ELSEVIER

Ecological Economics 41 (2002) 457–478

ECOLOGICAL ECONOMICS

This article is also available online at:
www.elsevier.com/locate/ecolecon

SPECIAL ISSUE: The Dynamics and Value of Ecosystem Services: Integrating Economic and Ecological Perspectives

Ecological production based pricing of biosphere processes

Murray G. Patterson *

School of People, Environment and Planning, Massey University, Private Bag 11-222, Palmerston North, New Zealand

Abstract

Ecological pricing theory and method is reviewed, and then applied to the valuation of biosphere processes and services. Ecological pricing values biosphere processes, on the basis of biophysical interdependencies between all parts of the ecosystem, not just those that have direct or obvious value to humans. The application of the ecological pricing method to the biosphere for 1994, indicates that the total value of primary ecological inputs (services) to be nearly $US 25 trillion. This compares with $US 33 trillion obtained in the Costanza et al. (1997) study. Our analysis also indicated a good correspondence between the shadow ecological price and the observed market price for all marketable goods, except fossil fuel which was undervalued by the market. © 2002 Elsevier Science B.V. All rights reserved.

Keywords: Valuation; Biosphere; Ecological prices; Biophysical models

1. Biophysical basis to ecological pricing

Ecological prices are the 'weighting factors' inferred from models which describe energy and mass flows through ecological and economic systems. It is critical that these models be constructed to adhere to biophysical principles and laws, if ecological prices are going to make ecological sense. If these base models are founded on questionable or inappropriate concepts (from a biophysical viewpoint), then it follows that the 'prices' and 'values' derived from such models will also be of questionable validity.

Patterson (1998a,b), for example, argues in this respect that the simple translation of the Sraffa (1960) model of price determination as suggested by England (1986) and Judson (1989), makes little or no biophysical sense, as it contravenes a number of biophysical principles. Amir (1989, 1994) similarly argues that uplifting Leontief (1986) equilibrium based input–output pricing models from economics and applying them to ecological systems produces results which are inconsistent with the Second Law of Thermodynamics.

The first principle that must be adhered to in ecological pricing is that of the *First Law of Thermodynamics*. That is, the energy input into all economic and ecological processes, must equal the energy output of those processes. The same applies to mass. Frequently, in neoclassical accounting systems some inputs (e.g. solar energy) and

* Present address: Landcare Research Ltd., Private Bag 11-052, Palmerston North, New Zealand

E-mail address: m.g.patterson@massey.ac.nz (M.G. Patterson).

PII: S0921-8009(02)00094-0

some outputs (e.g. low temperature heat) are systematically ignored. Indeed, very few economy-environment models explicitly take account of these mass or energy conservation principles, with Victor's (1972) input–output modelling system being one of the few exceptions.

The implications of the *Second Law of Thermodynamics* are also important in terms of the formalism of ecological pricing. The dissipative nature of all ecological and economic processes implies there is a degradation of the (thermodynamic) value of energy and materials, which must be taken into account in ecological pricing. This leads to methodological difficulties in ecological pricing, as *a priori* values need to be assigned to the value of 'waste heat' (Costanza and Neill, 1984; Patterson, 1998a,b). Often this requires using the Carnot Formula to assign value to 'waste heat' which is problematical due to the infinite time (irreversibility) assumptions that need to be utilized. More often, however, a zero value is assigned to 'waste heat' on the basis that it is, by definition, irrevocably lost from the biosphere (as far infrared radiation) and therefore is of no value to the system.

It must be acknowledged in ecological pricing methods that ecological and economic systems are *open systems*, based on the unidirectional degradation of low entropy energy and matter. As argued by Gilliland (1977) and Daly (1985), economic systems are not closed systems based on circular flow, as is often assumed in economic models. Instead, *economic systems* are open systems because they draw on 'resources' from the biophysical environment and release 'wastes' back into the biophysical environment. Increasingly with the globalisation of trade patterns, individual economic systems are becoming even more open and less localized, with ecological footprints that reach far beyond their national borders. *Ecological systems* (or a collection of ecological systems as used in the flow model presented in this paper) are also open systems because they involve the throughput of both energy and material flows—global systems of atmospheric circulation alone ensure this is the case, even though there might be otherwise containment in terms of internal nutrient cycling.

Ecological systems are complex, as are the processes within them. Each process has multiple inputs and outputs (joint products)—not single products as prescribed by some linear models in economics. The processes are all *interdependent* and interconnected. Ecological prices cannot, therefore, be determined by just considering one transformation between ecological commodities in isolation. Ecological prices are a function of many interconnected transformation processes.

Ecological prices will change over time as the system changes and adapts—we are dealing with *dynamic systems*, even though we may be taking a static snapshot of ecological prices. It is almost certainly the case, at any particular point in time, that the ecological system will most likely not be at equilibrium, if we are to believe the evidence of modern ecology (DeAngelis and Waterhouse, 1987; Pimm, 1991). Emergent evidence from ecological economics also suggests that economic systems at any point in time are most likely not to be at equilibrium (Perrings, 1996). Ecological pricing models must therefore not assume *a priori* general equilibrium conditions, which unfortunately often is the case; and furthermore they must be able to reflect dynamic changes in the system.

2. Defining ecological prices and efficiencies

2.1. What are ecological prices

Ecological prices are ratios that measure the 'value' of an ecological commodity—e.g. solar energy per kilogram of apples. In a broad sense they are analogous to market prices, which are also ratios that measure 'value' per unit of commodity—e.g. dollars per kilogram of apples.

The key difference between 'ecological prices' and 'market prices' is that ecological prices measure value in terms of the biophysical interdependencies in the system; whereas 'market prices' are based on consumer preferences and other factors that determine the exchange value in markets.[1,2]

[1] Typically, ecological prices are measured in terms of biophysical units (e.g. solar equivalents), although it is possible to express ecological prices in monetary equivalents (refer to Section 4 of this paper). This can be a useful device in communicating the results of an ecological pricing exercise to an audience more accustomed to using monetary units.

Essentially, there are two types of biophysical interdependencies in an ecological/economic system—backward linkages and forward linkages, both of which can play a role in determining the ecological price of a commodity. *Backward linkages* involve tracking the direct and indirect *inputs* of mass and energy into a process. Oppositely, *forward linkages* involve tracking the direct and indirect effects of *outputs* of energy and mass from a process. Given the complexity and level of feedback of ecological-economic systems, most commodities have both forward and backward linkages which both determine their ecological price. However, it is important to note that some commodities only have forward linkages which are the only determinants of their ecological price—e.g. geothermal energy, virgin minerals and solar energy in the biosphere model used in this paper only have forward linkages. Occasionally, but very rarely, some commodities only have backward linkages.

Ultimately ecological prices measure how much value one ecological commodity (e.g. solar energy) contributes to another commodity in the system (e.g. autotrophs or decomposers). In this context, if it takes 200 MJ of solar energy to produce 1 kg of biomass, then the ecological price of a kg of biomass is 200 MJ of solar energy. There is an implicit equivalence between solar energy and biomass, based on natural biophysical flows.

2.2. Graphical illustration of ecological prices and efficiencies

The essential mathematical characteristics of ecological pricing can be demonstrated using a simple two commodity × three process system, even though we know that such systems do not exist in the real world (refer to Fig. 1).

The two commodities are solar energy and biomass with three conversion processes (A, B, C). The fitted line (β_1) is the average conversion efficiency of the three processes which convert solar energy to biomass. This 'average conversion efficiency' is the ecological price. Odum (1996) uses the term 'transformity' to measure the efficiency of transformation between energy commodities—which in a sense is similar to the idea of 'ecological price'. It is important to note in this simple system that not all processes are equally efficient (B and C are more efficient than average, and A is inefficient). The vertical distance between the points (A, B, C) and the fitted line (β_1) is termed the residual (e). Previous methods of 'ecological pricing' have assumed a *priori*, that all processes are equally efficient by using square matrices ($m = n$) which either requires aggregation of processes or elimination (of inefficient) processes. Whether these less efficient processes are 'out-competed' over time (due for example because of the maximum power principle) is an interesting issue which requires more empirical and theoretical testing.

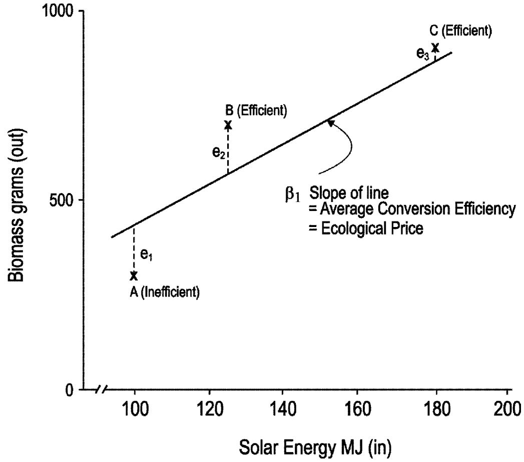

Fig. 1. Ecological prices and efficiencies for a three process × two commodity system.

The ecological price of the two commodities in this simple example can be determined by solving the following system of equations, by using the Eigenvalue–Eigenvector method developed by Wake et al. (1998) which finds the least squares solution:

[2] 'Pseudo' market prices can also be determined to measure the exchange value of commodities that have no market price (e.g. ecosystem services) by setting up 'pseudo markets' in contingent valuation surveys.

