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Abstract

Life cycle assessment (LCA) enables one to compare the environmental impact of alternative product systems, and identify the one with the least impact. However environmentally sound a product system is, however, it cannot be widely introduced into the economy unless it is also economically affordable. Without a wide ranging introduction into the economy, its potential to reduce environmental impact remains unexploited. The aspect of cost is thus of great importance for assessing the economic affordability and hence the economic sustainability of a product system which is found environmentally sound by an LCA. The aspect of cost that matters here is not limited to the conventional one referring to that of manufacturing only, but should be a broader one referring to the life cycle cost that encompasses the use and the end-of-life cost as well. Life cycle costing (LCC) is concerned with the comparison of life cycle costs among alternative product systems. This paper presents a new methodology of LCC, WIO-price model, that builds upon the WIO quantity model, a hybrid LCA tool developed by us that is specially designed for application to waste management (Nakamura and Kondo, 2002). The applicability of the methodology has been illustrated by a case study comparing the LCC of alternative treatment methods of End of Life-Electrical Home Appliances (TV sets, refrigerators, washing machines, and air conditioners), landfilling, intensive recycling that is consistent with Japanese law on the recycling of EL-EHA, and the advanced form of intensive recycling augmented by reuse of components with Design for Disassembling (DfD).

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In the Japanese recycling law on EL-EHA, the cost of recycling is not internalized in the cost to manufacturers, but is to be paid by final consumers at the time of disposal. The results indicate that internalization of the recycling cost can increase the average price of EHA by less than 1%, and that implementation of DfD can be effective in reducing this price increase by a wide margin. Internalization of the recycling cost is thus expected to give a strong incentive to EHA manufacturers to implement Design for Environment strategies including DfD for reducing life cycle costs. Combined with the result that minimization of environmental loads (global warming and landfill) is achieved under the reuse with DfD strategy, it follows that revising the current recycling law to internalize the recycling cost is effective in making environmentally benign strategy economically affordable as well.

keywords: LCC, LCA, EL-EHA, recycling, DfD, reuse

1 Introduction

Life cycle assessment (LCA) is a powerful tool to assess the environmental impact of alternative product systems (Guinée, 2002). Its application would enable one to compare the environmental impact of alternative systems, and identify the one with the least impact. However sound a product system is with regard to environmental impact, however, it cannot be widely introduced into the economy unless it is also economically affordable. Without a wide ranging introduction into the economy, its potential to reduce environmental impact remains unexploited.

The aspect of cost is thus of great importance for assessing the economic affordability and hence the economic sustainability of an alternative which is found environmentally sound by an LCA. The aspect of cost that matters here, however, is not limited to the conventional one referring to that of manufacturing only, but should be a broader one referring to the life cycle cost that encompasses the use and the end-of-life cost as well. Life cycle costing (LCC) is concerned with the comparison of life cycle costs among alternative product systems (Rebitzer, 2002; Norris, 2001).

With regard to the methodology of LCA, a hybrid use of input-output analysis (IOA) and detailed process modeling has increasingly become a standard one (Matthews and Small, 2001). Along this line of research, we elsewhere have proposed a new methodology termed the waste input-output (WIO) model that is specially designed for an LCA of waste management (Nakamura and Kondo, 2002). This paper is concerned with a new methodology of LCC, WIO-price model, that builds upon the cost and price counterpart of the WIO-quantity model. In IOA, it is well known that dual to any quantity system determining the quantity of flow of inputs and outputs for a given vector of final demand, there does exist a cost or price system that determines the cost per unit of production (and hence the price of output) for a given vector of the price of primary factors (Kurz and Salvadori, 1995; Miller and Blair, 1985). In conventional IOA, the derivation of a price system that is dual to a quantity system is straightforward. In WIO, however, it is not the case because of the presence of waste as joint products and the sale and purchase of waste materials, the value of which can be positive, zero, or negative.

Building upon a preceding Life Cycle Inventory (LCI) study by us (Kondo and Nakamura, 2003), the applicability of the proposed LCC methodology is illustrated by a case study of alternative life-cycle strategies for the end-of-life electrical home appliances (EL-EHA) in Japan consisting of TV sets, refrigerators, washing machines, and air conditioners. Three strategies are considered that consist of landfilling, intensive recycling, and reuse with design for disassembling (DfD). Under landfilling, EL-EHA are directly landfilled without any pretreatment except for the recovery and destruction of chlorofluorocarbon (CFC) 12. This used to be a dominant treatment method prior to the introduction of a law on the recycling of EL-EHA in 2001 (METI, 1999). Under intensive recycling, metals (both iron and non-iron) and non-metals such as glass and plastics are recovered for recycling, and (in addition to CFC12) CFC11 contained in the urethane foam of refrigerators is also recovered for destruction. The rate of material recovery under this method satisfies the norms set by the recycling law. Finally, reuse with DfD refers to an "advanced" scenario where, in addition to the features of intensive recycling, implementation of DfD makes possible the recovery of plastics at a purity required for reuse as plastic components, whereas under intensive recycling they can only be downcycled.

With regard to its institutional design, the Japanese law on EL-EHA recycling is characterized by the fact that the treatment cost is paid by final consumers at the end of use when the appliance is discarded (METI, 1999). The end-of-life cost is thus not internalized in the manufacturing cost of EHA. Critics of the law regard this as a shortcoming because it may induce illegal dumping while providing the manufactures of EHA with little incentive for developing products with smaller environmental impact (for instance, by implementing DfD and introducing more durable products). With regard to the first point, the experience over the last two years (after the enforcement of the law) has shown no significant increase in illegal dumping of EL-EHA (MOE, 2003). Given that the law is to be revised in 2006, it seems important to provide some quantitative information with regard to the second point. We do so by LCC of internalizing the cost of recycling in the manufacturing costs of EHA.

The paper is structured as follows. In Section 2 we start with a brief introduction to the WIO quantity model, proceed to the derivation of its price counterpart, the WIO price model, and then turn for its implementation to the Japanese WIO table for 1995. The WIO price model thus obtained is then applied in Section 3 to evaluate the alternative life cycle strategies. Concluding remarks in Section 4 close the paper.

2 The WIO-price model

2.1 Notations and basics

Let there be n^{I} goods- and service-producing sectors (henceforth "goods sector"), n^{II} waste treatment sectors, n^{w} waste types, and $n := n^{\mathrm{I}} + n^{\mathrm{II}}$. For ease of exposition, we define the sets of natural numbers referring to each of these sectors and waste types by $N^{\mathrm{I}} := \{1, \ldots, n^{\mathrm{I}}\}$, $N^{\mathrm{II}} := \{n^{\mathrm{I}} + 1, \ldots, n^{\mathrm{I}} + n^{\mathrm{II}}\}$, $N := N^{\mathrm{I}} \cup N^{\mathrm{II}}$, and $N^{\mathrm{w}} := \{1, \ldots, n^{\mathrm{w}}\}$.

