Public Meeting
CEQA- Composting General Order NOP
Deadline: 9/6/13 by 12 noon

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SWRCB Clerk

September 06, 2013

Ms. Jeanine Townsend
Clerk to the Board
State Water Resources Control Board
1001 I Street, 24th Floor
Sacramento, CA 95814
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Facsimile: 916-341-5620

Re: Comment Letter: CEQA – Composting General Order NOP

Dear Ms. Townsend:

We appreciate this opportunity to submit comments on the State Water Resources Control Board's ("State Board") Notice of Preparation for the upcoming Environmental Impact Report ("EIR") to evaluate the Board's proposal to adopt General Waste Discharge Requirements for Composting Operations ("General Order"). We urge that the EIR include:

- A thorough and systematic evaluation of the significant environmental impacts that could occur if organic materials are diverted from composting operations to disposal in landfills or direct land application, as a result of the adoption of the General Order; and

- A thorough evaluation of the alternatives that would mitigate these significant impacts, including a meaningful discussion of the comparative environmental detriments and benefits of implementing the no action alternative.

We submitted a letter to the State Board on October 9, 2012, in connection with the Board's prior version of the proposed General Order, detailing the potentially significant environmental impacts that could occur. We request the Board include that letter and its attachments in the administrative record for the EIR.

Recology manages the spectrum of municipal waste, focusing on recycling and composting services that span the needs of urban, suburban, and rural communities. Recology is recognized as an innovative leader in the field. Recology is dedicated to the science and practice of resource recovery, which is reclaiming materials that traditionally were viewed as waste and transforming them into the raw inputs used to create new and useful products. Recology strives to make the best and highest use of all resources and reduce negative impacts on the environment within the many communities it serves. Recology, therefore, is continually striving to increase the diversion of waste from landfills though environmentally beneficial practices such as composting and recycling. Through its composting facilities, Recology provides a sustainable organics infrastructure that "closes the loop" and promotes the beneficial reuse of materials that otherwise would be disposed of as waste.

Californians Against Waste is a statewide membership-based public interest environmental organization representing over 5,000 active members. As an environmental organization, Californians Against Waste strongly supports efforts to appropriately manage and regulate all decomposition of organic materials to achieve the highest and best use of these materials and protect public health and the environment.

We understand the State Board's interest in moving forward with uniform statewide standards for composting facilities. However, we are concerned about the negative impacts on existing and future composting operations that could result from the adoption of overly burdensome requirements that are not appropriately tailored to address the environmental risks and benefits at issue. In particular, if the standards in the General Order are not carefully crafted, they could reduce the number of composting facilities and the amount of compost production throughout the State. This would result in increased disposal of organic materials in landfills or direct land application of this material, which, in turn, would cause significant air quality, water quality, and other environmental impacts.

Scientific analyses by the California Air Resources Board and the United States Environmental Protection Agency show that landfill disposal of organic materials generates significantly greater greenhouse gas emissions than composting operations. Moreover, composting provides additional environmental benefits by minimizing the use of chemical fertilizers and pesticides; improving soil conditions to lessen erosion and runoff; reducing agricultural energy and water usage; minimizing the consumption of limited landfill capacity; and reducing the need for long-haul truck trips to transport organic materials to landfills.

This evidence demonstrates the importance of a thorough evaluation in the EIR of the significant environmental impacts the adoption of a statewide General Order could cause, as well as the need to analyze alternatives that would reduce those impacts to less than significant.

A. Negative Impacts on Composting Operations

In the fall of 2012, a number of public agencies and organizations submitted correspondence to the State Board on the prior version of the General Order expressing significant concerns about the negative effects on composting facilities across the State that would be caused by the imposition of excessive regulatory burdens. In a letter to the State Board, dated September 12, 2012, the California Department of Resources Recycling and Recovery ("CalRecycle") − the state agency charged with overseeing California's vital efforts to reduce, recycle, and reuse the waste it generates − emphasized the danger that overly stringent requirements imposed by the State Board would "suppress the growth of new composting facilities and may cause some existing facilities to go out of business."

CalRecycle expressed doubts as to whether composters could raise their rates to cover the increased costs without losing organic feedstocks to landfilling, land application, or even illegal dumping. CalRecycle also emphasized the critical need to expand, rather than hinder, the State's composting infrastructure to meet the 75 percent source reduction, recycling, and composting goal of AB 341.

Echoing CalRecycle's concerns, a coalition of public agencies and private companies representing owners and operators of composting facilities across California, in a letter to the State Board dated September 12, 2012, cautioned that onerous statewide requirements could result in a decrease in statewide compost capacity as facilities close or downsize in the face of significantly increased costs and regulatory obstacles.

It is clear this regulation will have a direct impact on the economic viability of existing composting facilities in the State, and it will hinder the development of new composting facilities. Since this regulation does not, in any way, reduce the amount of organic waste that is being generated in the State, it would result in more material being landfilled or directly applied to the land.

These widespread concerns highlight the need for the State Board's EIR to address the negative impacts the General Order could have on composting operations and the resulting adverse impacts on the physical environment.

B. The Resulting Adverse Impacts on the Physical Environment

1. Greenhouse Gas Emissions

It is well recognized that composting reduces greenhouse gas emissions, as compared with disposing of organic materials in a landfill. As the California Air Resources Board ("CARB") points out on its webpage on composting:

> Over 25 percent or approximately 10 million tons of organics are sent to landfills each year. The anaerobic decomposition of these wastes results in the emission of methane (a greenhouse gas). Composting of organic waste material has become an important method of managing California's solid waste stream. Composting diverts biomass residue from landfills. ***This reduces the need for landfill capacity and the production of GHG emissions.*** (Emphasis added.)[1]

As a result, CARB explains that it adopted a measure in its Climate Change Scoping Plan that specifically commits CARB staff to work with CalRecycle and other state agencies to provide direct incentives for the use of compost in agriculture and landscaping.

In turn, the Climate Change Scoping Plan explains that composting helps to reduce GHG emissions in several ways. First, methane emissions from landfills "can be substantially reduced by properly managing all materials to minimize the generation of waste, maximize the diversion from landfills, and manage them to their highest and best use." Second, "by recovering organic materials from the waste stream, and having a vibrant composting and organic materials industry, there is an opportunity to further reduce greenhouse gas emissions through the indirect benefits associated with the reduced need for water and fertilizer for California's agricultural sector."[2]

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[1] See Attachment 1 (printout of http://www.arb.ca.gov/cc/compost/compost.htm).

[2] See Attachment 2 (excerpt from CARB's *Climate Change Scoping Plan* (Dec. 2008), at pp. 62-63, available at http://www.arb.ca.gov/cc/scopingplan/document/scopingplandocument.htm).