Process A: $100\beta_1 + e_1 = 300\beta_2$

(100 MJ solar → 300 g biomass)

Process B: $125\beta_1 + e_2 = 700\beta_2$

(125 MJ solar → 700 g biomass)

Process C: $180\beta_1 + e_3 = 900\beta_2$

(180 MJ Solar → 900 g biomass)

The solution of these equations can be expressed in terms of biomass equivalents:

$$\beta_1 = \frac{4.9661 \text{ g biomass}}{1 \text{ MJ solar energy}}$$

$$\beta_2 = \frac{1 \text{ g biomass}}{1 \text{ g biomass}} \text{ (by definition)}$$

The ecological price (β_1) means that on average 1 MJ of solar energy produces 4.9661 g of biomass.

The ecological efficiencies can be determined by undertaking the following calculation for each process:

$$\frac{\sum(\text{Price} \times \text{Quantity of Outputs}) = \text{Total Value of Output}}{\sum(\text{Price} \times \text{Quantity of Inputs}) = \text{Total Value of Input}}$$

For example, the ecological efficiency of Process A can be calculated.

– Total value of outputs (Price × Quantity): $1.000 \times 300 = 300$.

– Total value of inputs (Price × Quantity): $4.9661 \times 100 = 496.61$.

– Efficiency of Process A (Total value of outputs/Total value of inputs): $300/496.61 = 0.60$.

Similarly, the efficiencies of Process B (1.13) and Process C (1.01) can be calculated. Process A therefore, has a below average efficiency (0.60), so it has a negative residual ($e_1 = -197$ g). Processes B and C have above average efficiencies so they have positive residuals ($e_2 = 79$ g, $e_3 = 6$ g).

2.3. Mathematics of ecological prices for more complicated systems

Real world ecological systems are far more complicated than that portrayed in the simple example (Fig. 1) in a number of respects.

1. There are *many commodities and many processes*, not just two commodities and three processes as described by Fig. 1. Even a relatively simple ecosystem is likely to have on the order of 10–20 commodities and processes, even at a coarse level of aggregation (Costanza et al., 1983).
2. There are *multiple inputs and multiple outputs per process*. For example, the photosynthetic process described in Fig. 1, in reality has more than the one input (solar energy) and output (biomass). Instead, it has multiple inputs and outputs:

   $$\text{solar energy} + CO_2 + H_2O \rightarrow \text{biomass} + O_2$$

   Some linear models (e.g. Leontief) assume only one output per process.
3. The transformation (conversion) *processes are interconnected* with each other through complex networks of energy and mass flow. In Fig. 1, there are three independent processes with no feedback of energy or mass.
4. Fig. 1 processes only involve one direct input and one direct output in the ecological price determination. In reality, *indirect inputs* (backward linkages) and *indirect outputs* (forward linkages) are also critical in determining ecological prices.

These ‘complications’ of real world systems mean that ecological prices cannot simply be graphically determined as portrayed in Fig. 1. Instead a system of simultaneous equations must be employed to describe the complexity of the system and to determine the ecological prices. Each equation in the system of equations describes the input and output of mass and energy for each process in the system. By solving these equations the ‘ecological price’ of each commodity in the system is determined.

In broad terms, there are three categories of mathematical solution methods that have been developed to determine ecological prices for different types of systems of simultaneous equations (Determined, Underdetermined, Overdetermined).

2.3.1. Determined systems

Costanza and Neill (1981a) first developed a method based on Leontief's input–output matrix algebra to determine the ecological prices of various biosphere commodities. The application of this method was replicated in various publications through the 1980s and 1990s: Costanza et al. (1983), Costanza and Hannon (1989) and Costanza (1991). This method requires the recognition of one primary input into the ecological system (solar energy), which is used as the numeraire, and assumes there is an equal number of endogenous commodities and processes.[3,4] This, of course, is rarely the case and instead the analyst is 'forced' to aggregate commodities and/or processes to arrive at a square matrix (Fruci et al., 1983; Patterson, 1991). Ultimately, this aggregation has an arbitrary element to it. Costanza and Neill (1981a) also encountered two further problems; (i) the situation of 'negative ecological prices' (resulting from joint production) being generated by this solution method which are difficult to interpret conceptually; (ii) the problem of invertability of the matrix due to linear dependence between rows in the matrix. To avoid these problems, this required manipulation and aggregation of the base data away from what the authors considered to the 'canonical' form of the matrix.

[3] Real world ecological systems almost always have more than one primary input, which presents problems in the strict application of this method—e.g. the biosphere has inputs of gravitational, rotational, geothermal, nuclear and lunar energy in addition to solar energy.

[4] This system of ecological pricing does not necessarily require solar energy to be used as the numeraire. Patterson (1998a,b) demonstrated that other commodities can be used as the numeraire, generating exactly the same price relativities as does solar energy when it is used as the numeraire.

2.3.2. Underdetermined systems

Underdetermined systems of equations have fewer equations than unknown variables. When these equations are accompanied by the specification of constraints and an objective function, we are dealing with a linear programming problem. In view of the problems of negative prices and an unequal number of commodities and processes, Fruci et al. (1983) and Costanza and Neill (1984) developed a linear programming method to determine 'optimal' ecological prices and 'optimal' process activities. This means that sub-optimal (uncompetitive) processes have zero activity and therefore they do not enter into the calculation of the ecological prices. This linear programming method requires the specification of just one objective function, which is difficult to defend on theoretical grounds. For example, Costanza and Neill (1984) selected the objective function of 'minimization of the total solar energy input into the system', which has overtones of Odum's (1971, 1996) maximum power principle. Others such as Jorgensen (1998) advocate exergy as being an appropriate objective function in ecosystem analysis. In general terms, however, it is doubtful whether an ecological-economic system does (or for that matter should) operate according to just one objective function.

2.3.3. Overdetermined systems

Patterson (1983, 1991, 1998a,b) has developed various methods suitable for solving overdetermined systems of equations where there are more equations (processes) than unknown variables (ecological prices). The advantage of these methods is that there is no requirement to aggregate processes prior to solution which may be quite arbitrary. Furthermore, there is no need to *a priori* specify an objective function as in the linear programming method. Essentially, these solution methods determine the 'average' conversion efficiency as the basis for the ecological price, rather than the 'optimal' conversion efficiencies as in the linear programming case. As depicted by Fig. 1, this then leads to the identification of the relative 'ecological efficiencies' of competing processes—some have lower than 'average' and others higher than 'average' efficiencies. Initially, regression methods were used to solve the overdetermined system of equations by selecting as the dependent variable the commodity that generated the highest R^2 (Patterson, 1983, Patterson, 1987). If 'negative prices' became an issue, constrained least squares methods such as those developed by Judge and Takayama (1966) and Lawson and Hanson (1974) could be utilized. More recently, Patterson (1998a,b), Wake et al. (1998), Collins and Odum (2001) have developed an Eigenvalue–Eigenvector method to obtain a more rigorous and generalised solution to these equations. This method involves determining the minimum eigenvalue $\boldsymbol{\lambda}_{\mathbf{min}}$ and its associated eigenvector $\boldsymbol{\beta}$ for:

$$(\mathbf{X}^{\mathbf{t}}\mathbf{X})\ \boldsymbol{\beta} = \boldsymbol{\lambda}_{\mathbf{min}}\boldsymbol{\beta} \tag{1}$$

Where, $\mathbf{X}$ is the flow matrix $(m \times n)$ describing the inputs $(-)$ and outputs $(+)$ from m processes in an ecological/economic system, measured in energy/mass units. There are n inputs and n outputs. $\mathbf{X}^{\mathbf{t}}$ is the transpose of $\mathbf{X}$. $\boldsymbol{\beta}$ is the eigenvector $(n \times 1)$ of ecological prices. $\boldsymbol{\lambda}_{\mathbf{min}}$ is scalar (1×1) representing the minimum eigenvalue.

It can be demonstrated that the eigenvector β associated with $\lambda_{\min}$ represents the best non-trivial solution according to the least sum of squares criterion.

3. Heritage of ecological pricing

There are clear antecedents for ecological pricing theory and method, both in the economics and ecology literature, as well as in the Energy Analysis literature.

3.1. Economic foundations

In the history of economic thought, there is a fundamental cleavage between the 'cost of production' and 'subjective preference' approaches to valuation and pricing. The 'ecological pricing' method is a 'cost of production' method in the sense that it measures embodied input (of mass and energy) and it uses physical equivalents as the numeraire. Whereas willingness-to-pay (WTP) and the other neoclassical methods referred to in this Special Issue and used in the original *Nature* paper are 'subjective preference' methods—they are based on ideas of consumer sovereignty and preferences and use monetary units as the numeraire.

The earliest theoretical antecedent of the 'ecological pricing' method can be traced to the Physiocrats (an early School of French economists in the 18th century). For the Physiocrats, all value is derived from land (nature) and in this sense agriculture is seen as the only 'productive' sector in the economy, with the manufacturing and the service sector considered to be 'sterile'. Within, the context of this ecological worldview, the Physiocrats considered the value of a commodity to be solely determined by the embodied land inputs required to produce the commodity. The more land required, the greater the ecological cost, and hence the more valuable the commodity.

