# Costs and Benefits of Bioprocesses in Waste Management

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## SYSTEM ANALYSIS OF SOLID WASTE MANAGEMENT

There is an increasingly wide-ranging array of possible approaches to seeking to inform decision-making regarding ‘optimal’ waste management solutions in a given context. Since the implementation of EU Directive 85/337/EWG, an environmental impact assessment (EIA) has been an obligatory element of licensing procedures for certain public or commercial projects. The EIA is required to identify and describe relevant impacts of the project regarding the environmental media and compartments. There are no further EU regulations providing technical guidance on how the EIA is to be carried out. The decisions concerning environmental compatibility or incompatibility are usually made with regard to legal environmental quality standards (EQS). Therefore the EIA normally uses exposure models to infer whether or not the associated emissions lead to the exceeding or otherwise of existing standards. As such, it does not necessarily determine impacts, *per se*, but frequently leads to statements as to whether or not existing limit values are likely to be breached as a consequence of the project’s operation. The EIA method is therefore suitable for assessing site-specific issues in the context of existing legislation, but it does not always assess the damages associated with ‘below limit value’ emissions.

Risk analysis methodologies resemble EIAs in a number of respects. Often an EIA includes elements of a risk analysis when the direct impact of a technique on the surrounding environmental media are estimated. In appropriate cases, specific toxicological aspects are analysed in detail. On the other hand, risk analysis is not standardised as a tool, and the user is free to choose the scope and processing methods. Risk analysis tends also to relate to the risk of specific consequences arising. It is also frequently the case that risk analysis, by its very nature, seeks to quantify things which are not readily quantifiable. This means that frequently, ‘any number’ is deemed better than none, leading to situations in which the boundaries of probabilistic risk and radical uncertainty are crossed in the attempt to arrive at ‘a number’ as the answer to the problem in hand.

Two techniques used with increasing regularity, though apparently by different communities, are life-cycle approaches, and economic valuation. Both have some strengths and weaknesses.

### Life Cycle Approaches

The most standardised environmental assessment methodology is life cycle assessment (LCA). The ISO standards 14040 to 14043 define the basic steps of this instrument. Several subsequent standards illuminate or instruct further technical aspects. During the last ten years, LCA methods have become established as one of the most commonly applied approaches to value systems of production (ecological product balance from cradle to grave) or waste management systems. The methodological approach (defining general system boundaries and balancing the material and energetic inputs and outputs) allows a comparison of systems with varying levels of complexity. In terms of system analysis the LCA approach can be regarded as superior to both the risk analysis and EIA approaches, not least since it does not limit itself to specific limit values, or the risk of certain problems occurring.

Despite these advantages, the following drawbacks are worthy of mention:

- **Weightings in impact assessment categories**
  The weightings attributed to specific pollutants in specific impact categories (e.g. human toxicity) are not universally agreed. These may be said to be subject to continuous review, which is as it should be).

- **'Data Dumping'**
  One frequently finds that life-cycle modelling makes use of data from an array of different sources of differing vintage. As one seeks greater 'completeness' in the data gathering exercise, the temptation inevitably arises to include whatever data is available for even the most obscure auxiliary input. Partly for this reason, one frequently finds that life-cycle studies are carried out using inventory data spanning periods of a decade or so. This is a similar problem to that one finds in issues of risk analysis. It barely needs pointing out that this leads to serious questions concerning the validity of the data, and one has to state quite bluntly that the whole enterprise will always suffer from this type of problem since one is necessarily 'running to stand still' to keep ahead of the 'data gathering' game. Indeed, the environmental optimist would rather hope that the data ***is*** out-of-date (since if it is not, this implies nothing has improved in the intervening years). Although it is sometimes claimed that this should not be a major problem, inspection of the weightings applied in specific impact categories used in LCA suggest otherwise. Where specific inventory items are weighted heavily in the impact assessment categories, the inclusion or absence of these can have significant impacts on the results. Hence, unless inventory data for processes which are the subject of the comparison sets are all measuring the same range of emissions, the use of datasets of differing completeness can lead to bias.
- **Insensitivity to location:**
  The analysis of impacts in life cycle assessment does not (usually) account for location specific impacts. To do so would require location dependent weightings in the impact assessment phase. As such, arguably, it is most useful for assessing global impacts or other impacts unrelated to the populations exposed.
- **Insensitivity to time:**
  Life cycle analyses are frequently insensitive to the issue of time. This means that not only are 'future impacts' weighted equally with current ones, but that slow releases of pollutants over the life-cycle are potentially treated in the same way as rapid releases of the same pollutant. This neglects the fact that pollutant concentrations may be critical in determining impacts, especially where threshold effects re likely to occur.
- **Inability to account for disamenity and other 'impacts' not associated with 'flows of pollutants' / materials:**
  The approach is not sensitive to impacts which are not associated with changes in flows of pollutants. This means that, for example, visual impacts and impacts on biodiversity (through ecological disturbance) are not accounted for.
- **Weighting *across* impact assessment categories:**
  Whilst LCA may be able to assess impacts within categories (however imperfectly) through weighting the effect of specific pollutants in those impact categories, it is relatively weak on assessing how the outcomes in the different impact assessment categories should be traded off against one another. How much 'global warming' is to be equated with 'how many units of human toxicity'?

## Economic Valuation (Benefits Assessment)

Economic valuation approaches usually lean on some of the other approaches, but add to them valuation of effects which are less easily quantified through other approaches, for example, the impact of loss of amenity, as elicited through questionnaire surveys. One apparent benefit of valuation is that all impacts are reduced to a common metric – that of money – so that the last of the problems discussed above in the context of LCA is, supposedly, overcome. But the flip-side is that not all impacts can be readily quantified. Even where they can be, it may be costly to do so, because the methodologies used may demand local surveys to elicit the preferences of those located near a particular activity. In addition, rather like the attempts to 'aggregate' contributions to a specific impact index under LCA, the attempts to do this through economic valuation are far from universally accepted. Some studies have begun to use LCA impact indices as a basis for what economists call 'benefits transfer' (e.g. RDC-Environment and PIRA 2001). This is a questionable exercise. Problems with the approach, therefore, are (for a discussion see Hogg *et al* 2000; Hogg *et al* 2002a):

- **An inability to account for all pollutants**;
  At least in life-cycle studies, a wide range of pollutants can be accounted for, albeit imperfectly. Under valuation studies, the impact of such a range is difficult to assess since credible valuation studies covering such a range of pollutants are not available.
- **The issue of location specificity**;
  Different studies are carried out either using 'bottom up' approaches, which can account for location and the exposure of populations, or 'top down' approaches, which make use of unit values for estimated damages from other studies. Bottom up studies are usually relatively expensive to carry out and involve mapping changes in pollutant concentrations to effects, and then monetising those effects. In top-down studies, the transfer of the valuation figures from one study to another is problematic given the fact that populations exposed are different in different locations. Furthermore, effects such as disamenity are likely to vary in different locations, so they demand location-specific analysis, or at worst, some form of meta-analysis.
- **The issue of time;**
  The issue of time is dealt with in economic valuation studies through use of a process known as discounting.