The U.S. EPA similarly explains the benefits of composting on its website, stating that "[c]omposting organic materials that have been diverted from landfills ultimately avoids the production of methane…"[3] The EPA further states: "Food scraps and yard waste make up 20-30% of the waste stream. Making compost keeps these materials out of landfills, where they take up precious space and release methane, a greenhouse gas 21 times more potent than carbon dioxide emissions in the atmosphere."[4]

As an illustrative example, the EPA notes that a composting program in Massachusetts run by the Center for Ecology Technology achieved significant greenhouse gas reductions by reusing organic wastes instead of disposing of them in a landfill. Specifically, the program achieved greenhouse gas reductions of 5,700 metric tons of carbon dioxide equivalent gases from the program's inception in 1996 to 2000 – an amount comparable to the amount of carbon that would be sequestered by 6,333 acres of five-year-old trees.[5]

The greenhouse gas benefits from composting operations over an extended period of time throughout California would be significantly greater. Indeed, even by conservative estimates, California could reduce its greenhouse gas emissions by one million metric tons of carbon dioxide equivalent gases by composting just 30% of the foodwaste that is currently disposed. That is equivalent to the carbon sequestered by 26 million tree seedlings grown for 10 years.[6]

The government of New South Wales, Australia likewise has concluded that composting "offers significant benefits to the environment, including substantial reduction in greenhouse gas emissions."[7] This conclusion is based on extensive scientific analysis.[8]

Indeed, there are many greenhouse gas benefits from composting organic materials, as compared with disposing of these materials in a landfill. These benefits include:

- Reducing methane emissions from landfills.

- Decreasing the need for chemical fertilizers and pesticides, thereby reducing greenhouse gas emissions from the use of fossil fuels associated with their production and application.

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[3] See Attachment 3 (printouts of http://www.epa.gov/composting/basic.htm and http://www.epa.gov/composting/benefits.htm).

[4] See Attachment 4 (U.S. EPA, *Backyard Composting: It's Only Natural* (Oct. 2009), available at http://www.epa.gov/composting/pubs/index.htm).

[5] See Attachment 5 (U.S. EPA, *Success Story: Turning Garbage into Gold* (July 2002), available at http://www.epa.gov/composting/pubs/index.htm).

[6] See Attachment 6 (printout of http://www.cawrecycles.org/issues/ghg/compost).

[7] See Attachment 7 (New South Wales Department of Environment and Conservation, *Fact Sheet: Organics Recycling Offers Major Environmental Benefits* (Mar. 2004, revised Nov. 2007), available at http://www.epa.nsw.gov.au/warr/recycledorganicspublications.htm).

[8] See Attachment 8 (New South Wales Department of Environment, Climate Change and Water, *Short Report: The Benefits of Using Compost for Mitigating Climate Change* (Feb. 2011), available at http://www.epa.nsw.gov.au/warr/recycledorganicspublications.htm); & Attachment 9 (New South Wales Department of Environment and Conservation & University of New South Wales, *Life Cycle Inventory and Life Cycle Assessment for Windrow Composting Systems* (2d ed. 2007), available at http://www.recycledorganics.com/publications/reports/lca/lca.htm).

- Allowing more rapid growth in plants, thereby increasing carbon uptake and storage within the plant. This is a form of carbon sequestration that removes CO_2 from the atmosphere.

- Sequestering carbon in soil that has received compost application.

- Improving tillage and workability of the soil, thereby reducing emissions of fossil fuel that would be used to work the soil.

- Improving soil structure and reducing erosion, thereby reducing the energy usage required for irrigation.[9]

In light of this extensive technical evidence, the State Board's EIR should include a thorough evaluation of the adverse greenhouse gas impacts that could occur as a result of the adoption of the General Order.

2. Air Quality Impacts from Truck Trips

The diversion of organics from composting facilities to landfills also could cause increased truck emissions.

Composting facilities work best when located close to the source of organics. Fostering local composting operations not only provides local communities with important environmental benefits, it also reduces the need for long-haul truck trips to transport organics to more distant landfills. By contrast, if heightened design and operational costs resulting from the General Order cause a reduction in local compost production, more of these long-haul truck trips would be needed. This would increase the amount of air emissions from trucks, including harmful emissions of diesel particulate matter.

The EIR should thoroughly examine the adverse air quality impacts that could occur as a result of the adoption of the General Order.

3. Water Quality & Water Supply

Although the General Order is intended to protect water quality, the adoption of overly restrictive composting requirements could have the opposite effect by creating a significant adverse effect on water quality. As CalRecycle explained in its September 2012 comment letter to the State Board:

> Diverting organic materials from landfills to produce compost and mulch results in improved soil structure and reduced erosion, resulting in less-energy intensive irrigation and less use of synthetic nitrogen fertilizers (the number one source of groundwater contamination in California). Additionally, composting provides a cost-effective method to stabilize animal manure, which the Water Board has identified as the number two source of groundwater contamination statewide.

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[9] See Attachment 8 at p. 38; Attachment 9 at pp. 64-108, 119; & Attachment 10 (Lou & Nair, *The Impact of Landfilling and Composting on Greenhouse Gas Emissions* (2008) at p. 3796).

Indeed, a University of California Davis report to the California Legislature, prepared in January 2012 on behalf of the State Water Resources Control Board, finds that "agricultural fertilizers and animal wastes applied to cropland are by far the largest regional sources of nitrate in groundwater." The report further explains that nitrate is one of the State's most widespread groundwater contaminants and poses significant public health concerns.[10]

As the U.S. EPA explains, composting has been shown to reduce or eliminate the need for chemical fertilizers.[11] Because composting improves soil structure and reduces erosion, compost enriched soil better retains any fertilizers that are used, so that less of these harmful materials run off the land to pollute waterways. The EPA thus explains that composting improves the physical, chemical, and biological properties of the soil, which reduces erosion, water loss and nonpoint source pollution.[12] For the same reasons, composting also reduces the amount of irrigation water needed for agricultural production, thereby helping to conserve the State's precious water supply.[13]

Given the scientific evidence showing the water quality benefits of composting, the EIR should include a thorough evaluation of the adverse water quality impacts that could occur as a result of the adoption of the General Order.

Additionally, it is important to note that composting operations are already required to comply with California's industrial stormwater requirements. These requirements are currently being revised to incorporate more protective standards than the previous requirements. The analysis in the EIR should take these important water quality protections into account when assessing the comparative environmental impacts and benefits of the different alternatives.

Finally, it is important to note that an increase in direct land application of organic waste in place of composting will result in material being diverted from controlled facilities with water quality protections to direct spreading on farmlands. Any concerns the State Board might have about water quality impacts from the decomposition of this material would be amplified if the material was directly applied to land. The direct land application of organic waste does not face the same regulatory requirements that are proposed in this General Order.

C. The Need to Evaluate Alternatives that Would Reduce Significant Impacts

One of the key purposes of an EIR is to evaluate alternatives to the proposed project that would reduce the project's significant effects on the environment. See CEQA Guidelines § 15126.6. Accordingly, the State Board's EIR should analyze alternatives that would reduce the significant

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[10] See Attachment 11 (University of California, Davis, *Addressing Nitrate in California's Drinking Water, Report for the State Water Resources Control Board to the California Legislature – Executive Summary* (Jan. 2012) at pp. 2-3, available at http://groundwaternitrate.ucdavis.edu/).