These 'cost of production' based ideas persisted in classical economics in the form of the 'labour cost' theory of value which formed one of its fundamental cornerstones. Adam Smith (1723–1790) first developed a 'labour cost' theory of value, which asserted that the value of a commodity is determined by the total labour it took to produce the commodity—i.e. by its *embodied labour* content. Interestingly, Adam Smith also developed a *labour commanded* theory of value, which implied that the value of a commodity is determined by what people are prepared to pay for it—i.e. how much labour the commodity can command. This is the clear forerunner of the Neoclassical subjective preference theory of value (and the WTP idea of value determination).

Ricardo (1772–1823) amongst the classical economists, in particular, focused his attention on developing the 'labour cost' theory of value unequivocally rejecting Smith's 'labour commanded' theory of value. Ricardo attempted to 'prove' the embodied labour theory of value by demonstrating that the embodied labour content of a commodity provided an explanation of market prices. He was only partially successful in this endeavour, and settled for what Stigler (1965) coined a '93% labour theory of value', as embodied labour only explained 93% of the variation in market prices. Ricardo also recognised that the labour input theory of value only holds true if there is a constant capital:labour ratio in all sectors. If this assumption is dropped then the direct relationship between labour input and price cannot be guaranteed, or worse still, there is interdependence between income distribution and price. Ricardo recognised that the resolution to this problem was to find an 'invariable standard of value' not affected by income distribution.

The 'cost of production' approaches, in fact dominated classical economics with all of the major theorists (Smith, Ricardo, Malthus, Marx, J.S. Mill) all subscribing to various adaptations of the 'labour theory of value'. For further discussion, refer to Farber et al. (2002) in this Special Issue.

The neoclassical (marginal) revolution took place at the end of the 19th century, culminating in Alfred Marshall's now well-known supply and demand curve diagram. According to this schema, value was now determined by marginal benefits (demand curve) and marginal costs (supply curve). From this point onwards, the 'cost of production' theories of value were largely forgotten by orthodox mainstream economists. Instead, the 'subjective preference' theory of value dominated the discipline. An in-depth discussion of the philosophical and historical underpinnings of the neoclassical 'subjective preference' theory of value is contained in this Special Issue—refer to Farber et al. (2002).

Despite the hegemony of the 'subjective preference' theory of value in 20th century economics, there were a number of theoretical developments from mainstream economics, which provide some foundations for the 'ecological pricing' approach. Firstly, von Neumann (1945), renowned mathematician who turned his hand to economics developed a general equilibrium-pricing model based on physical input–output relationships. Under an objective function of maximising profit rate and a constraint specifying a constant rate of growth, optimal prices and process activities were determined. These von Neumann 'prices' are similar to 'ecological prices' in the sense that they are determined by physical input–output relationships. Although there is a similarity between von Neumann's pricing and ecological pricing, clearly von Neumann's model was designed to be applied to economic systems and to reflect the normative assumption of the neoclassical mainstream of 'maximising profit'. Whereas, in the context of ecological pricing, no matter how mathematically convenient, the automatic adoption of this objective function (maximising profit) is not acceptable.

Secondly, Leontief's (1986) input–output analysis framework provided a basis for determining 'shadow pricing' from the physical input–output relationships in the economy. Price is determined using a 'dual formulation' of the standard Leontief model:

$$(\mathbf{I} - \mathbf{A}^{t})\mathbf{p} = \mathbf{v} \tag{2}$$

Where, **I** is the identity matrix ($q \times q$) measured in physical units; **A** is the technical coefficients matrix ($q \times q$) of q inputs into q sectors measured as physical inputs per physical unit of output; **p** is the price vector ($q \times 1$) measured in terms of \$/physical units; **v** is the value added vector ($q \times 1$) measured in monetary units (\$).

Strictly speaking, this is not a purely physical model as the vector **v** is enumerated in monetary terms (\$). Notwithstanding this point, the Leontief model is a 'cost of production' model of price determination, rather than one based on 'subjective preference'.

The Sraffa (1960) model, on the other hand, is purely a physical input–output model, and in addition it has a direct lineage with the classical cost of production models of price determination. In fact, the very motivation for the Sraffa model was to determine an analytical solution to Ricardo's quest to find an 'invariable standard of value'. Several contemporary ecological economists (England, 1986; Judson, 1989; O'Connor, 1996; Perrings, 1987) have advocated using the Sraffa (1960) model to determine 'ecological prices'—that is, prices that not only reflect the input–output flow relationships in the economy, but also reflect the input–output relationships of the ecological flows that support the economy. Patterson (1998a,b), however, argues that the direct translation of the Sraffa (1960) price determination model to the ecological context is inappropriate, as the Sraffa model violates fundamental biophysical principles: (i) the Sraffa model does not map physical flows of energy and mass, even though the inputs and outputs are measured in physical terms. The Sraffa processes are exchange processes; (ii) the Sraffa model does not explicitly conform to the principles of mass and energy conservation (First Law of Thermodynamics); (iii) the Sraffa surplus economy model violates both the First and Second Laws of Thermodynamics—something (the surplus) is produced from nothing; (iv) the Sraffa model is based on the circular flow of exchange value, rather than the ecological economics model of linear throughput of mass and energy. A circular flow without external inputs is a biophysical impossibility (i.e. a perpetual motion machine).

In summary, there is a heritage in economics of determining 'prices' on the basis of physical input–output relationships from the classical period through to the modern era, where the price determination procedures were formalised in terms of linear models (von Neumann, 1945; Sraffa, 1960; Leontief, 1986). Contemporary orthodox economists have, by and large, been dismissive of such approaches, instead preferring the 'subjective preference' methods. However, despite the misgivings of orthodox economists, there is a rich history of the use of 'cost of production' methods of pricing and valuation in economics, which have direct links to the ecological pricing methodology used in this paper.

3.2. Ecological and physical foundations

Ecological pricing approaches have also been developed by ecologists and physical scientists, sometimes with little or no knowledge of parallel attempts in economics. In ecology and in the physical sciences, problems arise when the analyst moves beyond mass and energy accounting to evaluation. For instance, in deciding what is the most efficient energy transformation process, the problem arises when comparing inputs and outputs that are in different units. How do we compare a joule of coal and a joule of solar energy, when quite clearly they have different energy qualities? Or more broadly speaking, how do we compare a tonne of biomass with a megalitre of atmospheric water. This is called the 'mixed units' problem or commensuration problem, which always arises in efficiency and evaluation analysis.

In order to resolve this problem, Ecologists and physical scientists proposed a 'pricing' system based on an 'embodied energy theory of value'. This controversial proposal was widely debated in the 1970s. The main advocates for an 'embodied energy theory of value' included Odum (1971), Odum and Odum (1976), Odum (1983), Slesser (1973) and Gilliland (1975), although the idea can be traced back as far as Nobel prize winner Soddy (1912). It is essentially a theory of value based on thermodynamics and systems thinking, where energy is considered to be the primary input that drives all economic and ecological systems.

The transformation of energy in all of these systems is perceived to be necessarily linear and unidirectional due to the entropy law. Energy quality is always degraded in economic and ecological processes, and therefore energy cannot be 'reused' to obtain the same amount of useful output. In ecological systems, solar energy is degraded and this transformation 'drives' the circulation of mass. For this reason, a number of ecologists consider solar energy to be the appropriate numeraire in their formulation of an energy theory value.

Odum (1983, 1996) has provided the most holistic expression of the energy theory of value perspective. His system models explicitly demonstrate how the countercurrent flows of energy and money provide the bases for all economic activity. Using this theoretical model, Odum (1996) then derives a series of 'tranformities' or 'quality factors' that measure the ecological prices, in energy terms, of various commodities in the economy. Patterson (1983, 1993, 1996) also developed a method for calculating 'energy quality' factors in complex systems of energy transformation by solving a system of simultaneous equations. Odum (1996), however, goes even further towards establishing an energy theory of value by invoking Lotka's (1922, 1925) Maximum Power Principle. Under this principle, Odum argues the processes or systems that obtain and use energy most efficiently are the most valuable as they will out-compete other processes in the long run.

The energy theory of value has been widely criticized, in particular, by neoclassical economists

(Heuttner, 1976) on the basis of methodological problems, such as the mixed units problem, but also on philosophical grounds for attempting to define value independent of consumer preferences. This last criticism is axiomatic rather than substantive, as the stated purpose of energy analysis has been to establish a biophysical theory of value not governed by social preferences.

Costanza and Neill (1981a) as well as Costanza and Hannon (1989) provided a more rigorous mathematical basis for calculating energy-based 'ecological prices', using input–output analysis and linear programming approaches similar to those used in economics. These 'ecological prices' used solar energy as the numeraire, based on the argument that solar energy was the only net input into the biosphere. A number of methodological problems emerged from these attempts which have been discussed in this paper.

Amir (1989) attempted to situate these ecological and biophysical approaches to 'pricing' within the broader context of thermodynamics, and to make linkages to generalised production linear-activity type models in economics (von Neumann, 1945; Koopmans, 1951; Malinvaud, 1953; Gale, 1960).

4. Ecological pricing applied to the valuation of the biosphere (1994)

The ecological pricing method described in Section 4.2 is used to value processes and quantities (commodities) in the biosphere for the year 1994. The biosphere system includes natural biogeochemical cycles and energy flows through terrestrial, oceanic, geological and atmospheric processes.

4.1. Construction of the biosphere flow matrix

The flow matrix **V**–**W** quantifies the flow of mass and energy through the biosphere system in 1994. In this analysis it is a 16×16 matrix, consisting of 16 quantities and 16 processes (refer to Table 1). The flow matrix **V**–**W** consists of an outputs matrix **V** minus an inputs matrix **W**. The matrices **V**, **W** and **V**–**W** consist of m rows representing processes and n columns representing quantities. Each row in the flow matrix **V**–**W** is a process with negative entries representing inputs from the process and positive entries representing outputs of the process.