We then denote, for sector j ($j \in N$), its output by x_j, the input from sector i ($i \in N$) by X_{ij}, the generation of waste k ($k \in N^{\mathrm{w}}$) by $W^{\oplus}_{kj}$, and the input of waste k by $W^{\ominus}_{kj}$. For a waste treatment sector, its "output" is measured by the amount of waste it treated. Similarly, we denote the final demand for i ($i \in N$) by $X_{i\mathrm{F}}$, the generation of waste k ($k \in N^{\mathrm{w}}$) from the final demand sector by $W^{\oplus}_{k\mathrm{F}}$, and the input of waste k into the final demand sector by $W^{\ominus}_{k\mathrm{F}}$.

Let $a_{ij} := X_{ij}/x_j$ be the conventional input coefficient denoting the input from sector i per unit output of sector j. In a similar fashion, we define the primary input coefficient $v_j := V_j/x_j$, the waste generation coefficient $g^{\oplus}_{kj} := W^{\oplus}_{kj}/x_j$, and the waste input coefficient $g^{\ominus}_{kj} := W^{\ominus}_{kj}/x_j$.

Using these coefficients, we obtain the following expression for the output of goods and the amount of waste treatment:

$$x_i = \sum_{j\in N^{\mathrm{I}}} a_{ij}x_j + \sum_{j\in N^{\mathrm{II}}} a_{ij}x_j + X_{iF}, \quad (i \in N^{\mathrm{I}}),$$

$$x_l = \sum_{k\in N^{\mathrm{w}}} s_{lk} \left(\sum_{j\in N^{\mathrm{I}}} (g^{\oplus}_{kj} - g^{\ominus}_{kj})x_j + \sum_{j\in N^{\mathrm{II}}} (g^{\oplus}_{kj} - g^{\ominus}_{kj})x_j + W_{kF} \right), \quad (l \in N^{\mathrm{II}}),$$

where $W_{kF} = W^{\oplus}_{kF} - W^{\ominus}_{kF}$ and s_{lk} is the allocation coefficient that refers to the share of waste k that is treated by waste treatment l (Nakamura and Kondo, 2002). Let S be an $n^{\mathrm{II}} \times n^{\mathrm{w}}$ matrix whose (i, j)-component is s_{ij}. In obvious matrix notations, these equations can then be expressed as

$$x_{\mathrm{I}} = A_{\mathrm{I,I}}x_{\mathrm{I}} + A_{\mathrm{I,II}}x_{\mathrm{II}} + X_{\mathrm{I},F}, \tag{1}$$

$$x_{\mathrm{II}} = S\,(G^{\oplus}_{\cdot,\mathrm{I}} - G^{\ominus}_{\cdot,\mathrm{I}})x_{\mathrm{I}} + S\,(G^{\oplus}_{\cdot,\mathrm{II}} - G^{\ominus}_{\cdot,\mathrm{II}})x_{\mathrm{II}} + S\,W_{\cdot,F}, \tag{2}$$

the solution of which can be given by

$$\begin{pmatrix} x_{\mathrm{I}} \\ x_{\mathrm{II}} \end{pmatrix} = \left(I - \begin{pmatrix} A_{\mathrm{I,I}} & A_{\mathrm{I,II}} \\ S\,(G^{\oplus}_{\cdot,\mathrm{I}} - G^{\ominus}_{\cdot,\mathrm{I}}) & S\,(G^{\oplus}_{\cdot,\mathrm{II}} - G^{\ominus}_{\cdot,\mathrm{II}}) \end{pmatrix} \right)^{-1} \begin{pmatrix} X_{\mathrm{I},F} \\ S\,W_{\cdot,F} \end{pmatrix}, \tag{3}$$

where I is an identity matrix of an appropriate order. The system of cost and price equations that corresponds to this quantity system needs to be derived.

2.2 The Wio-price model

2.2.1 Cost categories

The input-output account system has an identity that equates the value of output to the total cost. In the WIO model, the following five categories of the total cost can be distinguished:

(a) the cost for the input of goods,

(b) the cost for waste treatment,

(c) the cost for the input of waste materials,

(d) the revenue from the sale of waste materials, and

(e) the cost for the input of primary factors.

Corresponding to this factorization of the cost categories, the cost equation for sector j $(j \in N)$ can be given by:

$$p_j x_j = \underbrace{\sum_{i \in N^{\mathrm{I}}} p_i a_{ij} x_j}_{(a)} + \underbrace{\sum_{l \in N^{\mathrm{II}}} p_l \sum_{k \in N^{\mathrm{w}}} s_{lk}(g^{\oplus}_{kj} x_j - R_{kj})}_{(b)} + \underbrace{\sum_{k \in N^{\mathrm{w}}} p^{\mathrm{w}}_k g^{\ominus}_{kj} x_j}_{(c)} - \underbrace{\sum_{k \in N^{\mathrm{w}}} p^{\mathrm{w}}_k R_{kj}}_{(d)} + \underbrace{V_j}_{(e)}, \tag{4}$$

where p_j is the price of output of sector j $(j \in N)$, p^{w}_k is the price of waste $k \in N^{\mathrm{w}}$, V_j is the cost for primary factors of production that includes depreciations as well as taxes less subsidies, and $R_{kj} \geq 0$ is the quantity of waste $k \in N^{\mathrm{w}}$ that is used as input in sector $j \in N \cup \{\mathrm{F}\}$. The presence of the terms (b), (c), and (d) that give an explicit representation of the cost of waste treatment, and the sale and purchase of recovered waste materials, is a distinguishing feature of the WIO-price model. While these cost elements also occur in the traditional Leontief price model (Miller and Blair, 1985), its representation is implicit and does not provide detailed information. When there is no recycling of waste materials, $R_{kj} = 0$ holds for all k and j, and the terms (c) and (d) vanish, while the term (b) reduces to the treatment cost of wastes generated in sector j.

The sale of recovered waste materials is an important source of revenue for waste recyclers. A typical example is the disassembly of discarded automobiles, the major revenue source of which has been the sale of scrap metal to steel makers operating electric arc furnaces. There are, however, cases where the value of waste materials is negative, that is, the user is paid to accept the waste materials. For instance, several Japanese iron makers accept waste plastics against payment and use them, after a series of pretreatments, as a reduction agent in blast furnaces together with pulverized coal. Another example is the acceptance with payment of scrap tires and waste plastics by several Japanese cement manufacturers as supplementary heat sources in the kiln.

The price of waste can thus become negative. Based on its sign condition, three cases can be distinguished: the waste has a positive value when $p_k^w > 0$; it has zero value but can be accepted by other sectors as input with no payment when $p_k^w = 0$; and it has negative value and can be accepted only with payment when $p_k^w < 0$. Henceforth, R_{kj} is called "sale of waste" regardless of whether the price of waste k is positive, zero, or negative.