Future impacts are attributed reduced significance through use of a discount rate, supposedly reflecting preference of society for benefits today rather than in the future (a social discount rate). This is also used effectively to determine how financial resources should be allocated to address problems which have impacts beyond the present. Some LCA practitioners have criticised this process as being inconsistent with basic tenets of sustainability (Helweg 2002).

- **Uncertainty in valuation figures;**
  There are two sources of uncertainty in the valuation figures used in cost-benefit analyses. The first is the nature of the effect upon which a value is being placed. Most environmental impacts are shrouded in a degree of uncertainty, and in some cases, the extent of knowledge is 'radically uncertain'. This is a point which is not always appreciated in such studies, which frequently posit 'unique values' for unit damages associated with a given pollutant (and the same is true, of course, for the way emissions are attributed in life-cycle inventories). The other source of uncertainty is the valuation process itself. There are several different approaches used to value environmental effects. None are completely free of epistemological or moral problems. Given these two sources of uncertainty, the benefits or costs attributed to a given environmental effect typically cover a wide range, usually varying by a factor of ten or so. By way of example, one can readily appreciate that valuing the environment damages associated with the emission of a tonne of carbon dioxide are unlikely to be knowable with any great certainty.

## Selection of System Analysis Methodology

From the above discussion, it seems clear that in order to evaluate the environmental effects of waste management options, one is faced with a variety of approaches, each of which suffers from shortcomings. Perhaps the most important issue in seeking to carry out such analysis is, to state what seems to be obvious, to do the analysis well. Typically, this requires clear statement of limitations of the study, and almost certainly, modesty in arriving at conclusions associated with the study. Sensitivity analysis is clearly to be encouraged. A problem here is that in more complex analyses, sensitivity analysis is time consuming and the results can be laborious to present.

In a recent study, an attempt was made to understand the nature of the costs and benefits of separate collection of compostable materials and their subsequent treatment through composting and anaerobic digestion (Hogg *et al* 2002a). The study looked also at options other than separate collection and treatment, and at the fees paid for the typical waste treatment options. This chapter, however, is principally concerned with the *external* costs and benefits of composting and anaerobic digestion. The full study can be found on the European Commission's website.

# EMISSIONS FROM THE COMPOSTING PROCESS

Full life-cycle analyses of composting processes are difficult to carry out. One of the difficulties associated with the approach to life-cycle analysis regarding composting is the variety of processes available and the fact that different processes make use of different feedstocks. Ideally, as with all treatments, one seeks to relate specific emissions and outputs to the input materials. This has only been attempted seriously in one of the models that were surveyed (the ORWARE model, developed in Sweden). More recent work undertaken in the United States appears to have made a similar attempt (Komilis and Ham 2001).

## Gaseous Emissions

Tables 1 and 2 below show estimates of the gaseous emissions from good practice composting facilities and digestion facilities. The methane estimates from compost plants have been estimated as non-zero. Some would argue for a zero value at best practice facilities. The pollutants mentioned are those to which an economic value has been attributed.

Table 1: Gaseous Emissions Data From Composting Process (all figures are g per tonne of waste composted)

| Gas | g/tonne of waste composted |
|---|---|
| $CO_2$ | 350,000 |
| $CH_4$ | 983 |
| $N_2O$ | 11 |
| Dioxins/furans | 0 |
| Ammonia | 371 |
| VOCs | 24 |

Table 2: Gaseous Emissions Data From Anaerobic Digestion Process (all figures are g per tonne of waste digested)

| Gaseous Compound | **g emitted /tonne digested** |
|---|---|
| $CO_2$ | 440,000 |
| $CH_4$ | 0 |
| $NO_x$ | 10 |
| $N_2O$ | 0 |
| SOx | 2.5 |
| HCl | 0.011 |
| HF | 0.0021 |
| $H_2S$ | 0.033 |
| HC | 0.0023 |
| Halogenated HC and PCBs | 0.00073 |
| Dioxins/furans (TEQ) | 1E-08 |
| Ammonia | ND |
| Cadmium | 9.4E-07 |
| Chromium | 1.1E-07 |
| Lead | 8.5E-07 |
| Mercury | 6.9E-07 |
| Zinc | 1.3E-05 |

The energy used at the plant also contributes to the emissions associated with the composting process. Windrow composting systems seem to use less electricity and more liquid fuel while reactor systems use more electricity and less fuel. Energy use at compost systems might amount to approximately 50kWh per tonne and one litre of diesel per tonne of waste input. For AD systems, there is a utilisation of energy which is more than offset by the generation of energy from the system itself (see Table 3).

**Table 3: Assumptions Concerning Energy Production from Anaerobic Digestion**

| **Parameter** | **Low Value** | **High Value** |
|---|---|---|
| Biogas yield | 70 $m^3$ /t waste | 140 $m^3$ /t waste |
| Percentage methane | 55% | 60% |
| Calorific value of biogas | 385 kWh/t waste | 840 kWh/t waste |
| Electricity generated (30% efficiency) | 116 kWh/t waste | 252 kWh/t waste |
| Electricity for export (net of energy use) | 81 kWh/t waste | 176 kWh/t waste |
| Heat recovered for CHP option (70%) | 189 kWh/t waste | 412 kWh/t waste |
| Heat exported for CHP option (net of energy use) | 151 kWh/t waste | 329 kWh/t waste |

For the calculation of external costs associated with the process itself, the air emissions are the key ones to which costs can readily be attached. The results of the analysis of the external costs of the associated air emissions are shown in Table 4 and Table 5, shown for ‘high’ and ‘low’ values attached to the damage associated with different pollutants. Note that the unit damage costs used for the global warming externalities vary with the discount rate chosen for the economic analysis since the different gases have extended residence times in the atmosphere and their contribution to global warming occurs over an extended period of time. The damage costs associated with energy generation were taken from work undertaken as part of the European Commission’s ExternE programme and are specific to the country being examined in respect of energy generation (European Commission 1999). We have presented only the results for one country, Italy for which the externalities associated with energy generation are broadly representative of an average figure.