4.1.1. Quantities

The quantities (columns) included in the flow matrix are:

Biomass (Columns 1, 2). Biomass is divided into two categories: 'Agricultural and Forestry Biomass' (appropriated by human activity) and 'Natural Biomass' (not appropriated by human activities). Biomass is measured in terms of Pg Carbon or Pg dry weight.

Nutrients (Columns 3–6). The four main macronutrients included in the flow matrix are: nitrogen (TgN), carbon (PgC), phosphorus (TgP) and sulphur (TgS). These quantities do not include the nutrients contained in the biomass, as this would constitute double-counting.

Water (Columns 7, 8). Water is subdivided into atmospheric water and surface water. Surface water includes water contained in terrestrial water bodies (lakes, rivers) and in the oceans. The water quantities are measured in terms of Pg hydrogen or PgH_2O.

Energy (Columns 9–12). Energy quantities include Solar Energy, Fossil Fuels, Geothermal Energy and Useful Energy. Useful energy is the amount of energy that is utilised at the actual point of end-use once 'losses' have been taken into account—e.g. it is the amount of useful light energy (15%) that flows from an electric light bulb, not the electricity (100%) used by the light bulb. Energy is measured in terms of Exajoules (EJ).

Minerals (Columns 13–15). Minerals include Virgin Minerals located on or under the earth's surface, Mined Minerals (except Uranium) and Mined Uranium. Virgin Minerals and Mined Minerals are measured in Pg of total mass, and Uranium is measured in terms of Gg.

Final Economic Product (Column 16). This is the final economic output of the world economy. It includes household and government consumption of products and (manufactured) capital formation. It can be measured in physical terms (Pg) or in monetary terms ($US trillion of Value Added).

Table 1
Flow matrix (**V**–**W**) for the biosphere, 1994

Processes	Quantities															
	Biomass[g]		Nutrients[b]				Water		Energy				Minerals			Final[c] Economic Product $\$US10^{12}$
	Agri and Forest PgC	Natural PgC	Nitrogen TgN	Carbon PgC	Phosphorus TgP	Sulphur TgS	Atmosphere PgH	Surface PgH	Solar EJ	Fossil fuels EJ	Geothermal EJ	Useful Energy EJ	Uranium Gg	Virgin Minerals Pg	Mined Minerals & Fertiliser Pg	
Terres. Primary production		115.74	−987.06	−44.38	−106.56	−606.25	2000.00	−2220.00	−49399.85							
Terres. Consumption		−2.67	25.23	1.28		6.14										
Soil and terres. Processes		−114.60	1177.95	45.10	196.64	734.71	2160.00	−2960.00	−66462.75							
Surface ocean		28.99	−617.55	−28.99	−206.02	−213.28	41841.63	−40821.11	−967029.77							
Deep ocean		−26.33	510.71	26.33	133.55	142.26										
Fossil fuel formation			6.06	−0.51	0.03	−40.06				4.22						
Atmosphere							−47080.87	47080.87								
Natural gas use			0.60	2.60		1.09	0.28			−72.08		43.25				
Crude oil use			1.17	1.17		3.97	0.10			−139.22		55.69				
Hydroelectricity use							446.40	−446.40				6.83				
Geothermal energy use											−5.20	0.94				
Coal and wood use		−1.13	2.60	2.71		43.58	0.12			−90.45		29.68				
Uranium use												7.34	−31.97			
Mineral and fert sector			−80.00	−0.22	−12.00	−26.00						−1.66		−7.03	6.57	
Agri and forestry sector	15.26		−60.06	−6.00	−8.65	−58.59	523.39	−524.41	−38822.33			−2.78			−0.66	−1.73
Other economic sectors	−15.26		20.35	0.91	3.00	12.42	108.95	−108.95	−2043.28			−139.28			−5.91	27.53
Net input and output[d]									−1123757.88	−297.52	−5.20		−31.97	−7.03		25.79

[a] The conversion factor is $\times 2.5409$ to convert from PgC to Pg (dry weight) for Agricultural & Biomass; and $\times 2.4795$ to convert from PgC to Pg (dry weight) for Natural Biomass. [b] The conversion factor is $\times 18.0153$ to convert from Pg H to Pg H_2O. [c] The conversion factor is $\times 1.6233$ to convert from $\$US_{1994}$million to Pg (mass of Final Economic Product). [d] Column totals may not add up due to rounding.

4.1.2. Processes

The processes (rows) included in the flow matrix are:

Terrestrial Primary Production (Row n1). This is the net production of biomass by primary producers (autotrophs) in terrestrial systems. It does not include agricultural and forestry biomass production.

Terrestrial Consumption (Row n2). This is the consumption of natural biomass by terrestrial herbivores, carnivores and other heterotrophs, but not including decomposers.

Soil and Terrestrial Processes (Row n3). This primarily covers soil processes including the breakdown of biomass, but also other terrestrial processes such as the flow of water across the earth's surface.

Surface Ocean (Row n4). This includes all oceanic activities occurring in the euphotic zone, including net primary production, as well as abiotic processes, such as the inception of atmospheric and terrestrial compounds, and ocean hydrology.

Deep Ocean (Row n5). This includes all oceanic activities below the euphotic zone, including the downwelling and sedimentation of nutrients and products from the surface ocean.

Fossil Fuel Formation (Row n6). This includes the slow formation of oil shales, coal, natural gas and crude oil by geological processes that require large inputs of heat and pressure over geological time.

Atmosphere (Row n7). For simplicity in this study, this only includes the atmospheric processing of water, and not the chemical transformation of compounds such as oxidation of CH_4 to CO.

Energy Sector Processes (Rows a1–a6). These processes involve the transformation of primary energy to end-use energy. These energy processes include crude oil use, natural gas use, coal and wood use, use of water for hydroelectricity, geothermal energy use and uranium energy use.

Minerals and Fertilizer Sector (Row a7). This involves the mining of minerals for commercial use and the initial processing of these minerals, such as in fertilizer manufacture.

Agricultural and Forestry Sector (Row a8). This involves the production of agricultural and forestry products. It does not include the further processing of these products.

Other Economic Sectors (Row a9). This involves the secondary (manufacturing) and tertiary (services) sectors of the economy, as well as the direct final sales of the above sectors.

4.1.3. Data sources

The construction of the flow matrix V-W involved reconciling data collected from a wide variety of sources. The basic approach was to construct a matrix (120 processes $\times$ 80 quantities) of the flow of energy and mass through the biosphere. A mass and energy balance was determined for each process. This initial matrix was then aggregated to form the 16×16 flow matrix used in this paper (refer to Table 1). The main sources of data used include: Economic Output including Global GDP (World Bank, 1996; United Nations, 1993); Biogeochemical Cycles and Nutrient Flows (Ayres, 1996; Bowen, 1979; den Elzen et al., 1995; Schlesinger, 1991; Butcher et al., 1992; Wigley and Schimel, 2000); Water Flows and Usage (Bowen, 1979; Postel et al., 1996; Shiklomanov, 1993); Solar and Natural Energy Flows (Hubbert, 1963; Odum, 1996, 2000; Odum et al., 2000; Oort, 1970); Human Energy Use (Energy Efficiency and Conservation Authority, 1997; International Energy Agency, 1987, 1997; British Petroleum, 1996; Stout, 1990); and the Use of Minerals and Fertilisers (United Nations, 1996; World Resources Institute, 1996).

4.2. Determination of ecological prices of biosphere commodities

The ecological price of each commodity in the biosphere model can be determined by solving the following system of simultaneous equations:

$$\mathbf{W}\boldsymbol{\beta} + \mathbf{e} = \mathbf{V}\boldsymbol{\beta} \tag{3}$$

$$\boldsymbol{\beta} \geq 0$$

where $\mathbf{W}$ is a matrix ($m \times n$) representing n inputs into m processes in the biosphere system, measured in physical units; $\mathbf{V}$ is a matrix ($m \times n$) representing n outputs into m processes in the biosphere system, measured in physical units; $\boldsymbol{\beta}$ is the price vector ($n \times 1$); **e** is the residuals vector ($m \times 1$).

The constrained least squares regression method developed by Lawson and Hanson (1974) was used to solve these equations. The 16×16 matrix system is overdetermined by one degree of freedom, once one of the variables is made the numeraire (set to unity).