[11] See Attachment 3.

[12] See Attachment 12 (printout of http://whatcom.wsu.edu/ag/compost/fundamentals/benefits_benefits.htm, (Washington State University composting website); & Attachment 13 (excerpt from U.S. EPA, *Organic Materials Management Strategies* (July 1999) at p. 40, available at http://www.epa.gov/composting/pubs/index.htm).

[13] See Attachment 3; Attachment 6; Attachment 9 at pp.65-67; & Attachment 13 (EPA explains that "compost application results in water conservation benefits").

impacts caused by adoption of the General Order, including the significant effects caused by diverting organics from composting facilities to landfill disposal or direct land application.

Alternatives that should be considered for analysis in the EIR include the following:

- Delaying the implementation of the General Order until comparable requirements are developed for alternative end-of-life management of organic material (including landfilling and land application)
- Varying the standards for compost pads, pond liners, and drainage facilities;
- A permitting approach based on site-specific conditions relevant to water quality, and not merely tiers tied to the level of permitted capacity;
- Alternative approaches for implementing the General Order, including exempting all existing composting facilities or allowing a long-term phase-in period (e.g., 8-10 years) as a compliance timeline; and
- A meaningful evaluation of the environmental impacts and benefits of the no-project alternative.

The alternatives analysis in the EIR should include a comparative lifecycle and risk-based evaluation, and it should be based on the recognition that composting provides significant environmental benefits. A balanced and reasonable regulatory approach is needed so that these significant benefits are not lost or diminished through the diversion of organics to landfill disposal, land application, or other less environmentally beneficial practices.

D. Conclusion

We recognize that the State Board wants to adopt uniform baseline statewide standards for compost facilities with the goal of protecting water quality. However, the State Board should carefully consider the adverse environmental impacts that will flow from over-regulating the composting industry, which provides important and valuable environmental advantages over landfill disposal or land application.

Composting facilities do not generate the material they process, and, in fact, provide the least environmentally harmful management of this material. As a result, an increase in composting is the most effective way to reduce the water quality impacts of the decomposition of organic waste.

A General Order that is not thoroughly considered and carefully crafted could very well inhibit the development of new composting facilities and force existing facilities out of business, thereby increasing impacts from landfills, increasing the use of pesticides and fertilizers, increasing erosion and run-off of pollutants from agricultural operations, increasing agricultural water use, and increasing emissions from long-haul truck trips. These physical changes in the environment could increase greenhouse gas emissions in a number of ways and could also negatively affect the State’s air and water quality and its limited water supplies. CEQA requires that the State Board fully evaluate these impacts, and fully analyze alternatives that would avoid them, before adopting its General Order.

Respectfully submitted,

Rachel

Rachel Oster
Director of External Affairs
Recology

N. Lapis

Nick Lapis
Legislative Coordinator
Californians Against Waste

Attachments

cc: Brenda K. Smyth, CalRecycle
Barbara Schussman & Marc Bruner, Perkins Coie LLP

Compost Emissions Work Group

This page last reviewed December 21, 2011

Background:

Composting is a biochemical process that breaks down organic wastes into marketable products (soil amendments, animal bedding, and alternative daily cover at landfills); volatile organic compounds (VOCs); heat; and water. Composting uses wastes from forest management, landscaping, agricultural processing, crop harvesting, food consumption, and emergency animal mortalities. Composting can be conducted outdoors or in partially or fully enclosed facilities.

Over 25 percent or approximately 10 million tons of organics are sent to landfills each year in California. The anaerobic decomposition of these wastes result in the emission of methane (a greenhouse gas). Composting of organic waste material has become an important method of managing California's solid waste stream. Composting diverts biomass residue from landfills. This reduces the need for landfill capacity and the production of GHG emissions.

Currently, there are more than 400 compostable material handling facilities permitted by the California Integrated Waste Management Board (CIWMB). Among these include confined animal facilities, landfills, and waste water treatment facilities.

Composting is a source of GHGs, VOCs / reactive organic gases (ROG), particulate matter, and ammonia (NH_3). The California Air Resources Board's (ARB) 2008 emissions inventory estimates indicate that ROG emissions from composting were approximately 38.02 tons per day. These emissions accounted for less than 1 percent of the total ROG emissions in California.

Presently, Antelope Valley Air Pollution Control District, Mojave Desert Air Quality Management District, San Joaquin Valley Air Pollution Control District, and South Coast Air Quality Management District have proposed or adopted rules focused on the control of VOC, NH_3, and PM emissions from composting facilities.

Assembly Bill 32, the California Global Warming Solutions Act of 2006 (AB 32; Statutes of 2006, chapter 488), creates a comprehensive, multi-year program to reduce GHG emissions to 1990 levels by the year 2020. AB 32 requires that the ARB develop GHG reduction strategies that do not interfere with existing air pollution control measures. The AB 32 Scoping Plan contains the main strategies California will use to reduce the GHGs that cause climate change. The Scoping Plan (Measure No. RW-3) commits ARB staff to work with the CIWMB, the California Department of Food and Agriculture, the Department of Transportation, and others to provide direct incentives for the use of compost in agriculture and landscaping.

Proposed Rule 4566: Organic Material Composting Operations

Final Draft New Rule 4566 (Organic Material Composting Operations) August 18, 2011.
Archived Workshop

For questions, please contact: Mei Fong at (916) 324-2570

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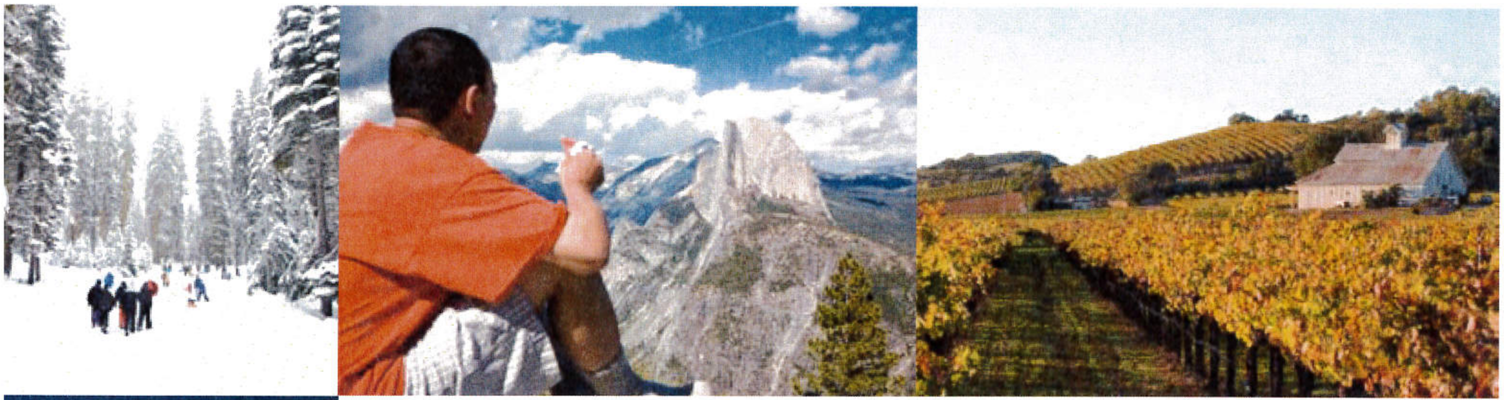