Table 4: External Costs Of Gaseous Emissions From Composting Process Including Energy Use (€/tonne)

| | | ITALY | |
|---|---|---|---|
| | | **Low** | **High** |
| Greenhouse Gases | | | |
| | *Carbon Dioxide (process)* | -8.56 | -9.01 |
| | *Methane* | -0.44 | -0.48 |
| | *Nitrous Oxide* | -0.09 | -0.16 |
| Other Air Emissions | | -0.02 | -0.07 |
| Energy Use | | -0.85 | -1.53 |
| Fuel Emissions | | -0.36 | -0.80 |
| Total External Costs | | **-10.33** | **-12.04** |

Table 5: External Costs Of Gaseous Emissions From AD Process Including Energy Generation (€/tonne)

| | | ITALY | |
|---|---|---|---|
| | | **Low** | **High** |
| Greenhouse Gases | | | |
| | *Carbon Dioxide (process)* | -10.76 | -11.33 |
| | *Methane* | 0.00 | 0.00 |
| | *Nitrous Oxide* | 0.00 | -0.71 |
| Other Air Emissions | | -0.08 | -0.13 |
| Avoided External Costs From Energy Generation | | | |
| | *Electricity* | 1.03 | 4.50 |
| | *CHP* | 1.48 | 6.45 |
| Total external costs, no displaced burdens | | **-7.72** | **-12.16** |
| Total external costs, displaced burdens from electricity | *Electricity* | **-9.81** | **-7.66** |
| Total external costs, displaced energy from CHP | *CHP* | **-9.37** | **-6.70** |

Note that the key contributing costs are those from carbon dioxide, energy use, and other air emissions, specifically particulate matter. These deserve some comment. Some studies do not attribute damages to carbon dioxide from compost since it is argued that these are emissions which would have occurred anyway had the material been left to degrade in the natural environment. The argument that this would be part of some natural cycling of carbon is, however, questionable since the materials which are degrading are frequently produced under techniques which require substantial human intervention, for example, paper from monoculture forests, or crops which are heavily dependent on significant quantities of external inputs. Therefore, the emissions are shown explicitly.

## Other Emissions from Biowaste Treatment Processes

Other emissions from the composting process which are of potential concern include:

(1) Leachate
(2) Odours
(3) Microbial pathogens and
(4) Dust.

### *Leachate*

Leachate is not problematic at well-controlled plants, especially where these are covered. Digestion systems may need to have waste water treated and this impose additional costs. It is possible, especially with dry digestion systems, to reduce this requirement if the output digestate is composted along with other (dry) feedstocks.

### *Odours*

Odours are more likely to be a problem at open air windrows and at plants which accept highly fermentable materials such as food wastes. Licensing laws in a number of countries require that plants be located away from dwellings, usually with the aim of reducing odour-related disamenity. Olfactometry tests are increasingly common. Lastly, it is important to note that control of odours has been an active area of research in recent years and problems associated with odours are likely to be significantly reduced relative to previous years. Obviously, the increasing shift to in-vessel technologies quickens this process.

### *Microbial Pathogens and Dust*

In some cases, siting requirements, especially of open air windrow systems, reflect concerns regarding the potential for bioaerosols to cause harm to immuno-suppressed citizens. The debate concerning the impact of bioaerosols is still live, though countries with lengthy experience with composting report few, and in some cases, no health-related effects from their compost systems. Generally, it is felt that problems are more likely to affect plant operators than the general public.

Exposure to any of these problems, to the extent that they are not controlled, will obviously be affected by the proximity of citizens to the plant (so that plant location affects the degree to which any of these issues is likely to lead to problems).

## Disamenity

Since, to the best of our knowledge, no study has looked at the disamenity associated with compost or digestion facilities, there is no basis for quantification. Certainly, these would be expected to deviate more significantly from zero where one or more of the following are true:

(1) The composting process is poorly managed (and odours are prevalent)
(2) The compost plant accepts inappropriate materials, i.e. those which are likely to give rise to problems in the context of the process technology being used
(3) The plant is of a significant scale, so that visual intrusion becomes an issue, as do transport movements. This may imply some double counting where transport externalities are considered; and
(4) The composting process occurs in close proximity to housing.

All of these increase the potential for significant disamenity, though any attempt to estimate disamenity may have to take care to avoid double counting of the effects of odour.

The key difference between composting and digestion facilities relates to the reduced potential for odour-related problems from the process, and to reduced release of bioarosols, at least relative to open-air facilities. Some of the problems mentioned above may still be of potential concern for digestion systems, not least where aerobic composting follows the digestion process. This may be good practice as regards preparation of a stable product for land application.

## BENEFITS FROM COMPOST UTILISATION

Table 6 provides a summary of some of the advantages and disadvantages of using compost compared to mineral fertilisers in agriculture. Similar approaches have been adopted for assessing the benefits from compost and from digestate following a post-digestion aerobic treatment.

The addition of compost to soil results in a reduction in bulk density, an increase in soil porosity and an increased water retention. All these factors have a positive effect on plant growth and subsequent crop yields. They may also act to reduce the potential and / or frequency of flooding in periods of high rainfall, and for soil erosion.

The issue of soil erosion is especially relevant to countries where the soils have a low organic matter content, as is common in Mediterranean countries. Montanarella (1999) estimated that over 150 million hectares of European soils are suffering from erosion, the problem being more acute in southern countries. Soil erosion is a major socio-economic and environmental problem throughout Europe. It reduces the productivity of the land and degrades the performance and effectiveness of ecosystems.

Table 6: Advantages And Disadvantages Of Using Compost Compared To Mineral Fertilisers.

| Material | Advantages | Disadvantages |
|---|---|---|
| Compost | Improves soil structure | Dilute source of nutrients |
| | Controls erosion | Even application can be difficult |
| | Supplies wide range of nutrients | High C:N ratios may rob soil N |
| | Method of waste disposal | |
| | Increases activity of soil micro-organisms | |
| Mineral fertilisers | Convenient | Easily leached |
| | Lower transport and handling costs | Overuse may lead to breakdown of soil structure |
| | Quick crop response | Supply only major nutrients |

## Displacement of Alternative Nutrient Sources

When compost is applied to the soil, it may displace nutrients which are otherwise applied through other means. Typically, it seems reasonable to assume that the product being displaced is synthetic fertiliser. This will not always be true.

The assumption made is that 10 tonnes of dry matter is applied per hectare, which is equivalent to approximately 16.7 tonnes of compost, derived from approximately 47.7 tonnes of waste material. This is based on assumptions that 1 tonne of waste material, consisting both kitchen and garden waste leads to production of 350kg compost with dry matter content of 60%. The modelling further assumes that the compost has the following composition in terms of nutrients:

1. Nitrogen: 1.5% dry matter
2. Phosphorous (as $P_2O5$): 1.0% dry matter
3. Potassium (as $K_2O$): 1.2% dry matter

The mineralisation rate of the nutrients is assumed to be 30% for all nutrients. This determines the time profile of the displacement effect, which in turn effects the external benefits associated with displacement via the discounting mechanism.

For synthetic fertilisers, a loss rate of 23% is assumed for nitrogenous fertilisers (based on Hydro AgriEurope 1995). At the same time, the loss rate from compost is assumed to be negligible given the small proportion of total N which is non-organic N in composts. This means that more nutrient has to be applied in a given year in the synthetic form than would be available in mineralised form from the composted materials. For an application of 10 tonnes dry matter per annum in one year, the N displacement would follow the evolution set out in Table 7. Equivalent projections for P and K displacement have been assumed.