By using $US billion as the numeraire, the following prices are obtained for the biosphere system:

β_1 1 $US billion/1$US billion Final economic products
β_2 236.01 $US billion/1PgC Agricultural and forestry biomass
β_3 124.25 $US billion/PgC Natural biomass
β_4 0.19 $US billion/TgN of Nitrogen compounds
β_5 0.00 $US billion/PgC of Carbon compounds
β_6 0.00 $US billion/TgP of Phosphorus compounds
β_7 22.28 $US billion/TgS of Sulphur compounds
β_8 2.80 $US billion/PgH of Atmospheric water
β_9 2.80 $US billion/PgH of Surface water
β_{10} 69.88 $US billion/EJ of Fossil fuels
β_{11} 0.0016 $US billion/EJ of Solar energy
β_{12} 29.18 $US billion/EJ of Geothermal energy
β_{13} 37.24 $US billion/EJ of Uranium
β_{14} 0.00 $US billion/Pg Virgin minerals
β_{15} 131.51 $US billion/Pg Mined minerals
β_{16} 162.11 $US billion/EJ of Useful energy

These ecological prices can alternatively be expressed in terms of Fossil Fuel Equivalents:

β_1 0.0143 Fossil fuels (EJ)/$US billion Final economic products
β_2 3.3774 Fossil fuels (EJ)/PgC Agricultural and forestry biomass
β_3 1.7780 Fossil fuels (EJ)/PgC Natural biomass
β_4 0.0027 Fossil fuels (EJ)/TgN of Nitrogen compounds
β_5 0.000 Fossil fuels (EJ)/PgC of Carbon compounds
β_6 0.000 Fossil fuels (EJ)/TgP of Phosphorus compounds
β_7 0.3188 Fossil fuels (EJ)/TgS of Sulphur compounds
β_8 0.0401 Fossil fuels (EJ)/PgH of Atmospheric water
β_9 0.0401 Fossil fuels (EJ)/PgH of Surface water
β_{10} 1.0000 Fossil fuels (EJ)/EJ of Fossil fuels
β_{11} 0.000024 Fossil fuels (EJ)/EJ of Solar energy
β_{12} 0.4176 Fossil fuels (EJ)/EJ of Geothermal energy
β_{13} 0.5329 Fossil fuels (EJ)/EJ of Uranium
β_{14} 0.0000 Fossil fuels (EJ)/Pg Virgin minerals
β_{15} 1.8819 Fossil fuels (EJ)/Pg Mined minerals
β_{16} 2.3197 Fossil fuels (EJ)/EJ of Useful energy

It is important to note that it is arbitrary which commodity (quantity) is used as the numeraire. That is, the relative prices remain constant—e.g. the ratio between β_3 and β_1 is the same, irrespective if fossil fuels or if $US billion is used as the numeraire. In this context, the debates over whether energy or money is the appropriate numeraire (Soddy, 1912; Edwards, 1976; Hall et al., 1992) are rendered irrelevant because in ecological pricing either can be used, and the same price relativities result.

4.3. Ecological prices as indicators of energy and material quality

Earlier applications of the ecological pricing method (Costanza and Neill, 1981a,b; Patterson, 1983) focused on using ecological prices as an indicator of energy quality. It is now clear that the ecological pricing approach also provides a framework for measuring material quality as well as energy quality which accommodates Georgescu-Roegen's (1979) protestation that 'matter matters too'. The energy quality factors determined by solving both the 16×16 and 120×80 biosphere matrix are summarised by

Table 2
Energy quality factors as measured by ecological prices[a]

Energy forms	Ecological prices	
	Fossil fuel equivalents (EJ) per exajoule of energy	$US Billion per exajoule of energy
Derived from 16×16 *matrix*		
Useful energy	2.32	162.11
Fossil fuel	1.00	69.88
Geothermal	0.42	29.18
Solar[b]	0.00002	0.0016
Derived from 120×80 *matrix*		
Electricity	1.76	122.70
Petroleum products	1.10	76.93
Crude oil	1.03	71.78
Gas	1.07	74.66
Coal	0.78	54.63
Wood	0.36	25.11
Geothermal	0.42	29.18
Solar[b]	0.0002	0.0018
Useful heat end-use[c]	1.64	114.23
Transport end-use[c]	6.92	483.39
Obligatory electricity end-use[c]	2.41	168.08
Reduced fe ores[c]	0.96	67.22
Reduced non fe ores[c]	2.03	141.80

[a] These energy quality factors are based on the idea of 'embodied energy'. These factors should not be confused with energy quality factors based on the 'work potential' of the energy source–e.g. in exergy analysis, exergy/enthalpy is used as an energy quality indicator. Refer to Patterson (1993) for a full discussion of energy quality indicators.

[b] The solar energy ecological prices do not exactly correspond to aggregation error.

[c] This is the 'end-use' energy portion. It is the useful energy actually produced by the end-use device, once losses have been deducted—e.g. only about 15% of the electricity used by a light bulb is converted to useful end-use in the form of photons, with the remainder being converted to waste heat.

Table 3
Material quality factors as measured by ecological prices

Material forms	Ecological prices	
	Fossil fuel equivalents (GJ) per tonne of material	$US per tonne of material
Uranium	532 903.82	37 239 930.22
Sulphur compounds	146.61	10 245.42
Fossil fuels	39.68	2 773.01
Final economic product	8.82	616.02
Mined minerals	1.88	131.51
Agricultural and forest biomass	1.33	92.88
Nitrogen compounds	1.26	88.11
Natural biomass	0.72	50.11
Atmospheric water	0.002	0.16
Surface water	0.002	0.16

Tables 2 and 3. These energy quality factors are similar to those obtained by Odum (1983, 1996) even though he uses different (non-algebraic) methods for calculating them.[5] Electricity is the highest quality energy form because more energy is required to produce electricity (backward linkages) and because electricity is more efficient in being converted to useful end-use energy (forward linkages). Whereas solar energy is the lowest quality form of energy, as it is less efficiently converted (than other energy sources) to useful end-use energy.[6]

Energy quality in the context of ecological pricing is measured in terms of a ratio of value (numerator) per total heat content of the energy form (denominator). Material quality is

[5] It should be noted that Odum (1983, 1996) calculated his energy quality factors (transformities) on the basis of backward linkages, whereas the ecological pricing method used in this paper measures both backward and forward linkages. There are, however, some exceptions in Odum's (1996) method, where he is 'forced' to use forward linkages for primary inputs such as tidal energy and deep heat energy. This is because these primary energy inputs cannot be traced back to solar energy inputs.

[6] It should be noted that previously analysts have used 'embodied energy' (backward linkages only) as a measure of energy quality (e.g. Odum, 1983). The ecological pricing method uses both forward and backward linkages to calculate energy quality factors.

Table 4
Comparison ecological prices (from the biosphere model) with market prices, 1994

Commodity	Ecological price ($US per tonne)	Market price ($US per tonne)	Data sources and calculation details
Uranium	37 239 930	36 376 273	Australian, Canadian and Nigerian prices for 1995, (United Nations, 2000)
Mined minerals and fertilizers	131	170	*Market prices* of minerals and fertilizers (United Nations, 2000) *Quantities* for the calculation of the weighted mean price (United Nations, 1996; World Resources Institute, 1996)
Surface water	0.16	0.21 (Agriculture) 0.79 (Domestic) 1.23 (Industry) 0.46 (Weighted mean)	*Market prices* (OECD, 1998; Dinar, 2000) *Quantities* for the calculation of the weighted mean price (Postel et al., 1996)
Fossil fuel energy	2773 ($US 70/GJ)	397 ($US 10/GJ)	*Market prices* (International Energy Agency, 1994)
Agricultural and forest biomass	93 (Dry weight)	313 (Rice) 121 (Lumber)	Rice, average of Thai (two types) and Vietnamese export prices for 1994 (United Nations, 2000) Lumber, US futures market 1994 price (United Nations, 2000)

analogous, being the value (numerator) per total mass (denominator). On this basis, there is a vast difference in the material quality factors ranging from Uranium ($US 37 239 930 per tonne), to Sulphur Compounds ($US 10 245 per tonne), Fossil Fuels ($US 2 773 per tonne), Final Economic Product ($US 616 per tonne), Mined Minerals ($US 131.5 per tonne), Agricultural and Forest Biomass ($US 93 per tonne), Nitrogen compounds ($US 88 per tonne), Natural Biomass ($US 50 per tonne) and finally to atmospheric surface water ($ 0.16 per tonne). The high material quality of uranium is due to its forward linkage into the energy sector, which is highly productive in terms of producing useful energy from a relatively small input of mass. On the other hand, the material quality factor of Final Economic Product is almost entirely determined by backward linkages in the economy, with only a small amount of Final Economic Product being fed back to the Agricultural and Forestry Sector.

4.4. Ecological prices and market prices

The ecological prices (derived from the biosphere model) can be compared with actual market prices only for those commodities that have a market. One might expect the ecological prices and market prices, not necessarily to be the same or similar, but at least to be of the 'same order of magnitude' if the biosphere-pricing model is to be valid and reliable. In fact, the mainstream interpretation would probably be that the ecological prices are 'shadow prices', being defined by how well the various commodities (inputs and outputs) contribute to Value Added (column 16) in the biosphere matrix. Another interpretation is, if the model has good explanatory power, then the ecological prices and market prices should correspond. This approach is somewhat reminiscent of Ricardo's attempt to 'prove' that labour costs explain variations in market prices. Caution, however, needs to be displayed in making either of these interpretations, as ecological prices are solely determined by biophysical interdependencies, which although being important in defining market prices, they are by no means the only factors.