The term (b) indicates that the amount of waste for treatment is reduced by the amount of R_{kj}. The sale of waste materials at positive prices can reduce the cost of production or treatment in two ways. First, it can reduce the cost directly by creating a new source of revenue other than the sale of "main" output. The term (d) refers to this component. Secondly, it can reduce the waste treatment cost that would have been necessary if the waste materials were not sold but had to be treated with payment. The term (b) refers to this component. On the other hand, the sale of waste at negative prices reduces the production cost (increases the revenue) of the sectors that use the waste as input.

Rearranging the terms in (4) yields the following expression, which shows the contribution of the sale of waste materials to the cost in a more explicit way

$$p_j x_j = \underbrace{\sum_{i \in N^I} p_i a_{ij} x_j}_{(a)} + \underbrace{\sum_{l \in N^{II}} p_l \sum_{k \in N^w} s_{lk} g_{kj}^{\oplus} x_j}_{(f)} + \underbrace{\sum_{k \in N^w} p_k^w g_{kj}^{\ominus} x_j}_{(c)} - \underbrace{\sum_{k \in N^w} \left(p_k^w + \sum_{l \in N^{II}} p_l s_{lk} \right) R_{kj}}_{(g)} + V_j, \tag{5}$$

Here, the term (f) refers to the waste treatment cost that would have been necessary if no waste materials were sold. When waste is sold to other sectors, it can affect the cost via the term (g). The extent to which the cost can be reduced by the sale of waste depends on the sign condition of the expression inside the parentheses of (g). When $p_k^w > 0$, the sale of waste certainly reduces the cost of production. It is important to note that even if $p_k^w \leq 0$, the sale of waste could reduce the cost as long as the following condition is satisfied:

$$p_k^w + \sum_{l \in N^{II}} p_l s_{lk} > 0 \Longleftrightarrow |p_k^w| = -p_k^w < \sum_{l \in N^{II}} p_l s_{lk}. \tag{6}$$

This refers to the case where the sale of waste to other sectors at negative prices costs less than submitting it to waste treatment.

2.2.2 Price equations

The term R_{kj} plays a vital role in the cost equation (5). It does not, however, occur in the system of equations (1) and (2) for the quantity model. It is necessary to establish the relationship between R_{kj} and the elements occurring in the quantity model. Let R_{kji} be the amount of waste k generated by sector j ($j \in N \cup \{\mathrm{F}\}$) that is used as input in sector i ($i \in N$). By definition

$$R_{kj} = \sum_{i \in N} R_{kji}. \tag{7}$$

For the sake of simplicity, we assume $W^{\ominus}_{k\mathrm{F}} = 0$; the household does not "directly" engage in recycling in the sense that it does not directly use waste, while it would indirectly engage in recycling by purchasing goods made of recovered waste or produced by using waste heat. The input of waste k into sector i can then be represented as:

$$W^{\ominus}_{ki} = g^{\ominus}_{ki} x_i = \sum_{j \in N \cup \{\mathrm{F}\}} R_{kji}. \tag{8}$$

Of waste k used in sector i, the portion that originates from sector j would be proportional to the share of sector j in the total generation of that waste, $W^{\oplus}_{kj} \,/\, W^{\oplus}_{k}$, where $W^{\oplus}_{k} = \sum_{j \in N \cup \{\mathrm{F}\}} W^{\oplus}_{kj}$. Accordingly, we obtain the following expression for the amount of waste k that is generated in sector j and is used in sector i:

$$R_{kji} = W^{\ominus}_{ki}(W^{\oplus}_{kj} \,/\, W^{\oplus}_{k}). \tag{9}$$

It then follows from (7) and (9) that

$$R_{kj} = \sum_{i \in N} R_{kji} = \sum_{i \in N} W^{\ominus}_{ki}(W^{\oplus}_{kj} \,/\, W^{\oplus}_{k}) = W^{\oplus}_{kj}(W^{\ominus}_{k} \,/\, W^{\oplus}_{k}) =: W^{\oplus}_{kj} r_k, \tag{10}$$

where r_k refers to the rate of recycling of waste k with $r_k := W^{\ominus}_{k}/W^{\oplus}_{k}$, and the third equality follows from $\sum_{i \in N} W^{\ominus}_{ki} = W^{\ominus}_{k}$. Recalling the definition of $g^{\oplus}_{kj}$, we obtain from the above

$$R_{kj} = W^{\oplus}_{kj} r_k = g^{\oplus}_{kj} x_j r_k. \tag{11}$$

Insertion of (11) into (5) yields the following expression of the cost equation:

$$p_j x_j = \sum_{i\in N^{\mathrm{I}}} p_i a_{ij} x_j + \sum_{l\in N^{\mathrm{II}}} p_l \sum_{k\in N^{\mathrm{w}}} s_{lk} g^{\oplus}_{kj} x_j + \sum_{k\in N^{\mathrm{w}}} p^{\mathrm{w}}_k g^{\ominus}_{kj} x_j - \sum_{k\in N^{\mathrm{w}}} \Big(p^{\mathrm{w}}_k + \sum_{l\in N^{\mathrm{II}}} p_l s_{lk} \Big) r_k g^{\oplus}_{kj} x_j + V_j.$$

Division of both the sides by x_j yields the following price equation (or unit cost function):

$$\begin{aligned} p_j &= \sum_{i\in N^{\mathrm{I}}} p_i a_{ij} + \sum_{l\in N^{\mathrm{II}}} p_l \sum_{k\in N^{\mathrm{w}}} s_{lk} g^{\oplus}_{kj} + \sum_{k\in N^{\mathrm{w}}} p^{\mathrm{w}}_k g^{\ominus}_{kj} - \sum_{k\in N^{\mathrm{w}}} \Big(p^{\mathrm{w}}_k + \sum_{l\in N^{\mathrm{II}}} p_l s_{lk} \Big) r_k g^{\oplus}_{kj} + v_j, \\ &= \sum_{i\in N^{\mathrm{I}}} p_i a_{ij} + \sum_{l\in N^{\mathrm{II}}} p_l \sum_{k\in N^{\mathrm{w}}} s_{lk} (1 - r_k) g^{\oplus}_{kj} + \sum_{k\in N^{\mathrm{w}}} p^{\mathrm{w}}_k (g^{\ominus}_{kj} - r_k g^{\oplus}_{kj}) + v_j \end{aligned} \tag{12}$$

where v_j refers to the average price of primary inputs used in sector j.