Table 7: Evolution In N Displacement Associated With 10 Tonnes Dry Matter Of Compost Applied To Farmland

| Year | Displacement of N (kg) | Cumulative Displacement |
|---|---|---|
| 1 | 58.4 | 58.4 |
| 2 | 40.9 | 99.4 |
| 3 | 28.6 | 128.0 |
| 4 | 20.0 | 148.0 |
| 5 | 14.0 | 162.1 |
| 6 | 9.8 | 171.9 |
| 7 | 6.9 | 178.8 |
| 8 | 4.8 | 183.6 |
| 9 | 3.4 | 186.9 |
| 10 | 2.4 | 189.3 |

### *Emissions Associated With Fertiliser Manufacture*

The use of compost as a replacement for fertiliser will displace the pollution and other impacts associated with fertiliser production. These are discussed in this section to derive an externality cost per tonne of fertiliser produced and therefore a subsequent benefit per tonne of compost/digestate applied. This study used data for the production of NPK fertiliser because this is the most widespread in terms of use, and the most readily available from a data perspective. No quality data for extractive processes was obtained for use in the analysis. Mining phosphate rock is an energy intensive activity and approximately 3.3 tonnes of phosphate rock are required to produce one tonne of phosphorous pentoxide ($P_2O_5$) (100%) (Bocoum and Labys 1993). Energy use for producing phosphate rock has been estimated at 73.5 kWh/tonne (UNEP and UNIDO 1998). Additional energy consumption for phosphate fertiliser is attributed on this basis. Because no information concerning emissions from potash production were available, the emissions data for $K_2O$ are likely to grossly understate the environmental benefits of displacement effects. It is important to note that these externalities are associated with Best Available Technologies for both new and existing plants. Therefore the costs in terms of gaseous emissions and energy requirements are likely to be underestimated as not all plants will be using Best Available Technologies.

### *Phosphate Fertiliser - Phosphogypsum and Process Wastewater Disposal*

One study sites a major review, undertaken by the US Environmental Protection Agency (EPA), of the production and environmental impacts of phosphoric acid production at 21 locations in the USA (Soulsby *et al* 2000). Significant impacts of process wastewater and phosphogypsum disposal have been identified. The study reports that no economic valuation studies are available in the literature that can provide support in estimating the monetary value of the externalities associated with the processes. However, based on US data, the same study looked at the costs to industry of the environmental regulation involved.

As an admittedly crude estimate of the externalities associated with $P_2O_5$ production which may be avoided when compost is applied, the estimated external costs were taken as equivalent to £59.91 (€95.86) per tonne $P_2O_5$, or €0.096 per kg $P_2O_5$.

## *Greenhouse Gases from Nitrogenous Fertilisers*

Nitrous oxide emissions from soil are complex since the gas is simultaneously produced and consumed in soils through processes of denitrification, nitrification, nitrate dissimilation and nitrate assimilation. The rates at which these processes occur are affected by temperature, moisture, the presence of plants and the soil composition, as well as the (related) activity of bacteria in the soil column.

It is generally accepted that nitrogenous fertilisers increase fluxes of $N_20$. Different fertilisers appear to be more or less susceptible to the loss of nitrogen as nitrous oxide. Ammonia products appear most susceptible, with anhydrous ammonia and aqua ammonia losing between 1 and 5% of nitrogen as nitrous oxide. Other products such as sodium nitrate appear to lose much less nitrogen in this way (Lashof and Tirpak 1990; Erlich 1990). The emissions depend upon temperature, soil moisture, fertiliser type, fertiliser amount, the timing and mode of application, and the type of soil and crop cultivated (McTaggart et al 1998).

In the analysis, it was assumed that in the 'Low' case, 0.5% of nitrogen applied as fertiliser is lost as $N_2O$. In the 'High' case, it was assumed that 2% would be lost. These figures were combined with the N replacement figures for the compost as outlined in earlier sections. For the 3% Discount Rate case, the benefits range from €0.21 – €1.46, a range which arises from combining low emissions with low unit damage costs, and high emissions with high unit damage costs.

## *Effects on Nitrate Leaching from Soil*

Nitrogen from mineral fertiliser is the major source of nitrogen input in the EU. Nitrogen in commercial fertilisers is readily soluble to facilitate uptake by crops, which in conjunction with excessive application can pose a threat to the environment and in some cases affect the fertility of the soil itself. Losses to the environment can be minimised if sustainable agricultural practices are followed and reasoned fertilisation is used. This must account for weather conditions to reduce the incidence of runoff and must involve applying at the appropriate stage of crop growth, using appropriate doses, etc.

Nitrogen when applied as uncomposted animal manures or inorganic nitrogen fertilisers also have the potential to volatise and lose more than 50% of their nitrogen to the atmosphere within the first few days following application to land. When animal manures are spread nitrogen is lost to the atmosphere through volatilisation as ammonia or as the greenhouse gas $N_2O$.

Nitrogen supplied from compost is not immediately available. Approximately 40% is available in the first year following application, 20% in the second year and 10% in the third, slowly decreasing every subsequent year (the rates depend on climate etc.). Therefore, composting when managed correctly is a form of nitrogen conservation. As most of the nitrogen in compost is not in a form that is immediately available to the soil, there is less risk of nitrogen volatilisation and nitrogen leaching. This is especially relevant in nitrate sensitive areas.

Nutrients that are not taken up by plants may be metabolised by micro-organisms in the soil which will improve soil fertility. However this is a slow process and there is a risk that soluble nutrients such as nitrate will run off into surface water or percolate into groundwater reservoirs. Combined, excessive amounts of nitrogen and phosphorous can result in eutrophication in lakes, rivers and coastal areas, resulting in the proliferation of toxic blue-green algae. Soils can also be at risk of eutrophication, where excess nutrients deplete the soil of oxygen, resulting in a reduction of natural microorganisms and subsequent reduction in soil fertility.

The effects of displacing the equivalent quantity of nitrate fertiliser from the soil are not only that one avoids burdens associated with their manufacture. In addition, there is a reduction in the leaching of nitrate into groundwater. Work on economic valuation of nitrate pollution of groundwater is relatively scarce. There are a number of difficulties associated with this, not the least of these being the fact that such leaching as occurs today may only affect people a generation or more in the future. The cost of removing nitrate from groundwater is not insignificant. A recent study in the UK estimates the cost of nitrate removal at £18.8 million per year (approximately €28 million) in capital expenditure in the years 1992-1997 plus £1.7 million operating expenditure (approximately €2.6 million) (Pretty et al 2000). The study estimated that 80% of nitrate originated from agriculture, giving a cost of removal of £16.4 million per annum (approximately €25 million per annum).

Current estimates do not provide a reliable basis upon which to value the 'displaced leaching' associated with the nitrogen displacement. However, the effect should not be ignored.