The ecological and market prices (for the five commodities for which there are markets) are compared in Table 4. There is a good correspondence between the two sets of prices, and certainly the 'ecological prices' determined by the model are in the 'right' order of magnitude. The uranium prices are remarkably almost identical (ecological price of \$US 37.2 billion/tonne c.f. a market price of \$US 36.4 billion/tonne). The ecological prices for Mined Minerals and Fertilizers, Surface Water and Agriculture and Forestry Biomass are below the market prices. This may be because it is very difficult to obtain a market price for these 'raw materials' which does not include at least some component of Value Added resulting from further processing or added services—hence the market prices for these commodities in Tables 4 and 5 may be slightly inflated. The most significant deviation between the ecological prices and market prices is for fossil fuels where the ecological price is seven times higher than the market price. This is suggesting that the market price does not reflect the 'true' value of fossil fuels in terms of how well fossil fuels contribute to economic production (Value Added) in the biosphere model. This finding, although apparently surprising, is consistent with the findings of at least three other econometric studies. Kummel and Lindenberger (1998, 2000) determined elasticities of production of 0.50, 0.45 and 0.54 for Germany, Japan and USA, respectively, implying that energy inputs attributed for between 45 and 54% of the economic production, but only had a total factor cost (\$) of about 5%. This essentially confirms the findings of an earlier study of the United States economy by Berndt and Wood (1987). Similarly, Patterson (1989) found that over the 1960–1984 period energy inputs accounted for 21.83% of GDP growth in New Zealand, but the energy cost share of GDP was only 6–7%. However, it still remains largely unexplained as to why uranium energy's ecological and market prices are very similar, whereas for fossil fuels there is a sevenfold difference.

Table 5
Ecological efficiencies of biosphere processes, 1994

Processes	Ecological efficiencies $\mathbf{\Phi}$
Terrestrial primary production	1.00
Terrestrial consumption	0.43
Soil and other terrestrial processes	1.00
Surface ocean	1.00
Deep ocean	1.00
Fossil fuels formation	0.33
Atmosphere	1.00
Natural gas use	1.40
Crude oil use	0.94
Hydroelectricity use	1.89
Geothermal energy use	1.00
Coal and wood use	0.90
Uranium use	1.00
Minerals and fertilizer sector	1.00
Agricultural and forestry sector	1.00
Other economic sectors	1.00

Ecological efficiency $\mathbf{\Phi}$ is the: total value of process outputs (price × quantity) *divided* by the total value of process inputs (price × quantity). Any quantity (solar energy, \$ trillion) can be used as the numeraire in the prices, and the same ecological efficiency results.

4.5. Ecological efficiencies of biosphere processes

Having determined the ecological prices $\boldsymbol{\beta}$, the ecological efficiency vector Φ can be calculated. This involves multiplying all input and output quantities by their appropriate prices $\boldsymbol{\beta}$, to obtain the total value of the inputs $\boldsymbol{\pi}$ and the total value of the outputs $\boldsymbol{\rho}$:

$$\mathbf{W}\boldsymbol{\beta} = \boldsymbol{\pi} \quad (4)$$

$$\mathbf{V}\boldsymbol{\beta} = \boldsymbol{\rho} \quad (5)$$

The elements in the gross outputs vector $\boldsymbol{\rho}$ are then divided by the corresponding elements in the gross inputs vector $\boldsymbol{\pi}$, to obtain the ecological efficiency of each process which is represented by vector $\mathbf{\Phi}$.

In 'equilibrium' models, all processes by definition have the same efficiency, which is not the case here in the biosphere model as the equations are inconsistent which implies different efficiencies and 'non-equilibrium' ecological prices.[7]

In terms of ecological efficiencies, the most instructive information that can be extracted from solving the 16×16 matrix concerns the energy conversion processes:

[7] The assumption in such models (von Neumann, Sraffa) is that over time 'inefficient' processes will be 'out competed', ultimately leaving an ensemble of processes, all of which have the same profit rate (efficiency) and hence have converged at an equillibrium point.

Hydroelectricity use $\Phi_{a3} = 1.89$
(falling water → useful end-use energy)
Natural gas use $\Phi_{a1} = 1.40$
(natural gas → useful end-use energy)
Uranium energy use $\Phi_{a6} = 1.00$
(uranium → useful end-use energy)
Geothermal energy use $\Phi_{a4} = 1.00$
(geothermal energy → useful end-use energy)
Crude oil use $\Phi_{a2} = 0.94$
(crude oil → useful end-use energy)
Coal and wood use
(natural biomass, coal → useful end-use energy) $\Phi_{a5} = 0.90$

Hydroelectricity use is more efficient than the fossil fuel processes, essentially because solar energy is transformed very efficiently through the hydrological cycle to electricity. This is significantly more efficient than transforming solar energy through the formation and then combustion of fossil fuels to electricity. In this case, the variation in relative efficiencies of natural gas use ($\Phi_{a1} = 1.40$), crude oil use ($\Phi_{a2} = 0.94$), and coal and wood use ($\Phi_{a5} = 0.90$) is solely due to the different conversion efficiencies from the primary fuel to useful end-use output. Coal, for example, is inherently a lower quality (less efficient) fuel than natural gas, as it can be converted to less useful output per amount of energy input (Patterson, 1993). The efficiencies for the two remaining energy processes (Geothermal and Uranium) are $\Phi = 1$, solely because they are 'non-basic' processes in the Sraffian sense—i.e. there are no other 'competitive processes', as there is only one process that uses geothermal energy and only one process that uses uranium energy. All other processes in this biosphere model (except process $n2$) have an ecological efficiency of $\Phi = 1$ for the same reason. With the disaggregation of the model to 120×80, more process efficiencies will become evident which should prove useful in terms of informing policy analysis.

4.6. Total and net value of biosphere processes

The total value of the inputs and outputs from each biosphere process can be calculated by multiplying the column elements in the matrix $\mathbf{V}-\mathbf{W}$ by the appropriate ecological price $\boldsymbol{\beta}$. The resultant matrix describes the flow of *total value* (*price × quantity*) through each of the interconnected biospheric processes—refer to Table 6. As an accounting device, this matrix is useful as it measures the transfer of value through the biosphere by way of input–output bookkeeping, hence avoiding 'double counting' which was one of the criticisms of the Costanza et al. (1997) analysis of the value of global ecosystem services.

By summing the columns in the total value matrix, the net inputs and net outputs of the biosphere are determined. These net inputs (Solar Energy, Geothermal Energy, Uranium Energy, Fossil Fuel Energy) are transformed through biogeochemical cycles and other processes to the net output (Final Economic Output). In a sense, the net inputs are the *primary ecological inputs* into the biosphere and the other flows represented by the total value matrix are intermediate inputs and outputs. As many of these intermediate flows are cyclical in nature they lead to no net output, but are nevertheless very important and sizable flows—e.g. the hydrological cycle flows lead to high gross output value due to the vast amount of solar energy required to drive them and the large volumes of water involved, but they lead to no net output from the biosphere. The *Nature* paper by Costanza et al. (1997) focused on these intermediate ecosystem flows and services.

The importance or activity of these intermediate ecosystem processes can be assessed by examining their gross output (refer to Table 7). The gross output of the surface ocean ($US 124.54 trillion) and the atmosphere ($US 135.94 trillion) is very high, which is mainly due to the role that these two systems play in the hydrological cycle. Of the natural processes, next were Terrestrial Primary Production ($US 20.61 trillion) and Soil and Other Terrestrial Processes ($US 23.35 trillion), both having similar gross outputs as they are sequential parts of the terrestrial trophic chain. All other natural processes had relatively small gross outputs, indicating low levels of activity. Of the human-appropriated processes, the Other Economic Sectors ($US 28.13 trillion) and the Energy Sector processes ($US 26.45 trillion) had the highest gross output.

Table 6
Total value matrix ($US trillion) for the biosphere, 1994

Processes	Quantities												
	Biomass		Nutrients		Water		Energy				Minerals		Final economic product
	Agri and forest	Natural	Nitrogen	Sulphur	Atmosphere	Surface	Solar	Fossil fuels	Geothermal	Useful energy	Uranium	Mined minerals and fertiliser	
Terres. Primary production		14.84	−0.19	−13.94	5.78	−6.41	−0.08						−1.73
Terres. Consumption		−0.34		0.14									
Soil and terres. Processes		−14.69	0.23	16.89	6.24	−8.55	−0.11						
Surface ocean		3.72	−0.12	−4.90	120.82	−117.88	−1.63						
Deep ocean		−3.38	0.10	3.27									
Fossil fuel formation				−0.92				0.30					
Atmosphere					−135.95	135.96							
Natural gas use				0.03				−5.20		7.23			
Crude oil use				0.09				−10.04		9.31			
Hydroelectricity use					1.29	−1.29				1.14			
Geothermal energy use									−0.16	0.16			
Coal and wood use		-0.15		1.00				−6.52		4.96			
Uranium use										1.23	−1.23		
Mineral and fert sector			−0.02	−0.60						−0.28		0.89	
Agri and forestry sector		3.72		−0.01	−1.35	1.51	−1.51	−0.07			−0.47		−0.09
Other economic sectors	−3.72			0.29	0.31	−0.31				−23.30		−0.80	27.53
Net input and output[a]							−1.90	−21.45	−0.16		−1.23		25.79

[a] The net value vector is the column totals of the total value matrix.

Table 7
Total output of biosphere processes, 1994 ($US trillion)

Process	Total output ($US trillion)
Terrestrial primary production	20.61
Terrestrial consumption	0.15
Soil & other terrestrial processes	23.35
Surface ocean	124.54
Deep ocean	3.37
Fossil fuel formation	0.31
Atmosphere	135.96
Natural gas use	7.26
Crude oil use	9.41
Hydroelectricity use	2.43
Geothermal energy use	0.16
Coal and wood use	5.97
Uranium use	1.23
Minerals and fertilizer sector	0.89
Agricultural and forestry sector	5.23
Other economic sectors	28.13

The Costanza et al. (1997) analysis concluded that global ecosystem services had a total value of 1.32 times the value of goods and services produced by the global economy.[8] Several commentators argued that this ratio of 'value of ecosystem services:GDP' was too high, as on theoretical grounds it should not exceed one (Ayres, 1998; Pearce, 1998). Interestingly, the biosphere pricing model presented in this paper, shows that the value of primary ecological inputs is $24.73 trillion, compared with $25.79 trillion for global GDP, giving a ratio of 0.98.