CLIMATE CHANGE SCOPING PLAN

a framework for change

DECEMBER 2008

Pursuant to AB 32
The California Global Warming Solutions Act of 2006

Prepared by
the California Air Resources Board
for the State of California

Arnold Schwarzenegger
Governor

Linda S. Adams
Secretary, California Environmental Protection Agency

Mary D. Nichols
Chairman, Air Resources Board

James N. Goldstene
Executive Officer, Air Resources Board

15. Recycling and Waste

Reduce methane emissions at landfills. Increase waste diversion, composting and other beneficial uses of organic materials, and mandate commercial recycling. Move toward zero-waste.

California has a long track record of reducing greenhouse gas emissions by turning waste into resources, exemplified by the waste diversion rate from landfills of 54 percent (which exceeds the current 50 percent mandate) resulting from recovery of recyclable materials. Re-introducing recyclables with intrinsic energy value back into the manufacturing process reduces greenhouse gas emissions from multiple phases of product production including extraction of raw materials, preprocessing and manufacturing. Additionally, by recovering organic materials from the waste stream, and having a vibrant composting and organic materials industry, there is an opportunity to further reduce greenhouse gas emissions through the indirect benefits associated with the reduced need for water and fertilizer for California's Agricultural sector. Incentives may also be an effective way to secure greenhouse gas emissions reductions in this sector. Table 20 summarizes the emissions reductions from Recycling and Waste sector.

Reduction in Landfill Methane

Methane emissions from landfills, generated when wastes decompose, account for one percent of California's greenhouse gas emissions. Greenhouse gas emissions can be substantially reduced by properly managing all materials to minimize the generation of waste, maximize the diversion from landfills, and manage them to their highest and best use. Capturing landfill methane results in greenhouse gas benefits, as well as reductions in other air pollutants such as volatile organic compounds. ARB is working closely with the California Integrated Waste Management Board (CIWMB) to develop a Discrete Early Action measure for landfill methane control that will be presented to ARB in January.

CIWMB is also pursuing efforts to reduce methane emissions by diverting organics from landfills, and to promote best management practices at smaller uncontrolled landfills. Landfill gas may also provide a viable source of liquefied natural gas (LNG) vehicle fuel. Reductions from these types of projects would be accounted for in the Transportation sector.

High Recycling / Zero Waste

This measure reduces greenhouse gas emissions primarily by reducing the substantial energy use associated with the acquisition of raw materials in the manufacturing stage of a product's life-cycle. As virgin raw materials are replaced with recyclables, a large reduction in energy consumption should be realized. Implementing programs with a systems approach that focus on consumer demand, manufacturing, and movement of products will result in the reduction of greenhouse gas emissions and other co-benefits. Reducing waste and materials at the source of generation, increased use of organic materials to produce compost to benefit soils and to produce biofuels and energy, coupled with increased recycling – especially in the commercial sector – and Extended Producer Responsibility (EPR) plus Environmentally Preferable Purchasing (EPP) also have the potential to reduce emissions, both in-state and within the connected global economy. This measure could also assist in meeting the 33 percent renewables energy goal through deployment of anaerobic digestion for production of fuels/energy.

As noted by ETAAC, recycling in the commercial sector could be substantially increased. This will be implemented through mandatory programs and enhanced partnerships with local governments. The provision of appropriate financial incentives will be critical. ARB will work with CIWMB to develop and implement these types of programs. ARB will also work with CIWMB, the California Department of Food and Agriculture, the Department of Transportation, and others to provide direct incentives for the use of compost in agriculture and landscaping. Further, CIWMB will explore the use of incentives for all Recycling and Waste Management measures, including for commercial recycling and for local jurisdictions to encourage the collection of residentially and commercially-generated food scraps for composting and in-vessel anaerobic digestion.

Table 20: Recycling and Waste Sector Recommendation - Landfill Methane Capture and High Recycling/Zero Waste

(MMTCO$_2$E in 2020)

Measure No.	Measure Description	Reductions
RW-1	Landfill Methane Control (Discrete Early Action)	1
RW-2	Additional Reductions in Landfill Methane • Increase the Efficiency of Landfill Methane Capture	TBD
RW-3	High Recycling/Zero Waste • Mandatory Commercial Recycling • Increase Production and Markets for Organics Products • Anaerobic Digestion • Extended Producer Responsibility • Environmentally Preferable Purchasing	 5 2 2 TBD TBD
	Total	**10**[44]

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[44] Reductions from RW-2 and RW-3 are not counted toward the AB 32 goal. ARB is continuing to work with CIWMB to quantify these emissions and determine what portion of the reductions can be credited to meeting the AB 32 2020 goal. These measures may provide greater emissions reductions than estimated.

Wastes - Resource Conservation

You are here: EPA Home Wastes Resource ConservationComposting Basics

Composting for Facilities Basics

Compost is organic material that can be used as a soil amendment or as a medium to grow plants. Mature compost is a stable material with a content called humus that is dark brown or black and has a soil-like, earthy smell. It is created by: combining organic wastes (e.g., yard trimmings, food wastes, manures) in proper ratios into piles, rows, or vessels; adding bulking agents (e.g., wood chips) as necessary to accelerate the breakdown of organic materials; and allowing the finished material to fully stabilize and mature through a curing process.

Related Links

Greenscapes
CalRecycle Organic Materials Management EXIT Disclaimer
Composting101.com EXIT Disclaimer
US Composting Council EXIT Disclaimer

Natural composting, or biological decomposition, began with the first plants on earth and has been going on ever since. As vegetation falls to the ground, it slowly decays, providing minerals and nutrients needed for plants, animals, and microorganisms. Mature compost, however, includes the production of high temperatures to destroy pathogens and weed seeds that natural decomposition does not destroy.

Benefits of Composting

- Reduce or eliminate the need for chemical fertilizers.
- Promote higher yields of agricultural crops.
- Facilitate reforestation, wetlands restoration, and habitat revitalization efforts by amending contaminated, compacted, and marginal soils.
- Cost-effectively remediate soils contaminated by hazardous waste.
- Remove solids, oil, grease, and heavy metals from stormwater runoff.
- Avoids Methane and leachate formulation in landfills.
- Capture and destroy 99.6 percent of industrial volatile organic chemicals (VOCs) in contaminated air. See Innovative Uses of Compost: Bioremediation and Pollution Prevention.
- Provide cost savings of at least 50 percent over conventional soil, water, and air pollution remediation technologies, where applicable. See Analysis of Composting as an Environmental Remediation Technology
- Reduces the need for water, fertilizers, and pesticides.
- Serves as a marketable commodity and is a low-cost alternative to standard landfill cover and artificial soil amendments.
- Extends municipal landfill life by diverting organic materials from landfills.