2.2.3 The price versus quantity WIO model

Using obvious matrix notations, (12) can be rewritten as

$$\begin{pmatrix} p_{\mathrm{I}} & p_{\mathrm{II}} \end{pmatrix} = \begin{pmatrix} p_{\mathrm{I}} & p_{\mathrm{II}} \end{pmatrix} \begin{pmatrix} A_{\mathrm{I,I}} & A_{\mathrm{I,II}} \\ S(I-D)G^{\oplus}_{\mathrm{I}} & S(I-D)G^{\oplus}_{\mathrm{II}} \end{pmatrix} + p^{\mathrm{w}} \begin{pmatrix} G^{\ominus}_{\mathrm{I}} - DG^{\oplus}_{\mathrm{I}} & G^{\ominus}_{\mathrm{II}} - DG^{\oplus}_{\mathrm{II}} \end{pmatrix} + \begin{pmatrix} v_{\mathrm{I}} & v_{\mathrm{II}} \end{pmatrix}, \tag{13}$$

where $p = (p_{\mathrm{I}}, p_{\mathrm{II}}) = (p_1, \dots, p_{n^{\mathrm{I}}}, p_{n^{\mathrm{I}}+1}, \dots, p_n)$, $v = (v_{\mathrm{I}}, v_{\mathrm{II}}) = (v_1, \dots, v_n)$, $p^{\mathrm{w}} = (p^{\mathrm{w}}_1, \dots, p^{\mathrm{w}}_{n^{\mathrm{w}}})$, and D is a diagonal matrix whose k-th diagonal component is r_k, i.e., $D = \mathrm{diag}(r_1, \dots, r_{n^{\mathrm{w}}})$. This can be further rewritten in a more compact way as

$$p = p \begin{pmatrix} A_{\mathrm{I},\cdot} \\ S(I-D)G^{\oplus} \end{pmatrix} + p^{\mathrm{w}}(G^{\ominus} - DG^{\oplus}) + v \tag{14}$$

Provided it is possible to solve (14) for p, this solution can be given by

$$p = \left\{ p^{\mathrm{w}}(G^{\ominus} - DG^{\oplus}) + v \right\} \left(I - \begin{pmatrix} A_{\mathrm{I},\cdot} \\ S(I-D)G^{\oplus} \end{pmatrix} \right)^{-1}. \tag{15}$$

Recall from (3) that the solution of the WIO quantity model is given by

$$x = \begin{pmatrix} x_{\text{I}} \\ x_{\text{II}} \end{pmatrix} = \left(I - \begin{pmatrix} A_{\text{I},\cdot} \\ S(G^{\oplus} - G^{\ominus}) \end{pmatrix} \right)^{-1} \begin{pmatrix} X_{\text{I},\text{F}} \\ S(W_{\text{F}}^{\oplus} - W_{\text{F}}^{\ominus}) \end{pmatrix}. \tag{16}$$

Comparing the inverse matrices occurring in (15) and (16), we find that the former reduces to the latter for arbitrary S and D only if $DG^{\oplus} = G^{\ominus}$ holds, that is, when the following holds:

$$r_k g_{kj}^{\oplus} = g_{kj}^{\ominus} \quad (k \in N^{\text{w}},\, j \in N). \tag{17}$$

Under this condition, each sector is self-sufficient with respect to the waste materials it requires, and hence the sale of waste to other sectors does not occur. The solution (15) then reduces to

$$p = v \left(I - \begin{pmatrix} A \\ S(G^{\oplus} - G^{\ominus}) \end{pmatrix} \right)^{-1}, \tag{18}$$

which corresponds to a dual price model in the conventional IOA (Kurz and Salvadori, 1995). Note that (17) includes the case where there is no recycling of waste at all, that is $r_k = 0$ for all k.

Except for the case where (17) holds, the system of price equations includes the matrix D referring to the rate of recycle. Because each element of D, r_k, depends on the level of output of all the sectors, it follows that the solution of the price model also depends on the level of output of all the sectors. In other words, the level of unit cost of a particular sector cannot be determined independent of the level of final demand in spite of the fact that the technology is assumed to be homogeneous of degree one. The non-substitution theorem (Samuelson, 1951) does not hold because of the presence of joint products, that is, waste.

2.3 Implementation to Japanese WIO data

The above model is implemented to the Japanese WIO data for 1995 (Nakamura, 2003). The WIO table comprises of seventy-eight industry sectors, five basic treatment methods (composting, gasification, shredding, incineration, and landfilling), thirty-four waste types that cover both municipal solid waste and industrial waste, and nine types of bulky waste (see Nakamura and Kondo (2002) for further details).

In (15) the price of waste p^{w} and the unit cost of primary inputs v are exogenous variables unless further information on the mechanism of their determination is available, while the coefficient matrices A, G and S are given parameters. Primary factors in the present context consist of labor, capital, imported intermediate inputs, and land. With regard to imports, note that the matrix of input coefficients A refers to domestic inputs only and does not include imports. Land is the essential input for landfilling; land is not a stock but a flow that is to be used up by landfilling. Following Tanaka and Matsuto (1998), the rent for landfill area per ton of waste was obtained assuming a price of land of 1000 yen per square meter and an interest rate of 3%.

Of the waste items, iron scraps, nonferrous metal scraps, and waste paper are characterized by the presence of well-established international markets. In the following, the price of these waste items is exogenously given, and its value taken from the Japanese IO table (MCA, 1999).

The situation is quite different for the remaining waste items. A nationwide market is almost nonexistent for them. Furthermore, it is frequently the case that their price takes a negative value even when they are used as input in production process; a typical example is the injection of waste plastics into blast furnaces in the iron industry. For the sake of simplicity, the price of these waste items was set equal to the negative of the weighted average of the price of relevant treatment methods with the weights given by the corresponding elements of the allocation matrix S. In other words, for these waste items, p^{w} is given by (6), with its inequality replaced by an equality, and is determined within the model simultaneously with p_{I} and p_{II}.

Figure 1 shows the computed price of sectoral output obtained from (15) under these assumptions. Because the original value of the price (index) of sectoral outputs is unity, the computed price should also take values that are close to unity. Given the set of strong assumptions that were introduced for simplification, the results can be regarded as acceptable because the computed prices lie within the interval of 1.00 to 0.95 for a majority of cases. The only noticeable exception is the result for the cement industry, where the level of price index is underestimated by more than 35%. The cement industry in Japan is a typical recycling industry that accepts a large amount of inorganic waste such as ash and slag as materials, and organic waste such as waste tires and plastics as fuel. Our simplifying assumption on the equality between the price of these waste items and the negative of the weighted values of the relevant treatment prices is certainly responsible for over-estimating the revenue from recycling in the cement industry. In order to reduce possible biases due to this over-estimation, a modification factor was introduced for its correction. This rather ad-hoc method was applied because in the present case the cement industry is only remotely connected to the recycling of EL-EHA.