5. Discussion

A recurrent theme in the ecological economics literature is the call for valuation and pricing methods that are more biophysical/biocentric to provide a counterbalance to the anthropocentric methods often used in valuation and cost-benefit studies (Cleveland et al., 1984; Costanza et al., 1991; Proops, 2000). Ecological pricing as developed in this paper is seen as a constructive move in this direction, being based on measuring *biophysical interdependencies* (or contributory values) implicit in the global ecological system and its economic sub-system. Although some of these biophysical interdependencies are clearly a result of human interventions and hence indirectly reflect human preferences, ecological pricing does however tend to highlight species and ecological functions that may not usually be detected by other valuation methods such as WTP (Costanza, 1991). For example, it is unlikely that the value of protozoa in the ecosystem would be measured in a WTP survey, whereas in ecological pricing the value of these protozoa would be taken account of by the forward and backward linkages they have with other components of the ecosystem. In this sense, ecological pricing is more 'biocentric' than neoclassical valuation methods that tend to have a more anthropocentric emphasis (Hannon, 1998).

In the spirit of methodological pluralism, ecological pricing is not seen as a replacement for neoclassical valuation, or for that matter, other valuation methods. Rather, it is seen as a complementary approach not to be used instead of the other methods, but alongside other methods. Or indeed ecological pricing could be used as a component of a multicriteria framework or methodological schema such as that proposed by Lockwood (1997).

Although the ecological pricing method developed in this paper has a number of antecedents and commonalities with methods used both in economics and ecology, it is unique in a number of ways. Firstly, it is based on a non-equilibrium model of pricing, insofar that there is no *a priori* assumption that the system is (or should be) at equilibrium as do the pricing models of von Neumann (1945), Sraffa (1960), Leontief (1986), and Costanza and Hannon (1989). That is, particularly in very disaggregrated models where there are more processes than quantities, it is almost inevitable that we will be dealing with inconsistent equations, which means that processes will have differing efficiencies. Equilibrium conditions are just an artifact of solving a square matrix ($m = n$) of equations—or of

[8] The original Costanza et al. (1997) analysis estimated the value of global ecosystem services to be $33 trillion for 1994 (c.f. $18 trillion for global GDP), giving a ratio of 1.83. Subsequently, it has been estimated that the value of global GDP for 1994 was about $25 trillion (World Bank, 1996), giving a ratio of 1.32 which is used above.

solving a matrix that has been reduced to a 'square' form by using an optimization method. Secondly, it is clear that it is not necessary that solar energy be used as the numeraire, as assumed in previous pricing models used in ecology. Any quantity (commodity) can be used as the numeraire and the relative prices remain constant, as do other system parameters, such as the ecological efficiencies of processes. Indeed, the Economic Output quantity ($ value added) can be used as the numeraire if so desired—this may be imperative in terms of 'communicating' the results of any ecological pricing exercise. Thirdly, and related to the last point, the ecological prices can be expressed in terms of monetary units, solar energy units (Emergy), or land equivalents (Ecological Footprint). All of these numeraires essentially give the same analytical result, but they communicate to different disciplinary audiences. Expressing the results of any ecological pricing exercise in terms of these alternative numeraires, may well be a fruitful tactic in terms of encouraging communication across these disciplinary boundaries.

The main empirical finding of this paper is that the ratio of the 'value of primary ecological inputs to Global GDP' was found to be 0.98 for 1994. This compares with a ratio of 1.32 for the 'value of global ecosystems: global GDP' determined by Costanza et al. (1997) in their Nature paper for the same year. One reason, why our study arrived at a lower ratio, may be that the input–output accounting eliminates double counting, by tracing the flow of value through the system to arrive at a net figure. In the Costanza et al. (1997) study it is possible that some of the ecosystem services may be overlapping, or one ecosystem service may contribute to another ecosystem service and hence be 'double-counted'. Another important empirical finding was that the ecological prices (with the exception of fossil fuels) show a good correspondence with actual market prices for those products that are traded on markets. The exception of fossil fuels is not surprising, even though the ecological price is about seven times the market price. This is because econometric studies have found that energy inputs contribute in the order of five-seven times more to GDP growth than their cost share indicates.

Acknowledgements

This work was conducted as part of the Working Group on the Value of the World's Ecosystem Services and Natural Capital; Toward a Dynamic, Integrated Approach supported by the National Center for Ecological Analysis and Synthesis, a Center funded by NSF (Grant # DEB-0072909), the University of California, and the Santa Barbara campus. Additional support was also provided for the Postdoctoral Associate, Matthew A. Wilson, in the Group. The discussions held with fellow 'Theory Group' members (Steve Farber, Matt Wilson, Karin Limburg, Bob O'Neill, Rudolph de Groot, Richard Howarth, Glen-Marie Lange) during our NCEAS Santa Barbara meetings in 1998 and 1999 proved particularly valuable as input into this paper. A special thanks is also due to Bob Costanza for his useful comments and suggestions concerning an initial draft of this paper. I am also grateful to two anonymous referees who provided detailed and helpful notes on the submitted draft of this paper.

The financial support from the New Zealand Foundation of Research Science and Technology (Contract C09X0007) is also acknowledged and much appreciated.

References

Amir, S., 1989. On the use of ecological prices and systems-wide indicators derived there from to quantify man's impact on the ecosystem. Ecological Economics 1, 203–231.

Amir, S., 1994. The role of thermodynamics in the study of economic and ecological systems. Ecological Economics 10, 125–142.

Ayres, R.U., 1996. Industrial Metabolism and the Grand Nutrient Cycles. INSEAD, Fontainebleau, France.

Ayres, R.U., 1998. The price–value paradox. Ecological Economics 25, 17–19.

Berndt, E.R., Wood, D.O., 1987. Energy price shocks and productivity growth: a survey. In: Gordon, R.L., Jacoby, H.D., Zimmerman, M.B. (Eds.), Energy: Markets and Regulation. MIT Press, Cambridge, MA, pp. 305–387.

Bowen, H.J.M., 1979. Environmental Chemistry of the Elements. Academic Press, London.

British Petroleum, 1996. BP Statistical Review of World Energy 1996. British Petroleum, London.

Butcher, S., Charlson, K.J., Orians, G.H., Wolfe, G.V., 1992. Global Biogeochemical Cycles. Academic Press, London.

Cleveland, C.J., Costanza, R., Hall, C.A.S., Kaufmann, R., 1984. Energy in the United States economy: a biophysical perspective. Science 225, 890–897.

Collins, D., Odum, H.T., 2001. Calculating transformities with an Eigenvector method. In: Brown, M.T. (Ed.), Emergy Synthesis: Theory and Applications of the Emergy Methodology. Center for Environmental Policy, University of Florida, Gainesville, pp. 265–279.

Costanza, R., 1991. Energy, uncertainty and ecological economics. In: Physical Chemistry. Proceedings of an International Workshop, Siena, Italy. Elsevier, Amsterdam, pp. 203–207.

Costanza, R., Neill, C., 1981a. The energy embodied in products of the biosphere. In: Mitsch, W.J., Boserman, R.W., Klopatek, J.M. (Eds.), Energy and Ecological Modelling. Elsevier, Amsterdam, pp. 745–755.

Costanza, R., Neill, C., 1981b. Energy embodied in products of ecological systems: a linear programming approach. In: Mitsch, W.J., Boserman, R.W., Klopatek, J.M. (Eds.), Energy and Ecological Modeling. Elsevier, Amsterdam.

Costanza, R., Neill, C., 1984. Energy intensities, interdependence and value in ecological systems: a linear programming approach. Journal of Theoretical Biology 106, 41–57.

Costanza, R., Hannon, B., 1989. Dealing with the mixed units problem in ecosystem network analysis. In: Wulff, F., Field, J.G., Mann, K.H. (Eds.), Network Analysis in Marine Ecology: Methods and Applications. Springer, Berlin, pp. 90–115.

Costanza, R., Neill, C., Leibowitz, S.G., Fruci, J., Bhar, L., Day, J.W., 1983. Ecological Models of the Mississippi Delta Plain Region: Data Collection and Presentation. Fish and Wild Life Services, US Department of Interior, Washington.

Costanza, R., Daly, H.E., Bartholomew, J.A., 1991. Goals, agenda and policy recommendations for ecological economics. In: Costanza, R. (Ed.), Ecological Economics: The Science and Management of Sustainability. Columbia University Press, New York, pp. 1–20.

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P., van der Belt, M., 1997. The value of the world's ecosystem services and natural capital. Nature 387 (15), 253–260.

Daly, H.E., 1985. The circular flow of exchange value and linear throughput of matter-energy: a case of misplaced concreteness. Review of Society and Economy XLII (3), 279–297.

DeAngelis, D.L., Waterhouse, J.C., 1987. Equilibrium and non equilibrium concepts in ecological models. Ecological Monographs 57, 1–21.

den Elzen, M., Beusen, A., Rotmans, J., 1995. Modelling Global Biogeochemical Cycles: An Integrated Approach. National Institute of Public Health and the Environment (RIVM). Bilthoven, The Netherlands.