The Composting Process

One of the most important steps for evaluating composting options is to become familiar with how the composting process works. Before you begin composting or start a composting program, you should understand the five primary variables that must be "controlled" during composting. These include the following:

Feedstock and nutrient balance. Controlled decomposition requires a proper balance of "green" organic materials (e.g., grass clippings, food scraps, manure), which contain large amounts of nitrogen, and "brown" organic materials (e.g., dry leaves, wood chips, branches), which contain large amounts of carbon but little nitrogen. Obtaining the right nutrient mix requires experimentation and patience and is part of the art and science of composting.

http://www.epa.gov/composting/basic.htm
Last updated on Wednesday, September 04, 2013

Particle size. Grinding, chipping, and shredding materials increases the surface area on which the microorganism can feed. Smaller particles also produce a more homogeneous compost mixture and improve pile insulation to help maintain optimum temperatures (see below). If the particles are too small, however, they might prevent air from flowing freely through the pile.

Moisture content. Microorganisms living in a compost pile need an adequate amount of moisture to survive. Water is the key element that helps transports substances within the compost pile and makes the nutrients in organic material accessible to the microbes. Organic material contains some moisture in varying amounts, but moisture also might come in the form of rainfall or intentional watering.

Oxygen flow. Turning the pile, placing the pile on a series of pipes, or including bulking agents such as wood chips and shredded newspaper all help aerate the pile. Aerating the pile allows decomposition to occur at a faster rate than anaerobic conditions. Care must be taken, however, not to provide too much oxygen, which can dry out the pile and impede the composting process.

Temperature. Microorganisms require a certain temperature range for optimal activity. Certain temperatures promote rapid composting and destroy pathogens and weed seeds. Microbial activity can raise the temperature of the pile's core to at least 140° F. If the temperature does not increase, anaerobic conditions (i.e., rotting) occur. Controlling the previous four factors can bring about the proper temperature.

Types of Composting

Backyard or Onsite Composting

Vermicomposting

Aerated (Turned) Windrow Composting

Aerated Static Pile Composting

In-Vessel Composting

Wastes - Resource Conservation - Reduce, Reuse, Recycle - Composting

You are here: EPA Home Wastes Resource Conservation Composting Environmental Benefits

Environmental Benefits


Man Holding Handful of Compost - Photo Courtsey of Jepson Prairie Organics

Compost use can result in a variety of environmental benefits. The following are a few of the most important benefits:

Compost enriches soils

Compost has the ability to help regenerate poor soils. The composting process encourages the production of beneficial micro-organisms (mainly bacteria and fungi) which in turn break down organic matter to create humus. Humus—a rich nutrient-filled material—increases the nutrient content in soils and helps soils retain moisture. Compost has also been shown to suppress plant diseases and pests, reduce or eliminate the need for chemical fertilizers, and promote higher yields of agricultural crops.

Compost helps cleanup (remediate) contaminated soil

The composting process has been shown to absorb odors and treat semivolatile and volatile organic compounds (VOCs), including heating fuels, polyaromatic hydrocarbons (PAHs), and explosives. It has also been shown to bind heavy metals and prevent them from migrating to water resources or being absorbed by plants. The compost process degrades and, in some cases, completely eliminates wood preservatives, pesticides, and both chlorinated and nonchlorinated hydrocarbons in contaminated soils.

Compost helps prevent pollution


Compost Used as Erosion Deterent

Composting organic materials that have been diverted from landfills ultimately avoids the production of methane and leachate formulation in the landfills. Compost has the ability to prevent pollutants in stormwater runoff from reaching surface water resources. Compost has also been shown to prevent erosion and silting on embankments parallel to creeks, lakes, and rivers, and prevents erosion and turf loss on roadsides, hillsides, playing fields, and golf courses.

Using compost offers economic benefits

Using compost can reduce the need for water, fertilizers, and pesticides. It serves as a marketable commodity and is a low-cost alternative to standard landfill cover and artificial soil amendments. Composting also extends municipal landfill life by diverting organic materials from landfills and provides a less costly alternative to conventional methods of remediating (cleaning) contaminated soil.

http://www.epa.gov/composting/benefits.htm
Last updated on Monday, May 13, 2013

United States Environmental Protection Agency

GreenScapes
U.S. EPA Sustainable Landscapes

Backyard Composting

IT'S ONLY NATURAL

Composting is nature's way of recycling organic materials back into the soil in order for the cycle of life to continue. The billions of living organisms in healthy soil transform dead plants into vital nutrients for new plant growth. Since healthy plants come from healthy soil, one of the best ways you can build healthy soil in your garden and lawn is by using compost. You can easily make compost with landscape trimmings and food scraps in your own backyard. With a small investment in time, you can improve the health and appearance of your yard, save money on fertilizers and mulch, all while preserving natural resources and protecting the health of your family and pets.

Why compost?

- **It's earth-friendly**: Food scraps and yard waste make up 20-30% of the waste stream. Making compost keeps these materials out of landfills, where they take up precious space and release methane, a greenhouse gas 21 times more potent than carbon dioxide emissions in the atmosphere.
- **It benefits your yard**: Compost improves soil structure and texture, increases the soil's ability to hold both water and air, improves soil fertility, and stimulates healthy root development in plants.
- **It's easy**: You can start with just leaves and grass, then work your way towards composting your food scraps.
- **It saves money**: Adding compost to your garden can reduce or eliminate the need to buy chemical fertilizers or compost. If you pay for the amount of trash hauled, composting can also cut down on your trash costs.

What do I need to do to make compost?

A **Bin or Pile?** Some people start with an easy pile, and then move to a bin when they're ready. You can give your pile some structure with chicken wire, snow fencing, or by nailing scrap wood together to make a four-sided box. A pile works great for just leaves and grass clippings, but when you want to incorporate food waste, it's time to use a bin to prevent rodents. Closed-top bins include turning units, stacking bins, and bins with flip tops. Many communities provide their residents free or discounted bins to encourage backyard composting. Bins can also be purchased from retail or mail order businesses. Take the time to consider your options and then select a bin or pile to fit your needs.

B **Space.** Select a dry, shady, or partly shady spot near a water source and preferably out of sight for your compost pile or bin. Ideally, the compost area should be at least three feet wide by three feet deep by three feet tall (one cubic yard). This size provides enough food and insulation to keep the organisms in the compost warm and happy and working hard. However, piles can be larger or smaller and work just fine if managed well.

C **Browns for carbon, greens for nitrogen, air for organisms, and water for moisture.**

Brown material provides carbon and includes:

- Paper, like shredded pieces of paper, cardboard, and paper rolls,
- Dry yard waste, like dry leaves, small branches, and twigs, straw, sawdust, and used potting soil.