## Disease Suppression

It has been shown that compost can help to control plant diseases and subsequently reduce crop losses in both agriculture and horticulture (see, for example, Hoitink and Fahy 1986; Ringer et al 1997; DeCeuster and Hoitink 1999). Disease control in compost has been attributed to four main mechanisms. These are;

- Successful competition for nutrients by beneficial micro-organisms.
- Antibiotic production by beneficial micro-organisms.
- Successful predation against pathogens by beneficial micro-organisms.
- Activation of disease resistant genes in plants by composts.

Compost quality is critical to the disease suppressive characteristics of compost. Composts that are allowed to mature have greater suppressive qualities than immature composts.

The beneficial effects of compost use can help growers to save money and reduce their reliance on pesticides, subsequently conserving natural resources. Disease suppressive soils are a well-known phenomenon, especially in organic agriculture. Supressiveness in soils is related to changes in microbial populations which the addition of compost enhances. There is also evidence to suggest that the application of compost can help to control numbers of parasitic nematodes by providing nutrients to the soil that encourage the growth of fungi and bacteria which compete with, or destroy the nematodes.

A number of attempts have been made to estimate in money terms the environmental costs of using pesticides (Pimentel *et al* 1993; Steiner *et al* 1995; James 1995; Foster *et al* 1998; Pretty *et al* 2000). These studies suggest that the external costs associated with pesticide use are not trivial. They lie between 25% and 125% of the total private costs of pesticide use. Here it is assumed that external costs of pesticide use lie between €13- €19 per kilogramme of active ingredient used. On a per hectare basis, in 1993-1995, purchases varied between between 1.2-16.3 kilograms of active ingredient used per hectare of arable and horticultural land across Europe (including set aside). It was assumed that pesticide use could be reduced by an equivalent of 20% of their use on arable land. Clearly this would vary according to the country concerned, the crops being grown on land where compost was applied and, therefore, the pesticides currently in use. On this basis, high and low levels of the external benefits that might be derived from compost use through the avoided external costs of pesticide use were calculated. On this basis, the figure would be equivalent to a benefit of €0.5 - €0.75 per tonne of waste composted for the Italian situation.

These estimates above are subject to considerable uncertainty. However, they represent a first attempt to capture the external benefits associated with reducing pesticide use through the application of compost. Of course, to the extent that other soil management practices change, these will have implications for pesticide use also.

It should be noted that the external benefits from the displacement, whilst they are uncertain, are based on estimates of pesticide externalities that are generally thought to be conservative. No account has been taken of the avoided external costs of the production of pesticides which might otherwise have been used. This is the exact opposite of the situation in respect of fertilisers where the attempt was made to capture the external benefits from avoiding production, but the avoided external costs of, for example, nitrate leaching following application were not estimated. It should also be noted that applications of pesticide vary across crops. Horticultural applications are typically associated with higher applications of pesticides. To the extent that composts are used in such situations, benefits may again be an underestimate, notwithstanding the somewhat arbitrary nature of the assumption regarding the proportion of pesticides displaced.

## Greenhouse Gas Emissions from Compost In Soil

The use of compost in agriculture can have a positive effect on soil carbon levels and subsequently act as a carbon reservoir. Compost does not result in the permanent and irreversible locking up of all carbon in soil. What compost can do is reverse the decline in soil organic matter which has occurred in relatively recent decades through contributing to the stable organic fraction in soils (effectively locking-up carbon). It is also important to realise that whilst the debate concerning 'sequestration' has emerged as a topical one in the wake of the debate on climate change, the role played by soil organic carbon is far more complex, and potentially far more important, than the single role played in terms of carbon sequestration. It is clear, for example, that the effects of soil organic matter on soil biota are at the heart of the disease suppressing effects of compost. The interrelationship between carbon and nitrogen largely determines the magnitude of soil microbial populations. Utilisation of carbon and nitrogen by microbes is also responsible for the turnover between organic and mineral forms of nitrogen. Hence, the biomass production potential of soil is largely dependent upon the ability of a soil to support microbes such as bacteria and fungi.

Three pools of organic carbon are available for microbial utilisation:

(1) The active soil fraction:
turnover of around two years, and representing short-term sequestration of carbon – provides source of energy for microbes, and soil carbon and nitrogen supply necessary components for amino acid synthesis

(2) The slow or decomposable soil fraction
turnover time two to three years – of great importance to developing good soil structure – disturbed by cultivation and other disturbances – provides a source of carbon for biological digestion by microbes, so linking to the active pool. – it can be viewed as mature compost
(3) The passive soil organic fraction:
turnover time of order 1000 years - resistant to oxidation processes – acts as a 'cement' that binds particles.

Only the first two of these pools contain carbon in readily available forms for microbial utilisation. The last pool contains carbon in a highly stable form. Some microbes can utilise this pool so depletion does occur. It can also be replenished from active and slowly decomposable fractions. It is the fact that this passive pool of carbon can be maintained or increased that leads to the idea that the passive pool can act to 'sequester' carbon in the soil. Clearly, this long turnover time does appear to imply that, for all intents and purposes (certainly for any economic analysis deploying a non-zero discount rate), this carbon is not released into the atmosphere.

The dynamics of soil organic carbon where it is applied in composted form have been modelled. The pathways modelled are outlined in Figure 1 below. The application of compost is assumed to lead to the readily available carbon being mineralised at y% whilst x% of the readily available organic carbon is converted to stable organic matter. Of this stable organic matter, some carbon is mineralised, but at a much lower rate than that at which the readily available matter is converted to stable organic matter.

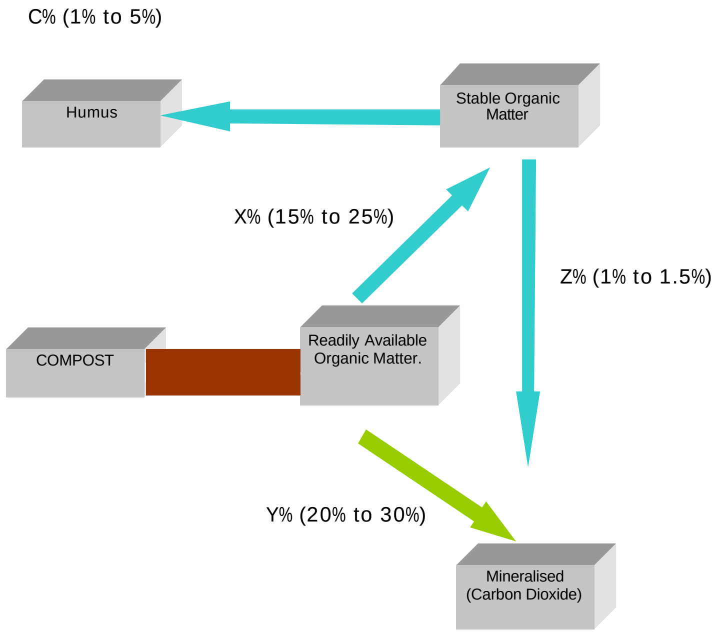


Figure 1: Basic Description Of Modelling Of Fate Of Carbon In Compost / Soil

This means that as compost is applied to the soil, there is a build up of organic matter in the soil since the rate at which the material is mineralised is, overall, much slower than the rate at which the material is applied. Eventually, the rate of loss of the remaining organic matter equals the rate of application so that the application of compost at different rates each year tends to take the soil organic matter, asymptotically, towards an equilibrium concentration of organic matter. In practice, this equilibrium level will depend upon other factors such as soil cultivation practices, and specifically, the attitude towards tillage of the soil. Already, in North America, one company (TransAlta) has looked at the possibility of securing carbon activities against the resort by farmers to no-till farming practices on the basis that this reduces the rate at which soil organic carbon is mineralised into the atmosphere.