Dinar, A., 2000. The Political Economy of Water Pricing Reforms. Oxford University Press, Oxford.

Edwards, G.W., 1976. Energy budgeting: joules or dollars. Australian Journal of Agricultural Economics 20, 179–191.

Energy Efficiency and Conservation Authority, 1997. Energy End-Use Database. EECA, Wellington, New Zealand.

England, R.W., 1986. Production, distribution and environmental quality: Mr Sraffa reinterpreted as an ecologist. Kyklos 39, 230–244.

Farber, S., Costanza, R., Wilson, M., 2002. Economic and Ecological Concepts for Valuing Ecosystem Services.

Fruci, J.R., Costanza, R., Leibowitz, S.G., 1983. Quantifying the interdependence between material and energy flows in ecosystems. In: Laurenroth, W.K., Skogerboe, G.V., Flug, M. (Eds.), Analysis of Ecological Systems: State-of-the-Art in Ecological Modelling. Elsevier, Amsterdam, pp. 241–250.

Gale, D., 1960. The Theory of Linear Economic Models. McGraw Hill, New York.

Georgescu-Roegen, N., 1979). Energy analysis and economic valuation. Southern Economy Journal 45, 1023–1058.

Gilliland, M.W., 1975. Energy analysis and public policy. Science 189, 1051–1056.

Gilliland, M., 1977. Energy analysis: a tool for evaluating the impact of end-use management strategies on economic growth. In: Fazzolare, R.A., Smith, C.B. (Eds.), Energy Use Management. Proceedings of the International Conference. Pergamon Press, New York, pp. 613–619.

Hall, C.A.S., Cleveland, C.J., Kaufmann, R., 1992. Energy and Resource Quality: The Ecology of the Economic Process. University Press of Colarado, Colorado.

Hannon, B., 1998. How might nature value man. Ecological Economics 25, 265–279.

Heuttner, D.A., 1976. Net energy analysis: an economic assessment. Science 192, 101–104.

Hubbert, M.K., 1963. Energy resources. A Report to the Committee on Natural Resources of the Natural Academy of Sciences-National Research Council, Washington, DC.

International Energy Agency, 1987. Electricity End-Use Efficiency. IEA, Paris.

International Energy Agency, 1994. Energy Prices and Taxes: 2nd Quarter 1994. IEA, Paris.

International Energy Agency, 1997. Energy Statistics and Balances of Non-OECD Countries 1994–1995. IEA, Paris.

Jorgensen, S.E., 1998. Exergy as an orientor of the development of ecosystems. In: Advances in Energy Studies: Energy Flows in Ecology and Economy. Museum of Science and Scientific Information, Rome, pp. 371–402.

Judge, G.C., Takayama, T., 1966. Inequality restrictions in regression analysis. Journal of the Statistical Association 61, 166–181.

Judson, D.H., 1989. The convergence of neo-ricardian and embodied energy theories of value and price. Ecological Economics 1, 261–281.

Koopmans, T., 1951. Activity Analysis of Production and Allocation. Wiley, New York.

Kummel, R., Lindenberger, D., 1998. Energy, technical progress and industrial growth. In: Advances in Energy Studies: Energy Flows in Ecology and Economy. Museum of Science and Scientific Information, Rome, pp. 207–222.

Kummel, R., Lindenberger, D., 2000. Productive power of energy and economic evolution. Indian Journal of Applied Economics 8, 231–262.

Lawson, C.L., Hanson, R.J., 1974. Solving Least Squares Problems. Society for Industrial and Applied Mathematics. Prentice Hall, New Jersey.

Leontief, W., 1986. Input–output Economics. Oxford University Press, New York.

Lockwood, M., 1997. Integrated value theory for natural areas. Ecological Economics 20, 83–93.

Lotka, A.J., 1922). Natural selection as a physical principle. Proceedings of the National Academy of Science 8, 147–151.

Lotka, A.J., 1925). Physical Biology. Williams and Wilkins, Baltimore.

Malinvaud, E., 1953. Capital accumulation and efficient allocation of resources. Econometrica 21, 233–269.

O'Connor, M., 1996. Cherish the future, cherish the other: A post classical theory of value. In: Faucheaux, S., Pearce, D., Proops, J. (Eds.), Models of Sustainable Development. Edward Elgar, Cheltenham, pp. 321–345.

Odum, H.T., 1971. Environment, Power and Society. Wiley, New York.

Odum, H.T., 1983. Systems Ecology: An Introduction. Wiley, New York.

Odum, H.T., 1996. Environmental Accounting: EMERGY and Environmental Decision Making. Wiley, New York.

Odum, H.T., 2000. Handbook of Emergy Evaluation: Energy of Global Processes-Folio #2. Center for Environmental Policy, University of Florida, Gainesville.

Odum, H.T., Odum, E.C., 1976. Energy Basis for Man and Nature. McGraw-Hill, New York.

Odum, H.T., Brown, M.T., Brandt-Williams, S., 2000. Handbook of Emergy Evaluation: Introduction and Global Budget-Folio #I. Center for Environmental Policy, University of Florida, Gainesville.

Oort, A.H., 1970. The energy cycle of the earth. Scientific American 223, 54–63.

Organisation of Economic Co-operation and Development,1998. Pricing of Water-Services in OECD Countries: Synthesis Report. Document 69368. Group on Economic and Environment Policy Integration, Environment Directorate, Paris.

Patterson, M.G., 1983. Estimation of the quality of energy sources and uses. Energy Policy 11, 346–359.

Patterson, M.G., 1989. Energy, Productivity and Economic Growth: An Analysis of New Zealand and Overseas Trends. Market Analysis Report 89/1006. Ministry of Energy, Wellington.

Patterson, M.G., 1991. A Systems Approach to Energy Quality and Efficiency. Unpublished revision of a PhD thesis, School of Architecture, Victoria University of Wellington, New Zealand.

Patterson, M.G., 1993. Approaches to energy quality in energy analysis. International Journal of Global Energy Issues 5, 19–28.

Patterson, M.G., 1996. What is energy efficiency? Concepts, indicators and methodological issues. Energy Policy 24, 377–390.

Patterson, M.G., 1998a. Commensuration and theories of value in ecological economics. Ecological Economics 25, 105–123.

Patterson, M.G., 1998b. Understanding energy quality in ecological and economic systems. In: Advances in Energy Studies: Energy Flows in Ecology and Economy. Museum of Science and Scientific Information, Rome, pp. 257–274.

Pearce, D., 1998. Auditing the earth: the value of the world's ecosystem services and natural capital. Environment 40, 23–28.

Perrings, C., 1987. Economy and Environment: A Theoretical Essay of the Interdependencies of Economic and Environmental Systems. Cambridge University Press, New York.

Perrings, C., 1996. Ecological resilience in the sustainability of economic development. In: Faucheaux, S., Pearce, D.W., Proops, J. (Eds.), Models of Sustainable Development. Edward Elgar, London, pp. 231–252.

Pimm, S.L., 1991. The Balance of Nature? Ecological Issues in the Conservation of Species and Communities. University of Chicago Press, Chicago.

Postel, S., Daily, G.C., Ehrlich, P., 1996. Human appropriation of renewable freshwater. Science 27, 785–787.

Proops, J., 2000. Research Challenges for Ecological Economics in the 21st Century. Paper Presented to the Sixth Biennial Meeting of the International Society for Ecological Economics 5–8 July 2000, Canberra, Australia.

Schlesinger, W.H., 1991. Biogeochemistry: An Analysis of Global Change. Academic Press, San Diego.

Shiklomanov, I.A., 1993. World water resources. In: Gleick, D.H. (Ed.), Water in Crisis: A Guide to the World's Fresh Water Resources. Oxford University Press, New York, pp. 13–24.

Slesser, M., 1973. Energy analysis in policy making. New Scientist, November, pp. 328–330.

Soddy, F., 1912. Matter and Energy. Williams and Northgate, London.

Sraffa, P., 1960. Production of Commodities by Means of Commodities: Prelude to a Critique of Economic Theory. Cambridge University Press, Cambridge.

Stigler, G.J., 1965. Essays in the History of Economics. University of Chicago Press, Chicago.

Stout, B.A., 1990. Handbook of Energy for World Agriculture. Elsevier Applied Science, London.

United Nations, 1993. Trends in International Distribution of Global World Product. United Nations, New York.

United Nations, 1996. Industrial Commodities Statistics Yearbook. United Nations, New York.

United Nations, 2000. World Commodity Survey 1999–2000. Markets, Trends and the World Economic Environment. United Nations, New York.

Victor, P.A., 1972). Pollution: Economy and Environment. Allen and Unwin, London.

von Neumann, J., 1945. A model of general equilibrium. Review of Economic Studies 13, 1–9.

Wake, G., McKibbin, R., Cole, A., Patterson, M.G., 1998. Ecological pricing: an eigenvalue–eigenvector solution method. Working Paper, Department of Resource and Environmental Planning, Massey University, Palmerston North, New Zealand.

Wigley, T.M.L., Schimel, D.S., 2000. The Carbon Cycle. Cambridge University Press, Cambridge.

World Bank, 1996. The World Bank Atlas, World Bank, Washington, DC.

World Resources Institute, 1996. World Resources 1996–1997. Oxford University Press, New York.