Green material provides nitrogen and includes:

- Wet yard waste like fresh grass clippings, green leaves, and soft garden prunings
- Food scraps like vegetable and fruit peels, coffee grounds, and tea bags.

Vermicomposting is a method of composting using a special kind of earthworm known as a red wiggler (Elsenia fetida), which eats its weight in organic matter each day. Vermicomposting is typically done in a covered container with a bedding of dirt, newspaper, or leaves. Fruit and vegetable scraps can then be added as food for the worms. Over time, the food will be replaced with worm droppings, a rich brown matter that is an excellent natural plant food. Vermicomposting requires less space than normal composting methods, and is therefore ideal for classrooms, apartments, and high-density urban areas.

How do I make compost?

1 Add your brown and green materials (generally three parts browns to one part greens), making sure larger pieces are chopped or shredded. The ideal compost pile contains browns and greens (of varying sizes) placed in alternate layers of different-size particles.

2 Mix grass clippings and green waste into the pile and bury fruit and vegetable waste under 10 inches of compost material.

3 As materials breakdown, the pile will get warm and on cold days you may even see some steam.

4 Every time you add to the pile, turnover and fluff it with a pitchfork to provide aeration, unless your bin has a turner.

5 When material at the bottom is dark and rich in color, with no remnants of your food or yard waste, your compost is ready to use. There may be a few chunks of woody material left; these can be screened out and put back into a new pile. The resulting compost can be applied to lawns and gardens to help condition the soil and replenish nutrients. Compost should not be used as potting soil for houseplants because it may still contain vegetable and grass seeds.

Troubleshooting Your Pile

Problem:	Cause	Solution:
Rotten egg smell	*Insufficient air or too much moisture*	Turn pile and incorporate coarse browns (sawdust, leaves)
Ammonia smell	*Too much nitrogen*	Incorporate coarse browns (sawdust, leaves)
Pile does not heat up or decomposes slowly	*Pile too small*	Add more organic matter
	Insufficient moisture	Turn pile and add water
	Lack of nitrogen	Incorporate food waste, grass clippings, or manure (chicken, rabbit, cow, horse)
	Not enough air	Turn pile
	Cold weather	Increase pile size or insulate with straw or a tarp

How do I get started?

What to add

Greens:

- Uncooked or cooked fruits and vegetables
- Bread and grains
- Coffee grounds and filters
- Grass clippings
- Paper tea bags with the staple removed, if there is one.
- Hair and fur
- Chicken, rabbit, cow, horse manure

Browns:

- Cotton or wool rags
- Dryer and vacuum cleaner lint
- Eggshells
- Nut shells
- Fireplace ashes (from wood burning)
- Sawdust
- Hay and straw
- Yard trimmings (e.g., leaves, branches, twigs)
- Houseplants
- Used potting soil
- Wood chips
- Leaves
- Shredded newspaper
- Cardboard rolls
- Clean paper

What not to add

- Aluminum, tin or other metal
- Glass
- Dairy products (e.g., butter, milk, sour cream, yogurt) & eggs
- Fats, grease, lard, or oils
- Greasy or oily foods
- Meat or seafood scraps
- Pet wastes (e.g., dog or cat feces, soiled cat litter)
- Soiled diapers
- Plastic
- Stickers from fruits or vegetables (to prevent litter)
- Black walnut tree leaves or twigs
- Yard trimmings treated with chemical pesticides
- Roots of perennial weeds
- Coal or charcoal ash
- Firestarter logs
- Treated or painted wood

Success Story

Turning Garbage into Gold

> **"Composting can work in the marketplace and provide ongoing environmental and economic benefits."**
>
> John Majercak
> Director of Waste Management Programs
> Center for Ecological Technology

Organic waste comprises a significant portion of the U.S. municipal solid waste (MSW) stream. EPA estimates that the nation's MSW contained 85.7 million tons of paper and paperboard in 1999, 25.2 million tons of food discards, 27.7 million tons of yard trimmings, and 12.3 million tons of wood—adding up to 66 percent of the total waste stream. Similarly, compostable waste in Massachusetts accounts for as much as 70 percent of the state's total MSW by weight.

Composting Organic Waste in Massachusetts

The Center for Ecological Technology (CET), a nonprofit organization that promotes sustainable technologies in New England, successfully created an innovative market-based infrastructure for diverting commercial and agricultural organic waste from disposal in landfills. CET's on-farm composting program turns the waste into organic material suitable for sale as a market product or for use on the farm.

A Market-Based Approach

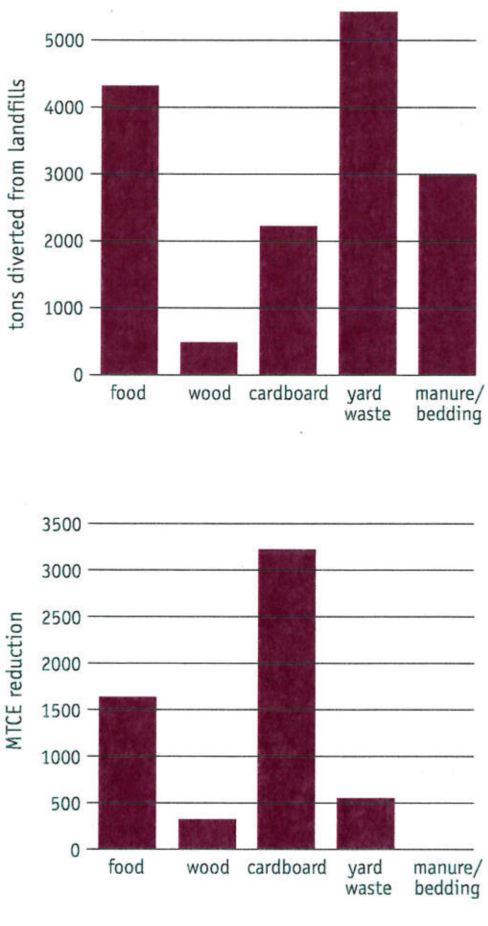
Massachusetts has innovative state policies on composting, an active agricultural sector, and existing composting activities. CET, aware of these advantages, seized the opportunity to turn composting into a regular "way of doing business." With funding from the U.S. Department of Agriculture, the Massachusetts Department of Environmental Protection, and foundations, CET launched an extensive outreach and technical assistance program in 1996.

CET's on-farm composting project targeted farmers, waste haulers, and commercial waste generators. Within three years, CET had enlisted 7 farmers, 6 commercial haulers, and 45 commercial waste generators at more than 70 locations such as supermarkets, restaurants, schools, and large wholesale food distributors. The haulers transport the waste to the farms where it is composted, and the finished product is then marketed to customers such as landscapers and home gardeners. The program is continuing and expanding through industry-led efforts.