Using the figures X = 25%, Y = 20%, Z = 1%, and with an initial organic matter concentration of 2%, one can plot the effects of different rates of compost application. This is shown in Figure 2.

The production of compost and incorporation in topsoil has the potential to act as a reservoir for carbon. When combined with responsible agricultural practices it could have a positive impact on reducing the rate of global warming. However, as compost is applied, the processes of mineralisation do lead to releases of carbon dioxide. The progressive loss of soil organic matter over time following one year's application of compost suggests that after 50 years, some 13% of the original carbon applied still remains in the soil. This implies a potential to 'buy time' whilst other measures in respect of climate change mitigation measures are allowed to take effect.

Note that the combustion of waste to generate energy, whilst it may appear superficially attractive, is a poor idea where organic materials are concerned. These have low calorific value, and there is a suggestion that, for example, if these were combusted alone, their high moisture content / low calorific value would make energy recovery impossible using current technologies.

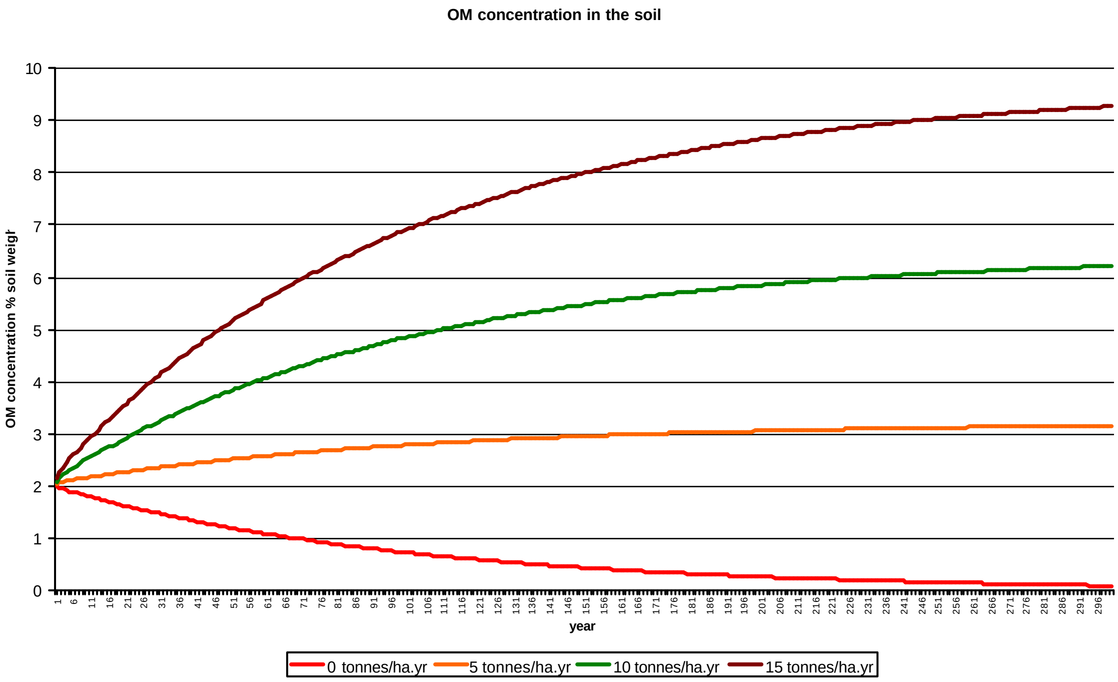


Figure 2: Effect Of Different Rates Of Compost Application On Soil Organic Matter Levels

## Compost as an Alternative to Peat

Compost may, in some countries, substitute for peat in pot cultivation (and home gardening in some countries). The main use of peat in amateur gardening is a soil improver. Compost can be used as a substitute for peat as a soil improver. It performs well and is likely to be accepted by the public as an alternative to peat. However there are several barriers to overcome before this is the case. Retailers and consumers must be assured that compost is of an acceptable quality (fit for purpose) and that there is uniformity between batches.

The other main use of peat is as the main constituent in growing media. Growers need to be assured that compost is free from contaminants and that it will not have any phyto-toxic effects. These assurances need to be given based on substantiated research and adequate quality and user guidelines.

### *Environmental Costs of Peat Extraction*

The environmental costs of peat extraction are difficult to quantify in monetary terms. The primary losses to the environment through peat extraction are:

- Loss of biodiversity;
- Loss of landscape and recreational value;
- Loss of palaeoecological and archaeological value; and
- Increased carbon emissions loss of carbon reservoirs.

**Loss of Biodiversity**

Many rare and protected species thrive in Europe's peatlands and bogs. The bog moss *Sphagnum imbricatum* is entirely restricted to bogs and is the principal peat forming species in oceanic peatlands. It is becoming increasingly rare as more sites are being developed. There is also the loss of rare and unique plants which have potential medicinal properties. These benefits are difficult to value although studies concerning biodiversity loss have reported high values reflecting the willingness of citizens to pay for conservation.

**Loss of Landscape and Recreational Value**

In Europe areas of peatlands and bogs have a cultural importance as some of the last true remaining wilderness areas. They attract visitors for this reason. Travel-cost and contingent valuation studies capture consumer surplus associated with, and preferences for, respectively, the continued existence of these landscapes. One study in the UK translates values for the Somerset Levels into a value of £7,245 per hectare (Ingham 1996, citing Willis et al 1993), equivalent to approximately €12,200 per hectare in current terms. Another UK study estimated a preservation value of £68.4 million, or £4.1 million per annum using a 6% discount rate (Hanley and Craig 1991), equivalent to approximately €116 million in total, or €7 million per annum

**Loss of Palaeoecological and Archaeological Value**

Peatlands and bogs contain a rich archive of information about our history. Examination of peatlands provides an insight into past climates, culture and economy. These non-use benefits of peat are lost once they are developed for exploitation.