Before turning to application, a remark is due on the algorithm for solving the model. We have introduced above the WIO-price model as a linear system. The WIO-quantity model, however, is intrinsically non-linear because of its incorporation of an engineering submodel of waste treatment that is based on physical-chemical causality between waste feedstock, the input of utilities and chemicals that are required for its treatment, and the output of residues and emissions (Tanaka and Matsuto, 1998). Its solution is obtained iteratively until the vector of waste feedstock stabilizes in that it satisfies a given criterion of convergence (Nakamura and Kondo, 2002). Accordingly, the WIO-price model is also non-linear, with a linear expression such as (15) holding only for a given volume and composition of waste feedstock. Its actual computation is carried out by solving the linear system (15), after the solution of the WIO-quantity model, and hence the coefficients on the right hand side of (15) has been obtained in the iterative computation.

3 Application: Recycling of EL-EHA in Japan

We now turn to application of the proposed LCC methodology to a case study of three alternative life cycle strategies of end-of-life electrical home appliances (EL-EHA) in Japan consisting of TV sets, refrigerators, washing machines, and air conditioners. The first strategy is a landfilling scenario where all EL-EHA are landfilled except for recovery and destruction of CFC12, that was dominant in Japan until the introduction of an EL-EHA recycling law in 2001 (METI, 1999). The second strategy is an intensive recycling scenario under which metals, glass, and plastic components, as well as CFC11, are recovered in accordance with the recycling law (AEHA, 1999). Augmenting the recycling strategy with the reuse of plastics under implementation of design for disassembling (DfD) yields the last strategy, termed the reuse with DfD scenario. Under this scenario, 99% of polypropylene (PP) and polystyrene (PS) are recovered at a purity adequate to be reused as materials for industrial plastic products with a positive price, while the remaining plastics are accepted with payment for injection into blast furnaces in the iron and steel industry. Due to its implementation of DfD, the efficiency of disassembling under this scenario is assumed to be twice the level of the recycling scenario. Under the recycling scenario, injection into blast furnaces with payment is the only way of utilizing recovered waste plastics.

We have elsewhere performed an LCI of the three scenarios based on WIO with regard to landfill consumption and the emission of carbon dioxide (Kondo and Nakamura, 2003), a summary of which is reproduced in Figure 2. The results indicate that the recycling scenario reduces both environmental loads, and that the additional implementation of reuse with DfD is effective for a further reduction in landfilling. We now turn to LCC of these environmentally benign scenarios to see the extent to which they are economically affordable.

3.1 Life Cycle Costing

Despite the similarity of their names, LCC and LCA have major methodological differences, which result from the fact that LCC and LCA are each designed to provide answers to very different questions (Norris, 2001). In particular, a fully integrating meaningful economic analysis into LCA would require introducing variables that have no causal dependence upon inventory flows and also to capture risks, which implies a significant departure from the prevailing LCA methodology, which is mostly linear and deterministic. On the other hand, it is also true that there is no uniform understanding of the term life-cycle costing nor is there a standardized methodological framework that is commonly used in business (Rebitzer, 2002).

Because of its simplicity and close relationship to LCI, we have chosen to use the following LCC concept (Rebitzer, 2002):

$$LCC := R\&D + MAT + TRNS + MANF + USE + EOL + TC \tag{19}$$

where $R\&D$, MAT, $TRNS$, $MANF$, USE, EOL, and TC refer to the costs for research and development, materials, transports/logistics, manufacturing, use, end-of-life, and transaction costs, respectively. By nature, an IO table depicts all monetary flows of inputs and outputs including the items *MAT*, *TRNS*, *MANF*, and *TC* so far as they refer to current expenditures. In the Japanese IO table, the current expenditure for research and development is also recorded as an input item. To the extent that the current expenditure for research and development recorded in the IO table (including WIO) corresponds to the above concept of *R&D*, implementation of the above LCC concept within an IO model is rather straightforward except for the terms *USE* and *EOL*. Because the same use pattern of EHA applies to each of the scenarios, it is unnecessary to consider the cost associated with the use phase *USE*.

Under the Japanese law on EL-EHA recycling, the end-of-life cost is not internalized in the cost of manufacturers, but is paid directly by the consumers when they generate EL-EHA. In the following, we incorporate the EOL cost into (4) à la (19) for the purpose of estimating the "latent" LCC, because only internal and internalized costs should be accounted for in LCC (Rebitzer and Hunkeler, 2003).

Durables like EHA become waste with the passing of its product life in a dynamic process. The WIO model, however, is a static one, and cannot deal with a dynamic process over time where durables are transformed into waste. Explicit consideration of such a dynamic process would require detailed information on the products cohort and its distribution over time (Yokota et al., 2003).

In the present case of EL-EHA recycling in Japan, however, a simple stationary model can be applied because of the fact that it deals with very mature markets where any new purchase will be a replacement of an old appliance by a new one, and that under the recycling law the old appliance to be replaced is taken back by the same retailer delivering the new product and transported to the recycling facilities of the relevant manufacturers. The EOL cost can then be approximated by the treatment (recycling) cost of EL-EHA to be replaced, and (19) can be written as

$$p_j^* x_j := LCC = \text{(a)} + \cdots + \text{(e)} + \text{recycling cost of EL-EHA}, \tag{20}$$

where (a) to (e) refer to the cost components occurring in (4), and p_j^* refers to the price level when the EOL cost is internalized. While (20) should only be seen as a pragmatic approximation of LCC, it would give an exact representation in a stationary situation where the new EHA and the EL-EHA it replaces are the same except for their age.

In the current version of the Japanese WIO table, EHA does not occur as a single sector but is included in an aggregated sector called "Household electrical appliances" where the share of EHA in output accounts for about 30%. In the following analysis, the effects on the price of EHA were obtained from that on the aggregated sector by multiplying the latter with the reciprocal of the output share.

Recently, the construction of a new landfill site in Japan has become increasingly difficult, and the landfilling fee has been steadily rising. For the purpose of evaluating the possible effects on the price of goods of an increase in the landfilling fee, its level is henceforth given exogenously. The price level of landfilling obtained above by use of (15) serves as the reference value.

3.2 Results

3.2.1 Effects of cost internalization

Figure 3 shows the major effects of EL-EHA recycling on the price of goods relative to the reference case where all EL-EHA are landfilled. Increased supply of recovered materials like copper, aluminum, and glass brought about by the recycling of EL-EHA reduces the price of goods that make an intensive use of these materials. This applies to rolled and drawn copper (−1.7%) and aluminum and (−0.2%), and glass products (−0.1%).