Benefits

By reusing the organic wastes rather than disposing of them, CET achieved greenhouse gas reductions of approximately 5,700 metric tons of carbon equivalent (MTCE) from the program's inception in 1996 to 2000—an amount comparable to the amount of carbon that would be sequestered annually by 6,333 acres of five-year-old trees.

Results at a Glance: 1996-2000



In addition to reductions in greenhouse gases, diverting organic waste from landfills can help reduce leachate production and free up limited landfill space.

The project has economic benefits as well. Communities and commercial waste generators benefit from the lower tipping costs charged by the farmers versus landfills. Farmers receive income by accepting the commercial waste and marketing the finished compost to the public. Haulers, seeking local disposal options, and communities, seeking to prolong landfill life, benefit from the availability of farm composting facilities.

The program also helps promote sustainable agriculture, as farmers can better manage their own wastes and substitute the compost for petroleum-based fertilizers.

Challenges

CET overcame a number of obstacles. At the outset, farmers needed assurance that enough waste would be available to make the program cost-effective, and waste generators wanted to be sure that farms would accept the waste. CET met these barriers through extensive outreach and technical assistance to ensure a critical mass of participants to reduce the risks and achieve needed economies of scale.

Quality control to ensure an end product suitable for sale presented another obstacle. CET employed technical experts to train participants in techniques to ensure that the waste loads are free from contamination and are composted correctly.

Farmers accept organic waste such as food waste, corrugated cardboard, paper, and yard trimmings and turn it into a marketable finished product.

CET recognized the need to make on-farm composting cost-effective. High tipping fees at New England landfills ($65 to $85/ton) enable farmers to charge tipping fees for food scraps and other compostable materials ($25 to $35/ton) that are sufficient to help make their operations profitable.

They supplement the tipping fees through retail sales of $30 per yard of compost (wholesale is only $8/yard). The lower the tipping fee, the higher the price the composter needs to secure for the finished product. When replicating the program in regions where tipping fees are lower, the retail market is even more critical.

Replicating the Project

Communities interested in replicating the CET model should begin by making a shift from thinking of composting as a disposal option to viewing it as manufacturing a product and the composter as capturing the inherent value left in a material. Line up waste generators, haulers, and composters so that everyone is ready to initiate the program at the same time. Ensure that the project has enough sites to reduce the risk that the effort would fall apart if one site shuts down. Be ready to provide extensive follow-up technical assistance.

In line with CET's goal of modifying practices that have adverse impacts on the natural environment, the organization's On-Farm Composting project proves that innovative waste management practices can reduce emissions of destructive greenhouse gases.

Additional Information

"Building a Market-Based System of Farm Composting and Commercial Food Waste in Western Massachusetts — Final Report," available on the CET Web site at www.cetonline.org.

John Majercak, CET director of waste management programs, email: johnm@cetonline.org.

EPA's Climate and Waste Program increases awareness of climate change and its link to waste management in order to (1) make greenhouse gas emissions a factor in waste management decisions and (2) employ waste management as a mitigation action for reducing greenhouse gas emissions. For additional information on EPA's Climate and Waste Program, see www.epa.gov/mswclimate.

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Composting: A Greenhouse Gas Mitigation Measure

Ton for ton, composting reduces GHG emissions from organics management over any other management option.

Compostable organics make up 30% of California's overall waste stream, contributing over 12 million tons annually to our state's landfills. In landfills, this material undergoes anaerobic decomposition and produces significant quantities of methane, up to 80% of which is not captured by a landfill gas system. Composting, on the other hand, is a fundamentally aerobic process, and **well managed compost facilities do not produce any methane**. Composting offers an environmentally superior alternative to landfilliing organics that eliminates methane production, provides a series of economic and environmental co-benefits, and has a substantial impact on greenhouse gas reduction.

- **Composting organic material reduces GHG emissions compared to landfilling with energy recovery systems**. According to the most conservative estimates, which fail to account for many (if not most) GHG-reducing benefits of composting, California could reduce its GHG emissions by one million MTCO2E by composting just 30% of the foodwaste that is currently disposed. This is equivalent to the carbon sequestered by 26 million tree seedlings grown for 10 years.
- **Compost can significantly reduce agricultural energy demand**. Plants grown in compost-rich soil require less irrigation because of the increased infiltration and storage capacity of root systems and the reduction of water runoff, evaporation, and water usage by weeds. Research has shown that the application of compost can reduce the need for irrigation by 30-70%. Given that approximately 8% of the electricity generated in the state is used to run California's massive water supply infrastructure, a substantial decrease in water consumption would significantly reduce energy consumption.
- Composting provides nutrient-rich soils, which multiple studies have shown results **in greater carbon storage in crop biomass.**
- **The application of compost results in a reduced need for GHG producing petroleum-based chemical fertilizer, pesticides, herbicides, and additives**. These chemicals are carbon-intensive in their production and emit large quantities of global warming pollutants during application and as they decompose in the soil. The use of compost can reduce the need for fertilizers for vegetable crops by 33-66%.
- **The application of compost greatly increases the amount of carbon sequestered in soil.** Experimental studies have shown that increased carbon sequestration in soil from composting application was 6 to 40 tons of carbon per hectare.

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organics

Organics recycling offers major environmental benefits

FACT SHEET

Over 1.4 million[1] tonnes of organic materials are diverted from landfill and processed into a range of valuable materials in NSW every year. Most of this material is sold into a range of domestic, horticultural and agricultural markets for use as mulch and soil conditioners.

Councils play an important role in the source-separated kerbside collection of organic material. This material is processed by the commercial compostingindustry, playing an important role in processing the recovered organic material into compost. The industry contributes to the development of end-markets for compost, without which organic materials would be disposed of as waste. As a result, significant savings in limited landfill space are achieved, and reduces the impacts that landfills have on the environment.

Life cycle study

A full life cycle study was conducted in NSW in 2003.[2] The study — an international first — assessed the total environmental impacts of windrow composting systems throughout their life cycle and quantified the benefit to the environment of applying compost to agricultural soils.

The life cycle assessment covered the environmental impact of:

- manufacturing of composted products;
- transport of composted products to end-markets and users; and
- application of composted products in agriculture.

A wide range of environmental indicators was used, including energy consumption, greenhouse effect, photochemical oxidant formation, human and eco-toxicity, resource depletion, eutrophication, and land use.

The study represents a significant advance on previous life cycle studies that have generally considered only one impact category, such as greehouse gas emissions.

As a consequence, previous studies have under-estimated the full environmental benefits of composting and organics recycling.