**Greenhouse Gas Emissions (Loss of Carbon Reservoirs)**

The greenhouse gas emissions associated with peatbogs are extremely complex, and they change once the process of development (for extraction) occurs. In northern peatlands, the anaerobic conditions and cold temperature result in increased sequestration of carbon (relative to other wetlands). Wetlands store carbon in short- and long-term reservoirs. Storage occurs when primary production is high and exceeds the rate of decomposition, or when the rate of decomposition is slowed by a process known as anoxia, and cold temperatures (leading to accumulation of undecomposed organic matter). Unperturbed peatbogs, whilst they may act as a sink for carbon, may also emit methane. However, as long as they are unperturbed, they most likely retain a balance between methane emissions and carbon sequestration.

Drainage and degradation of peatlands increases carbon dioxide emissions. It also increases nitrous oxide emissions significantly (Roulet *et al* 1993; Regina *et al* 1998; Freeman *et al* 1993). Wetlands store carbon in short- and long-term reservoirs. Storage occurs when primary production is high and exceeds the rate of decomposition, or when the rate of decomposition is slowed by a process known as anoxia, and cold temperatures (leading to accumulation of undecomposed organic matter). It has been estimated that peatlands contain between 329 and 528 billion tonnes of carbon (equivalent to 1,200-1,900 billion tonnes of carbon dioxide). Unless the bogs are disturbed by extraction, drainage or other human intervention, much of the carbon will remain in-situ for near geological timescales.

Drainage of peatlands and other wetlands acting as carbon reservoirs will result in the oxidation of the organic matter, releasing it to the atmosphere as carbon dioxide, methane and other greenhouse gases. Conversely, restoration or creation of new wetlands may provide additional carbon sinks (Environment Canada 1998).

### *Quantification*

The use of peat would seem to incur considerable environmental costs. Although these can be captured in various ways, it is very difficult to impute an environmental cost per tonne of peat extracted. Data was taken from the Finnish life cycle study used in the ExternE National Implementation study (European Commission 1999). This was adapted since that study, which covers emissions from various aspects of the peat fuel life cycle, includes emissions from power production and from restoration. The former are subtracted from the data for obvious reasons. The latter are subtracted because at non-zero discount rates, the effects of the restoration ‘savings’ are likely to be relatively small.

These gaseous emissions are used as the basis for the external cost savings from compost use where it displaces peat. Peat is replaced by compost more on the basis of volume than on weight. The density of peat is low, estimated here at 200kg/m3. The density of compost, on the other hand, is of the order 500kg/m3 for a compost with dry matter content 60%. This implies that to replace one tonne of peat would require compost resulting from 7.14 tonnes of waste material.

The external cost savings would appear to be a significant under-estimate of the avoided external costs to the extent that pressure to develop new peatbogs is reduced. This is due to the fact that the non-use values of peatbogs appear to be significant.

## Other Potential Benefits

There are a number of other potential benefits from the application of compost. Some are discussed below, but no valuation data were available for inclusion in the study.

### *Reduced Requirement for Liming*

Since compost acts as a buffer against falling pH, there is likely to be a reduced requirement for liming, which might be required to offset the increased acidity of the soil. The extraction of lime implies quarrying for the material, though sometimes, industrial by-products are applied in this context.

### *Reduced Susceptibility to Soil Erosion*

The condition of a soil surface determines whether rainfall infiltrates the soil or simply runs off. Soil therefore regulates and partitions water. When water runs off land, it tends to carry soil particles. This results in costs to farms in terms of lost productivity and off-farm impacts ranging from damage to commercial and recreational fishing, increased pressure on water treatment facilities, increased flood damages and requirement for repairs from redredging damaged waterways.

It is increasingly recognised that off-farm costs of soil erosion are probably greater than on farm ones. The off-farm costs associated with soil erosion in the US due to waterways alone were estimated at $2-$17 billion (National Research Council 1989; Ribaudo 1989; Pimentel *et al* 1995). In the UK, a 1996 study estimated soil erosion impacts at between £23.8 - £50.9 million (1991 prices) with off-farm losses responsible for as much as 80% of this figure (Evans 1996). In future, severe storms may generate the bulk of soil erosion losses, and this may be a possible ‘positive feedback’ associated with global warming in the future. Air-borne soil particles may also have impacts on human health, and their presence could be reduced through greater use of organic matter to bind soil into stable aggregates.

Management factors play a role in reducing erosion, but so also does the soil texture and organic matter content. It is difficult (for obvious reasons) to estimate the incremental reduction in soil erosion associated with applications of compost. However, the benefits associated with reduced soil erosion are significant. Not quantification was attempted.

### *Benefits from Improved Infiltration*

Benefits from improved infiltration of water arise through reduced risk of flooding (and soil erosion – see above) and reduced requirement for irrigation water. No quantification has been carried out.

### *Reduced Irrigation Requirement*

Water holding capacity can be increased by as much as 3-5% through application of soil organic matter. The avoided environmental burden is difficult to assess at the margin, though it is possible to place values upon water in specific contexts. Indeed, many argue for the use of tradable permits as an allocation mechanism to ensure efficient use of water. Other benefits relate to the increased survival rates of unmanaged young trees in dry periods.

It is difficult to quantify these savings, either in terms of environmental benefits or financial savings, because of the varying nature of the demand for water for agriculture across countries. The financial savings to be realised depend very much on the charging regime for water. In many countries, water for agriculture is still made available on a flat fee or per hectare basis. Consequently, there is no marginal benefit to be gained from reduced consumption. Such savings may become more important in the future, however, as it is likely that more and more countries will move towards marginal cost pricing for water resources.

### *Improved Tilth*

The fact that compost improves soil structure means that it is actually easier to work the soil with agricultural machinery. There are likely to be savings in fuel use resulting from this change in soil quality. No attempt has been made to quantify these.

## EXTERNAL BENEFITS OF COMPOST APPLICATION

Putting the picture together suggests that the external benefits of compost application (and energy recovery in the case of digestion) can offset, though not completely, the external costs of composting and digestion (see Table 8 and Table 9).

It is an open question as to whether a 'complete' analysis, if it were possible, would show positive numbers, or whether the numbers would remain negative. It is the view of this author, however, that, when compared with other waste management options, the unquantifiable external costs are likely to be less significant than those unquantifiable external costs for the other principal waste management options. At the same time, the benefits of energy recovery techniques (energy itself) are relatively well characterised. The benefits of compost utilisation are less well understood and less amenable to ready quantification. This work should be seen, therefore, as an early attempt to point the way forward to more complete assessments of costs and benefits of composting and digestion processes.

