocation of expenditure by households. As long as the rebound is less than 100%, so that backfire does not occur, after the efficiency improvement households will spend a smaller proportion of their income on energy and a greater proportion on all other goods and services. IO methods have then been used to assess and measure the embodied energy effects of this re-allocation of expenditure. Two interesting examples are found in Druckman et al. (2011) and Freire-Gonzàlez (2011) for UK and Spanish (Catalonia) case studies respectively. Indirect energy use embodied in re-spending decisions is found to be large in a number of scenarios modelled.

In considering IO multiplier results it is crucial to recognise that as well as the increased energy embodied in the consumption goods that are allocated a higher spending share, there will also be the reduced embodied energy requirements from energy-savings (inputs and outputs of the energy supply industries in the IO model) where the direct rebound is less than 100%. Increased consumption of non-energy goods and services involves increased embodied energy requirements down their supply chains given positive multiplier effects. But reduced consumption of energy involves decreased embodied energy requirements as less energy and non-energy inputs are needed in the supply chains of energy producers through negative multiplier effects. Moreover, energy production (for example, electricity generation in gas- or coal-fired plants) tends to be both directly and indirectly energy-intensive. Thus, there is a strong chance that redirected spending away from the energy-intensive outputs of energy supply sectors in favour of less (directly and indirectly) energy-intensive non-energy goods and service will lead to a net negative embodied energy effect. This effect therefore operates to reduce the overall rebound effect and could make rebound negative. That is to say, the incorporation of embodied energy in the relevant supply chains could lead to actual reduction in energy use being greater than the initial efficiency gain.

For this to be captured by IO models requires two steps. First an estimate must be made off-line as to the changes in household consumption generated by the efficiency improvements. Second a full set of expenditure changes, both positive and negative, need to be introduced in the standard IO model. Lecca et al. (2014) demonstrate that the net impact of this indirect element of the total general equilibrium rebound effect is likely to be negative. Tamba (2014) extends this approach to focus on improved household efficiency in the use of electricity through the introduction of smart meters. She explicitly micro-models the impact on electricity, gas, and other household consumption and considers the implications for embedded intermediate demands for energy if gas and electricity are complementary or competing goods.

This general perspective is disputed in a paper by Guerra and Sancho (2010). They argue that negative multiplier effects in the energy supply chain, that is reductions in direct and indirect use of energy by energy producers through their intermediate use of both energy and non-energy inputs, should be considered as part of potential (expected) energy savings. That is to say, in rebound calculations the "expected" reduction in energy use should not just be "expected" direct reduction in the use of energy but also the reduced energy that is embedded in the supply chain producing the energy. Thomas and Azevedo (2013a,b) follow the Guerra and Sancho argument in their consideration of indirect rebound. This, by definition, increases rather than decreases the size of the measured indirect rebound effect.

The issue can be illustrated with an example which features an improvement in energy efficiency in consumption with prices and consumer income held constant. If an initial share, θ_E, of household consumer spending, c, is spent on energy, in rebound studies the standard "expected" reduction in energy use (measured at constant prices) of an energy efficiency improvement of γ is $\gamma\theta_E y$. The actual proportionate reduction in household energy use will typically be positive, so that backfire does not occur, but will be less than the expected reduction, primarily because the fall in the price of energy to households, measured in efficiency units. IO can't help us with this: it is a calculation that has to done off line. If the actual change in household energy use is λ, it is straightforward to calculate the rebound, R_P, from a purely partial equilibrium perspective (i.e. simply focus on the change in household energy consumption and holding output and household income fixed). Using equation [14]:

[14] $$R_P = \left[1 - \frac{\lambda}{\gamma}\right] \cdot 100 .$$

One very common argument is that this calculation underestimates the rebound effect. This is because where the household expenditure on energy falls, the expenditure on other commodities will rise. The present calculation fails to take into account the energy required for the production of the increased output of non-energy household consumption. This embodied energy can be calculated using a Type I energy multiplier.

For illustrative purposes, imagine that the economy has only two sectors, energy and non-energy (E andN). Using the conventional Leontief inverse a row vector of energy multipliers, **m**, can be constructed as follows:

$$[15]\ \mathbf{m} = \boldsymbol{\eta}\left[\mathbf{I} - \mathbf{A}\right]^{-1}$$

where $\boldsymbol{\eta}$ is a row vector of direct energy coefficients. The elements $m_E$and m_N of the multiplier vector are the energy directly and indirectly embodied in one unit of final household consumption of energy and non-energy goods respectively. If the actual energy saving is $\lambda\theta_{EC}$, the additional energy indirectly embedded in the reallocated household expenditure will be $m_N\lambda\theta_{EC}$ This has to be subtracted from the direct energy saving, so that in equation (16), λ is replaced by$(1-m_N)\lambda$. The new rebound value will, which we label as R_1, is calculated as:

$$[14]\ R_1 = \left[1 - \frac{(1-m_N)}{\gamma}\right]\cdot 100 .$$

Clearly $R_1 \geq R_P$, so that including these multiplier effects increases the rebound value (and therefore reduces the measured energy saving). But if the embodied energy in the increased non-energy spending is to be incorporated, then we should also include the increased energy savings from the reduced amount of energy needed to produce final demand household energy. This equals $m_E\lambda\theta_E y$. This should be added to the energy saving, so that the true rebound value, R_2, (given the assumptions of fixed prices and household income) is:

$$[14]\ R_2 = \left[1 - \frac{\lambda(1-m_N+m_E)}{\gamma}\right]\cdot 100 .$$

Given that energy production is generally relatively energy intensive, we expect $m_E > m_N$, so that $R_1 \geq R_P > R_2$.

Taking into account the changes in the embedded energy required to meet the changes in household consumption actually reduces the rebound value. This runs counter to the common view that extending the scope of the rebound calculation to take in more general equilibrium effects increases the rebound value. Further, if $\frac{\lambda}{\gamma} > \frac{1}{(1-m_N+m_E)}$ then the rebound value R_2 is actually negative. This means, for example, that a policy initiative that increases energy efficiency in household consumption by 5% will, in fact, reduce energy use across the economy as a whole by an amount greater than 5% of the initial household energy use.

Guerra and Sancho (2010) and Thomas and Azevedo (2013a,b) define the "expected" energy saving to include the reduced energy embedded in the production of the household energy use. This generates the rebound value R_3, (given the assumptions of fixed prices and household income) specified in equation (15):

$$[15]\ R_3 = \left[1 - \frac{\lambda(1-m_N+m_E)}{\gamma(1+m_E)}\right] \cdot 100 .$$

Given our assumptions about the relative values of the energy multipliers for energy and non-energy household consumption, this rebound value will always be above the partial equilibrium measure. The ordering is $R_1 \geq R_3 \geq R_P > R_2$.

There are three issues here. First, if rebound is taken to be a measure of all the endogenous changes in the extent of energy use that occur as a result of the initial efficiency improvement, then the changes in intermediate energy use, as identified using IO, should be included. Second, only including the embedded energy in the increased consumption of non-energy goods, as in R_1, is clearly incorrect. Third, although there is an intuitive appeal to including the embedded energy in the production of energy itself as a part of the "expected" change in energy use, as in R_3, it means that the energy multiplier effects are treated in an asymmetric way. This is rhetorically misleading in that it artificially increases the rebound value, which is often taken by policy makers as a negative indicator of the effectiveness of the policy.

4.4 Supply constraints, endogenous technology change and price effects

In the conventional demand-driven IO model, supply is passive. Issues of resource scarcity, choice of technique or price effects do not arise endogenously in the model. In each industry there is one technique, unlimited supplies of the non-produced inputs and constant returns to scale. However, there is a supply-driven version of the IO model where output is determined by supplies of non-produced inputs (Ghosh, 1958). Additionally, linear systems of production, essentially IO sets of accounts, can form the basis for, and are embedded in, more detailed analysis of income distribution between institutions and also household consumption available in Social Accounting Matrix (SAM) analysis (Round, 2003). Similarly, such IO accounts are central to the classical treatment of price and aggregate distribution approach of Sraffa, (1960), and the more neo-classical Computable General Equilibrium models (Conrad, 1999). Finally, IO systems can be used in linear programming to identify optimising choice of technique in systems operating under resource and legislative constraints (ten Raa, 2006, Ch. 11).

For many environmental applications, issues such as the restricted nature of natural or common pool resources are central. Of course, the implications of exogenous changes in final demand or technical coefficients on the availability of such resources or, more usually, policy-driven limitations to their use, can be analysed in a standard demand-driven setting and many examples have already been presented of such an approach. However, there are models that use an IO base to make endogenous choices about production techniques. These have also been applied to tackle environmental issues.

Typically, the system-wide impacts of environmental policy which are driven by supply-side concerns or which are strongly affected by supply-side constraints are most straightforwardly dealt with in a Computable General Equilibrium (CGE) analysis, where, in effect, conventional IO would be a special, limiting, case. Nevertheless, it is possible to use IO, especially where linked to linear programming or other optimising procedures, to tackle certain supply-side issues.