On the other hand, the reduction in the price of pig iron (−0.2%) and converter steel (−0.1%) cannot be explained by the increased supply of iron scraps; the production of pig iron does not make use of iron scraps, and converter steel is produced from pig iron. Recall that the iron and steel industry accepts waste plastics with payment, and uses it as a substitute for coke in blast furnaces. The decline in the price of pig iron indicates that the revenue from this waste treatment activity plus the cost for coke saved by substitution by waste plastics exceeds the additional cost of pretreatment that is required to make waste plastics suitable for injection into blast furnaces. While Figure 3 shows the sectors characterized by large effects only, the picture it conveys is a representative one insofar as the cost of recycling is not internalized. In particular, there is no goods producing sector where the price of output is increased under the recycling scenario.

Internalization of the recycling cost increases the price of EHA by 0.7%, but its effect on the price of other goods remains the same as the case where the cost is not internalized, because EHA is a final product that is used by final consumers only. It is important to note that this increase in the price of EHA is an offsetting result of the increase in cost due to the addition of recycling cost and the decline in the cost of materials due to increased material recovery under recycling as mentioned above.

3.2.2 Cost internalization and DfD

It is uncertain whether an increase in the price of less than 1 percent is large enough to induce a significant reduction in the demand for EHA on the side of consumers. With this regard, one should also note that while internalization of the recycling cost would raise the price of EHA, given that any new purchase generates EL-EHA to be disposed of with payment, the actual cost for consumers including both new purchase and disposal is not affected by the internalization. Because we do not consider substitution effects in consumer expenditure, institutional setups about who pays the recycling cost directly does not affect the net total cost (new purchase and recycling) for the consumers.

It is certain, however, that internalization of the recycling cost would give a strong incentive for manufacturers to reduce the cost. Disassembling is seen as "the first and most important point in the recycling process" (Knoth et al., 2001). Increasing the efficiency of disassembling thus emerges as the most important factor for cost reduction. How can this be realized? It is important to recall that "costs and revenues as well as the environmental impacts of products are determined to a high percentage in the design phase" (Hunkeler and Rebitzer, 2003). Internalization of the recycling cost would thus lead to a broad implementation of DfD that increases the yield as well as the quality of recovered materials while reducing at the same time the amount of residue that has to be treated as waste. Accordingly, implementation of DfD can reduce the cost of recycling in two ways: by reducing the cost of disassembling and the waste treatment of residue, and by increasing the revenue from the sale of recovered materials at a purity that was not attained without implementing DfD.

Figure 3 shows that (under our assumptions) implementation of the reuse with DfD scenario can reduce the rise in the EHA price due to cost internalization by 0.5% from 0.7% to 0.2%. This is made possible by a significant reduction in the recycling cost that is brought about by the increase in the efficiency of disassembling (reduction by 50% in the amount of inputs required for processing a given amount of EL-EHA), in the recovery of plastics (PP and PS) at a purity that can substitute virgin materials, and by the associated reduction in the amount of shredding residue that has to be landfilled. The prices of recovered PP and PS were set equal to the mean price levels of PP and PS used in the home electric appliance sector (0.22 million yen per ton) taken from the Japanese IO table (MCA, 1999). Recovery of high quality plastics and its supply as materials to plastics manufacturing reduces the price of plastics by 0.3%. The increase in the recovery of high quality plastics reduces the amount of mixed plastics of low quality, which were accepted with payment in the iron and steel industry for injection into blast furnaces. This works to reduce the price lowering effect of EL-EHA recycling on pig iron and converter steel.

3.2.3 Effects of the landfilling fee

We saw above that internalization of the recycling cost increases the unit manufacturing cost of EHA and hence its price, while the extent of increase in the cost and price could significantly be reduced by implementation of a reuse with DfD strategy. Needless to say, our computation is subject to a number of assumptions with different degrees of uncertainty and variability.

The price level of recovered materials and the level of the landfilling fee are important determinants of the recycling cost. The level of these exogenous variables has been subject to high variability. Iron scraps and copper are international commodities, the price of which fluctuates in a stochastic fashion. On the other hand, the level of the landfilling fee is to a large extent determined by domestic conditions, because waste cannot be exported to foreign landfill sites. The tightening of environmental regulations with regard to the operation of landfill sites (landfilling of shredder residue in Japan is allowed only to a controlled type landfill site equipped with leachate control facilities, while it used to be disposed of in simple inert type sites until 1995) and the increasing difficulty of new construction due partly to resistance by local people has resulted in a shortage of landfill capacity and in a steady increase in the fee. For instance, in the Tokyo metropolitan area, where the shortage of landfill has been especially acute, the fee rose by 100% from 1997 to 2001 (METI, 2003). In contrast to the price of recovered materials that are subject to stochastic fluctuations, there is hardly any indication of a reverse trend for the development of the landfilling fee. A further increase in the fee is thus likely to be the case in the foreseeable future.

With this background, the effect of an increase in the landfilling fee on the price of EHA is evaluated assuming that the treatment cost is internalized. Figure 4 shows the results when the fee was increased by 100% and 200%. The bar at far left gives the normalized price of EHA that corresponds to the reference case where all EL-EHA are landfilled. The bars in the left block termed Control refer to the results in Figure 3. With an increase in the landfilling fee, the comparative cost disadvantage of recycling against landfilling decreases. When the fee is twice the level of 1995, the price associated with the reuse with DfD scenario becomes equal to that of the landfilling scenario. When the fee is further increased to three times the level of 1995, the price level under the reuse with DfD scenario becomes 0.02% lower than that of the landfilling scenario, and the price difference between the recycling (without DfD) and the landfilling scenario is reduced to 0.03%, which is less than half the difference that prevailed when the landfilling fee was at the level of 1995. It follows that with a strong increasing tendency of landfilling cost, the comparative cost disadvantage against landfilling of recycling in general and its combination with a reuse with DfD strategy in particular is likely to be substantially reduced.

In the above scenarios, possible costs of risks associated with the emission of hazardous substances such as lead contained in CRT (lead is recovered under recycling, but is landfilled otherwise) are not considered. Inclusion of these cost elements would further work to improve the cost disadvantage of recycling against landfilling. Of the three treatment scenarios considered, the reuse with DfD scenario is characterized by the least environmental impact (Figure 2). The above results indicate that the reuse with DfD scenario can in fact be the one that is economically most efficient in a country like Japan where landfilling space is becoming a scarce resource and the landfilling fee is steadily rising. Internalization of the recycling cost would give a strong incentive to manufacturers for implementing various Design for Environment (DfE) strategies including DfD. Internalization of the recycling cost thus appears effective in making the environmentally favorable strategy of EHA recycling economically affordable as well.

4 Concluding Remarks

A hybrid LCC methodology was presented that builds upon the hybrid LCI methodology based on WIO. The applicability of the methodology has been illustrated by a case study comparing the LCC of alternative treatment methods of EL-EHA (TV sets, refrigerators, washing machines, and air conditioners), landfilling, intensive recycling that is consistent with Japanese law on the recycling of EL-EHA, and the advanced form of intensive recycling augmented by reuse with DfD.