Table 8: Net Costs And Benefits Of Compost Treatment And Application (in € per tonne of waste).

| | ITALY | |
|---|---|---|
| | **Low** | **High** |
| Greenhouse Gases | | |
| *Carbon Dioxide (process)* | -8.56 | -9.01 |
| *Carbon Dioxide (post application)* | -3.13 | -3.30 |
| *Methane* | -0.44 | -0.48 |
| *Nitrous Oxide* | -0.09 | -0.16 |
| Other Air Emissions | -0.02 | -0.07 |
| Energy Use (kWh) | -0.85 | -1.53 |
| Fuel Emissions (litres) | -0.36 | -0.80 |
| Total External Costs | **-13.46** | **-15.34** |
| External Benefits from Nutrient Displacement | 0.13 | 1.74 |
| External Benefits from Pesticide Reduction | 0.40 | 0.60 |
| External Benefits from avoided nitrous oxide emissions | 0.04 | 0.65 |
| External Benefits from avoided process wastewater disposal | 0.01 | 0.03 |
| External Benefits from avoided peat extraction | 0.36 | 0.52 |
| Net Externality | **-12.52** | **-11.80** |

Table 9: Net Costs And Benefits Of AD Treatment And Compost Application (in € per tonne of waste).

| | ITALY Low | High |
|---|---|---|
| Greenhouse Gases | | |
| *Carbon Dioxide (process)* | -10.76 | -11.33 |
| *Carbon Dioxide (post application)* | -3.13 | -3.30 |
| *Methane* | 0.00 | 0.00 |
| *Nitrous Oxide* | 0.00 | -0.71 |
| Other Air Emissions | -0.08 | -0.18 |
| Avoided External Costs From Energy Generation | | |
| *Electricity* | 1.38 | 5.38 |
| *CHP* | 1.97 | 7.72 |
| Total external costs, no displaced burdens | **-13.97** | **-15.51** |
| Total external costs, displaced burdens from electricity | **-12.59** | **-10.13** |
| Total external costs, displaced energy from CHP | **-12.00** | **-7.79** |
| External Benefits from Nutrient Displacement | 0.12 | 1.67 |
| External Benefits from Pesticide Reduction | 0.40 | 0.60 |
| External Benefits from avoided nitrous oxide emissions | 0.04 | 0.65 |
| External Benefits from avoided process wastewater disposal | 0.01 | 0.02 |
| External Benefits from avoided peat extraction | 0.36 | 0.52 |
| Total external costs, no displaced burdens | -13.05 | -12.05 |
| Total external costs, displaced burdens from electricity | -11.67 | -6.67 |
| Total external costs, displaced energy from CHP | -11.08 | -4.33 |

## Pricing of Compost Products

There is an interesting question as to whether the private savings which arise from the use of compost should be considered in an analysis of 'external costs and benefits' of the use of compost. Theoretically, and in many cases, in practice also, such savings should not be included. People who buy compost would be expected to 'internalise' such savings in their decision as to whether or not to purchase the product. In this case, these benefits would be internalised in the market price for compost.

However, there may be effects which arise from the use of compost which might not be obviously attributable to the use of compost itself. A good example would be the disease suppressing effect of compost. Reduced outlay on pesticides might not necessarily be linked to the application of compost, not least since it is not easy to know what the counterfactual scenario would have been. Arguably, the more people understand the benefits associated with compost, the smaller is the justification for considering the private savings as 'an external benefit'. In the world of perfect information, the effect is internalised in the decision making process (and arguably, the market for compost would improve where such benefits were understood and realised). However, for completeness, we estimate them here (see **Fehler! Ungültiger Eigenverweis auf Textmarke.**). If one were to include estimates of savings in terms of the 'fertilisers and pesticides avoided', the picture would appear more positive.

Table 10: Private Savings Associated with Avoided Pesticide and Fertiliser Use

| **Memorandum Items** | **Compost** | | **AD** | |
|---|---|---|---|---|
| | Low | High | Low | High |
| Private savings from avoided fertiliser use | 0.77 | 1.93 | 0.58 | 1.45 |
| Private savings from avoided pesticide use | 1.00 | 1.00 | 1.00 | 1.00 |

## Composting Versus Other Waste Treatments

Note that these externalities, though negative, are less negative than those of, for example, incineration. Though the estimation of external costs is an incomplete and uncertain science where waste treatments are concerned, there are good reasons to suppose that composting and digestion would remain superior in an 'ideal' analysis. This relates to the fact that the majority of emissions from compost appear to be relatively benign. Those from incinerators, on the other hand, are not, and indeed, whilst the effects of some are subject to scientific debate (as with some emissions from compost), the effects of others – particulates, dioxins, sulphurous oxides, oxides of nitrogen – on human health are relatively well-established.

Two key observations appear to come from this study:

(1) First, the potential benefits from compost are wide-ranging and include benefits which are only relatively poorly understood at present. As a material for use in sustainable, knowledge-intensive (as oppose to 'synthetic external-input intensive' agriculture), compost may have an important role to play.

(2) Second, composting as a waste (resource?) treatment method for dealing with biodegradable materials, implies some negative externalities, but these are rarely 'large ones'. Other non-landfill waste treatment technologies have been shown to generate negative externalities of a potentially more significant magnitude (as well as, in the case of incinerators, hazardous waste materials). Judged by this criterion, the process is relatively benign.

## Separate Collection of Biowastes

Consequently, the rationale for encouraging effective source separation of biodegradable wastes seems relatively strong. This is particularly true in Southern Europe where the need for organic material in soil is of paramount importance. However, temperate zones are also losing organic matter from the soil. Maintaining soil health and quality in the context of a simultaneous movement towards these twin goals suggests the wisdom of collecting source separated organic wastes for the creation of quality composts.

Further support for a strategy focused on source separation is to be found in the fact that the separate collection of compostable materials does not necessarily incur significant additional costs over standard refuse collection. This depends upon the specifics of the collection system, but moving to less frequent refuse collection enables the costs of systems collecting compostables separately to be constrained at levels similar to those where no separate collection occurs.

Hence, the private costs of separate collection and composting / digestion can be quite favourable, depending upon the prevailing regime governing the costs of other waste treatment options. In countries such as Austria, Netherlands, Belgium (Flanders), and Germany, separate collection is well-advanced. It should come as no surprise to learn that it is in these countries where the private costs of incineration are relatively high (because of the regulatory demands on flue gas emissions, the costs of treating ash residues, and the more limited support for 'renewable energy' from incineration) and where there are constraints on landfilling of untreated waste (either through bans on landfilling in Flanders and Netherlands, or requiring pre-treatment through mechanical biological treatment in Germany and Austria). Hence, there is an argument to be advanced that if the (regulatory and economic) regime for determining the costs of other waste treatment options were more harmonised across the EU, separate collection of compostables would flow naturally as a consequence of the fact that the alternatives to separate collection and composting would be a higher cost combination of refuse collection and more expensive residual waste treatment.

Note that this chapter has not considered the significance of the issue of standards, though this has been addressed elsewhere (Hogg *et al* 2002b). It is clear that standards, to the extent that they draw distinctions between wastes and products, encourage the use of biowaste treatment for the treatment of wastes which are separated at source for the development of quality products. Quality assurance schemes can help to support such standards in developing the market for quality composts derived from source separated biowastes.

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