The solutions to resource use problems often involve influencing the choice of particular production technique, generally towards the adoption of more sustainable technologies. In a CGE analysis this involves adjusting a production or consumption parameter and allowing the model to then generate the endogenous choice of technique, given the subsequent income and price changes. This obviously has important advantages but makes the change rather abstract. Take as an example the seemingly paradoxical "backfire" result that in the conventional neo-classical analysis, of which CGE is typically an example, increasing the energy efficiency in production will simultaneously increase the energy intensity of production if substitution elasticities in production are greater than one (Turner, 2013). Whilst this is a straightforward proposition for economists, environmental analysts might, quite understandably, not wish to categorise such a change as an energy efficiency improvement at all. A situation, as with IO, where technological alternatives can be defined in specific and discrete forms, therefore has advantages.

IO approaches can be used to compare specific technologies. For example, Ebiefung and Kostreva (1993) formulate the Leontief system as a generalised linear complementarity problem and use this to choose amongst possible specific new technologies. Pan (2006) extends this general approach and uses a dynamic IO system that incorporates endogenous technical change and the choice between vintage technologies. In this particular case, the analysis is used in an indicative attempt to model the time path of the replacement of fossil fuels by nuclear generation in the Chinese economy.

Ten Raa and Shestalova (2014) also demonstrate how IO can be employed in a manner that allows the choice of technique to be endogenous, but here driven by trade. Their linear-programming based model treats the introduction of environmental standards as additional constraints, similar to those imposed by fixed supplies of capital and labour. In what they refer to as a pilot application, they model the optimal distribution of industrial output across Spain, Denmark and Belgium under various forms of regulation, if their activities were co-ordinated and facilitated by free trade between these countries. Specialisation between countries here affects the choice of technique and product price.

5. Practical problems of implementation

An important practical issue that is likely to influence the applicability and acceptance of environmental IO methods is the availability and quality of data. Creating an IO database is a highly resource intensive task. But this is mitigated by the fact that IO accounts are now routinely constructed for most industrialised countries as part of national accounting under the UN System of National Accounts and, within the EU, Eurostat. However, while standard IO tables are typically published with very high levels of sectoral disaggregation, a lack of appropriate disaggregation is a common problem for environmental IO applications.

As Hawdon and Pearson (1995) explain, because IO tables are not normally designed with the main purpose of exploring energy-environment questions, the sectoral classifications often over-aggregate important energy sectors and combine industries with significantly different pollution characteristics. Lenzen (1998) and Gale (1995) both cite over-aggregation as a principle shortcoming of their analyses, with respect to fuel-use and electricity sector data respectively. Lange (1998) also raises the issue of IO sectoral classifications being incompatible with environmental concerns, particularly if the modeller wishes to track natural resource use at a detailed level. Similarly, as argued in Allan et al. (2007) and demonstrated by Leontief and Ford (1972), the identification of cleaning, recycling and treatment sectors is problematic.

Therefore, there may be a need for further, often extensive, disaggregation of existing IO tables. This is a process which is likely to have significant cost implications or to rely on assumptions as to how an existing sector should be further disaggregated.[9] While environmental satellite accounts are increasingly being constructed with particular focus on identifying key polluting/resource using sectors, there are often issues in terms of mapping to the Standard Industrial Classification that underlies the categorisation of IO sectors. The NAMEA framework (de Haan and Keuning, 1996) – National Accounting Matrix with Environmental Accounts – does attempt to resolve this issue. This has been the focus of work by several international bodies such as the European Environment Agency[10], Eurostat and the OECD.

Inter-country IO databases are required for the application of the systems detailed in Section 3.2 commonly used for consumption-based accounting of carbon emissions. The availability of such accounts has improved dramatically in recent years as a result of projects such as EXIOBASE (Tukker et al., 2009, 2013) and WIOD (Dietzenbacher et al., 2013). Still, two

[9] For example Allan, et al, (2006), had to directly approach individual plants to disaggregate the Scottish electricity generation sector whilst Gale (1995) uses assumptions based on foreign data in order to disaggregate the same sector in Mexico.
[10] See results at http://www.eea.europa.eu/data-and-maps/data/external/namea-project-eu-27-calculations.

crucial problems remain. First, these tend to be more highly aggregated than published national IO accounts, so that the problems discussed above concerning over-aggregation reappear in this context. Second, whilst the existing inter-country databases result from projects funded by programmes such as the EU FP7 framework, they do not yet have the same status as published national IO accounting data. Similarly the OECD have been working for some years on harmonising national IO tables and bi-lateral trade data to improve their own inter-country IO database, which has already been used for carbon accounting.[11] Nevertheless, it is likely to be some time before these data are judged to be considered as reliable, particularly by policy communities.

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[11] See http://www.oecd.org/industry/ind/carbondioxideemissionsembodiedininternationaltrade.htm

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