The results indicate that internalization of the recycling cost can increase the average price of EHA by less than 1%, and that implementation of DfD can be effective in reducing this price increase by a wide margin. In the Japanese recycling law on EL-EHA, the cost of recycling is not internalized in the cost to manufacturers, but is to be paid by final consumers at the time of disposal. Internalization of the recycling cost is thus expected to give a strong incentive to EHA manufacturers to implement DfE strategies including DfD for reducing life cycle costs. The rising tendency of the landfilling fee gives further impetus toward the broad implementation of DfD. Combined with the LCI result that the reuse with DfD strategy minimizes environmental loads (global warming and landfill), we conclude that revising the current recycling law to internalize the recycling cost is effective in making environmentally benign strategy economically affordable as well. The same methodology can be applied to the LCC of the recycling of used automobiles that is going to be introduced in the EU and Japan.

Before closing, we would like to point to certain weaknesses of our methodology that should be considered in future research. For the sake of simplification, we assumed a steady state where the production and disposal of durables take place simultaneously. While this may serve as a useful approximation in the present case, it cannot deal with issues concerned with the life of a product, which is another important aspect of DfE. The issue regarding the life of a product cannot be properly dealt with without considering the dynamics of waste generation, which in the case of durables can easily stretch over dozens of years. A combination of WIO with the population-based approach in LCA (Yokota et al., 2003) may prove to be a promising future direction for research.

Exclusion of any possibility of substitution in response to price changes is another weakness of this study. In economics, it is usual to consider the possibility of substitution by introducing a utility and production function of top-down types like nested constant elasticity of substitution (CES) functions. This approach is extensively used in so-called Computable General Equilibrium (CGE) models. This approach, however, makes use of a set of rather strong assumptions on the structure of technology in terms of "separability structure" and suffers from ambiguity with respect to the choice of specific values for unknown parameters such as "elasticity of substitution", and also by possible violation of the mass balance condition that is of fundamental importance in LCA. An alternative to this top-down approach would be a hybrid approach that builds upon decision analytic extension based on linear programming of the WIO model that incorporates detailed technical information (Kondo and Nakamura, 2002). Research is currently underway to implement this approach into the WIO-price model.

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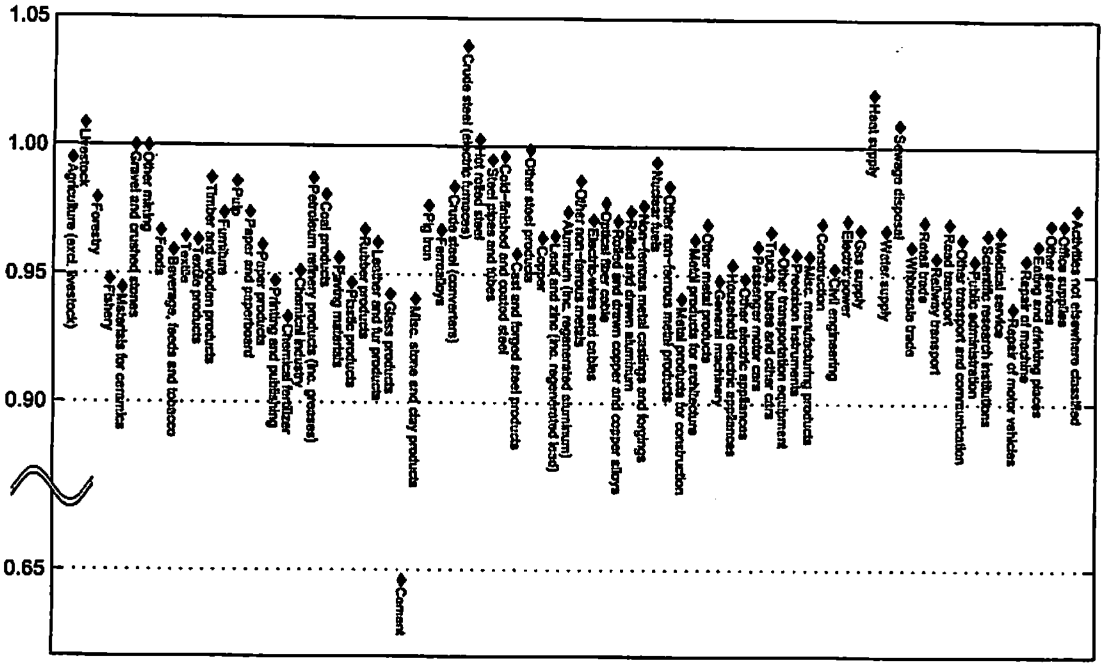

Figure 1: The price level of sectoral output obtained by the WIO-Price model
The dots refer to the price level of the sectoral output on the horizontal bar. Because the price is normalized to unity in the original data, the computed price should be close to unity.

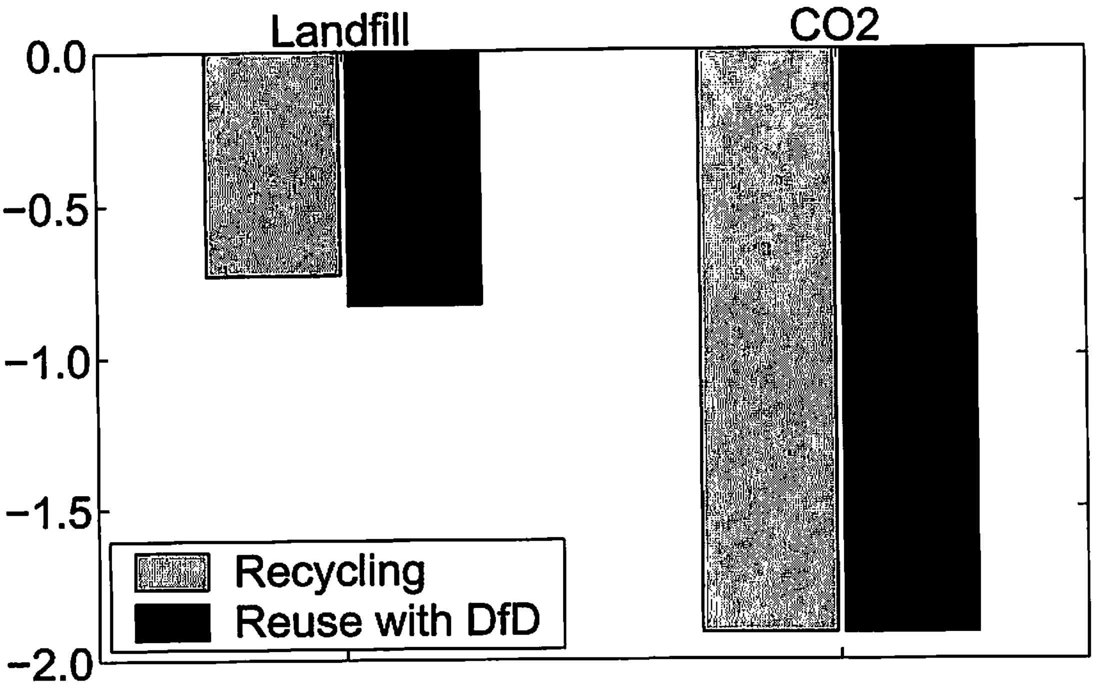

Figure 2: LCI of alternative EL-EHA treatment
Source: Kondo and Nakamura (2003).
The bars refer to the rate of change in percentage relative to the case where EL-EHA are landfilled. "Landfill" refers to the amount of landfilling in weight. "CO2" refers to the emission that results from the burning of fossil fuels, limestone, and the GWP100-equivalents of methane from landfill sites.

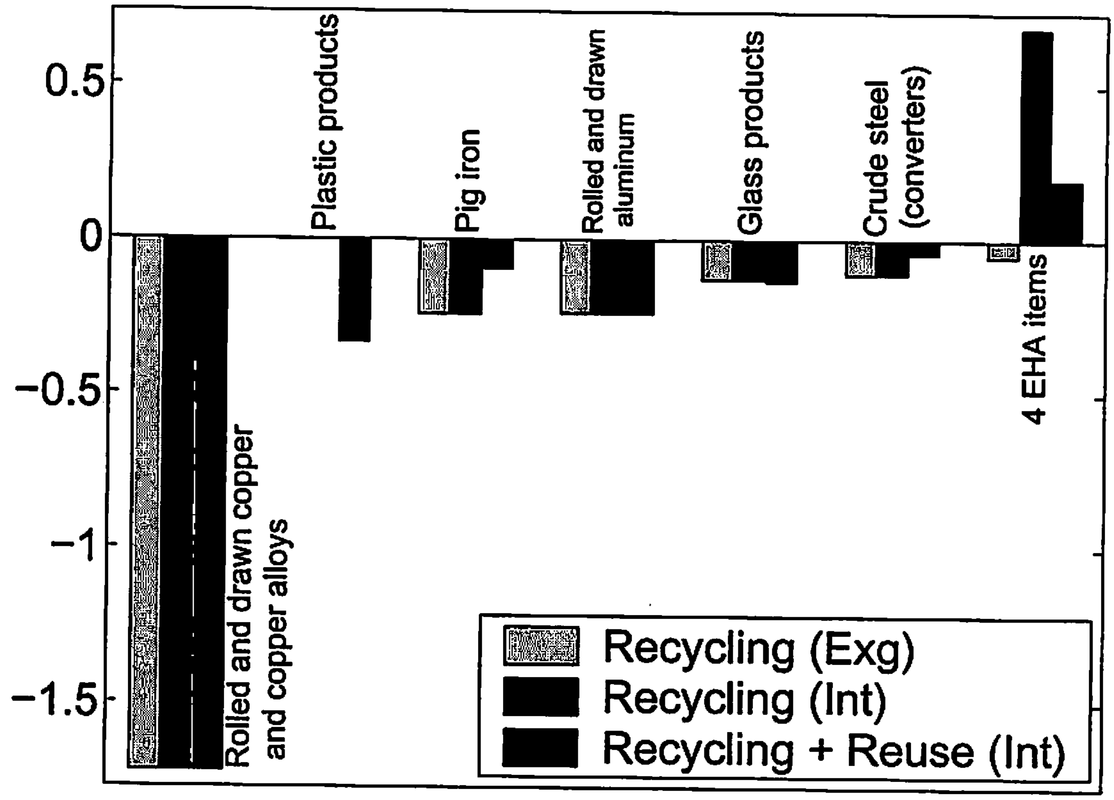

Figure 3: Effects of EL-EHA recycling on the price of goods
Rate of change (in %) in the price of output relative to the case where all EL-EHA are landfilled. "(Exg)" refers to the fact that the cost of recycling is directly paid by the consumers generating EL-EHA, while "(Int)" refers to the case where the recycling cost is internalized in the manufacturing cost.

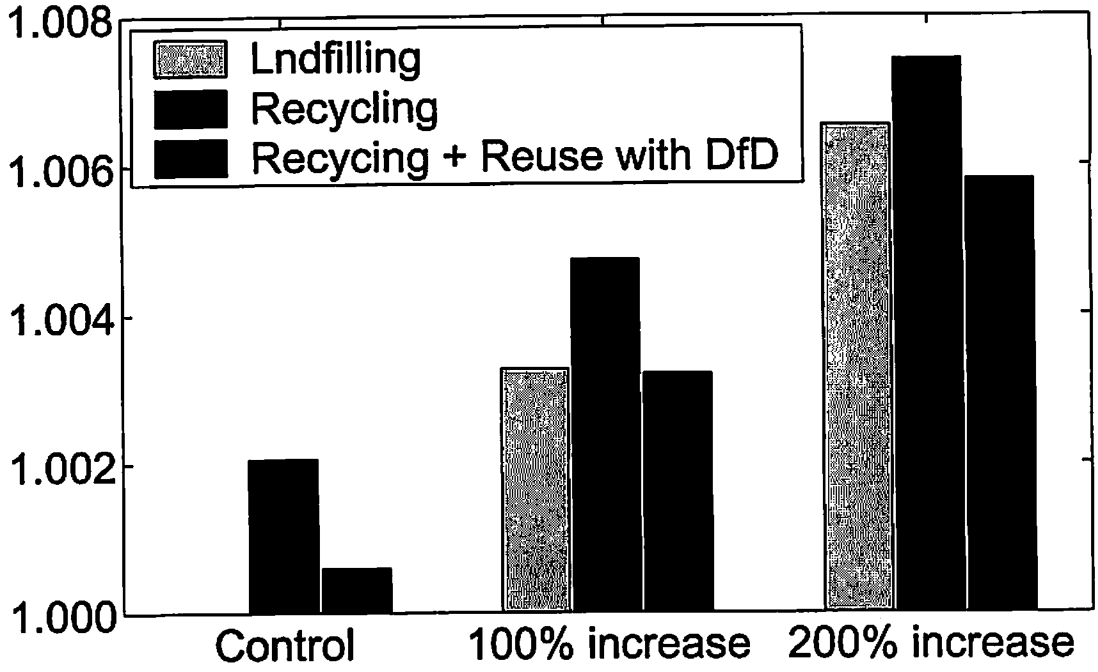

Figure 4: Effects of landfilling cost on the price of EHA
The bars show the average price level of EHA (TV, refrigerator, washing machine, air conditioner) under alternative treatment scenarios and different levels of landfilling fee when the end-of-life cost is internalized in the manufacturing cost. The price level under landfilling scenario evaluated at the 1995 level of landfilling fee serves as reference that is normalized at unity.