Councils play an important role in collecting and recycling organic materials in NSW

Department of Environment & Climate Change NSW

Key findings

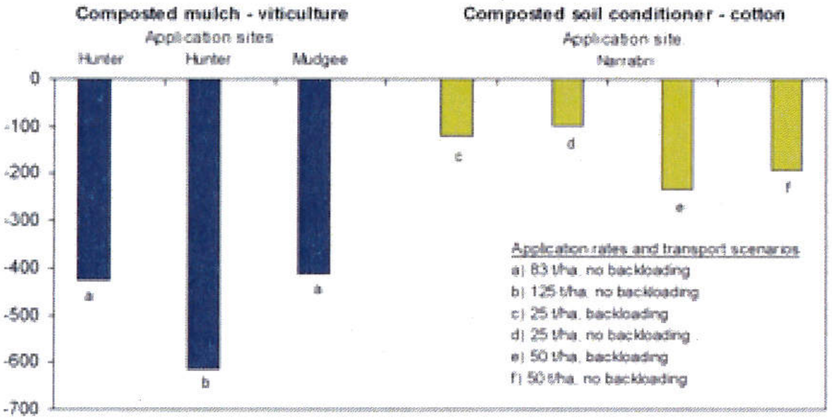
The results of the study show that the management of organics through source-separated collection systems, commercial composting and the application of compost to agriculture, offers significant benefits to the environment, including substantial reduction in greenhouse gas emissions, for example:

- Commercial composting of organic waste and application of compost materials to agricultural soils results in net greenhouse gas reduction, even if the recycled materials have to be transported up to 600km for agricultural applications (Figure 1).
- Positive benefits were found under all the other environmental indicators, including reduced potential for human toxicity, eco toxicity and eutrophication.
- Small negative environmental impacts related to photochemical oxidation potential and abiotic resource depletion. This arose largely from the production and use of diesel fuel consumed while applying the compost material to agricultural soils.

Conclusions

- The findings of the study support the development and implementation of strategies for source-separated collection and recycling of organic materials.
- Service providers, such as Councils, can justify the introduction of source-separated collection services for organics.
- Communities can achieve major environmental benefits by separating and recycling organic materials.

Figure 1. Greenhouse benefits of producing and using compost (negative values = reduction in greenhouse gases or global warming potential).



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our environment it's a living thing

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NSW GOVERNMENT

Environment, Climate Change & Water

Short report: The benefits of using compost for mitigating climate change

The Organic Force

Acknowledgement:

The benefits of using compost for mitigating climate change report is a joint partnership project between the author Johannes Biala and Department of Environment, Climate Change and Water NSW .

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Please note:

The Organic Force, in its provision of advice, acts in good faith and takes all reasonable steps to ensure that the advice offered is correct and applicable to the individual circumstances being advised. However in dealing with the Department of Environment, Climate Change and Water NSW or any other State or Federal jurisdiction, it is suggested that the relevant department is contacted by the reader to clarify any points of law or procedure relating to the individual situation and written answers are obtained to all queries.

This short report is generally not referenced, except for graphs and tables. Please refer to the full report for acknowledgements of the data used, a full list of proper references and a comprehensive list of abbreviations. Most data related to the use of organic soil amendments such as manure, biosolids and compost is sourced from Europe and North America, and therefore does not necessarily reflect soil and environmental conditions that might be encountered in Australia. In most cases the terminology used by authors of reviewed papers and reports has been adopted, thus terms such as 'sewage sludge' and 'biosolids', or 'soil carbon' and 'soil organic matter' are used in parallel.

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Table of contents

Executive summary

This report summarises an extensive and detailed literature review entitled *The benefits of using compost for mitigating climate change*, which was commissioned by Department of Environment, Climate Change and Water NSW. The report findings support the NSW and Federal government priorities to reduce waste and tackle climate change.

In NSW, the State Plan, Waste Avoidance and Resource Recovery Strategy (WARR) and the NSW Climate Action Plan set waste and greenhouse gas (GHG) reduction targets and identify priority actions and strategies that guide the work in the key areas of waste reduction and climate change adaptation. Returning recycled organics to the land in the form of compost can deliver many benefits and assist in achieving these targets.

A key objective of this report is to summarise the scientific literature reporting on research on the use of compost and related products in mitigating climate change. Compost and related products are processed from recycled organic materials such as garden organics, food organics, crop residues, biosolids and manures. Diverting these materials from landfill reduces methane emissions. Applying the products leads to climate change benefits through carbon sequestration in soil, substitution of nitrogenous and other synthetic fertilisers and the flow-on effects of improved soil health and water holding capacity following their application.

Across Australia an estimated 3.7 million tonnes (Mt) of garden and food organics and a percentage of wood residue were diverted and recycled from landfill in 2007–08, preventing methane generation equivalent to ~ 4.28 Mt CO_2-e. Methane is a potent greenhouse gas (21 to 25 times as potent as carbon dioxide) that is normally produced when organic materials break down in landfill. The drop in greenhouse gas emissions from the waste sector has largely been attributed to the rise in landfill gas capture measures (~4.5 Mt CO_2-e in 2007–8); however, what goes unreported is the fact that emissions figures would be almost twice as large if the organics which are currently recycled were to be landfilled instead.

If only 50% of the 9.68 Mt (2007–8 figures) of organic residues sent to landfill was recycled, methane generation of more than 5 Mt CO_2-e per annum could have been prevented. This would have brought methane generation savings through organics recycling activities to around 10 Mt CO_2-e in Australia.

There are also significant opportunities for mitigating greenhouse gas (GHG) emissions through improved agricultural management, particularly of croplands. Greenhouse gas emissions from agriculture (excluding emissions caused by 'Land use, Land-use Change and Forestry' – land clearing, soil carbon in grazing and cropland and forest management) and waste management contributed 15.2% and 2.5%, respectively, to Australia's total GHG emissions in 2008. Of the estimated global technical agricultural mitigation potential by 2030, about 89% is from soil carbon sequestration and about 2% from mitigation of soil nitrous oxide (N_2O) emissions (Smith *et al.* 2007a[*]). The soil carbon content can be raised by increasing the carbon input, decreasing the output or a combination of both.

---

[*] quoted in IPCC: Smith, P., D. Martino, Z. Cai, D. Gwary, H. Janzen, P. Kumar, B. McCarl, S. Ogle, F. O'Mara, C. Rice, B. Scholes and O. Sirotenko (2007). Agriculture, Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Climate Change 2007: Mitigation. O. R. D. B. Metz, P.R. Bosch, R. Dave, L.A. Meyer (eds). Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, IPCC: 497-540.

Increases in soil carbon storage can be achieved by changes in the following areas of agricultural production, expressed as average annual global carbon sequestration rates over a 20-year period:

agronomy	0.29 to 0.88 t CO_2-e ha^{-1} yr^{-1}
nutrient management	0.26 to 0.55 t CO_2-e ha^{-1} yr^{-1}
tillage/residue management	0.15 to 0.70 t CO_2-e ha^{-1} yr^{-1}
water management	1.14 t CO_2-e ha^{-1} yr^{-1}
manure/biosolids use	1.54 to 2.79 t CO_2-e ha^{-1} yr^{-1}
compost	1.10 to 1.80 t CO_2-e ha^{-1} yr^{-1}

The addition of external carbon sources (such as **compost**) has the **highest soil carbon sequestration potential**. There are factors which limit soil carbon sequestration including nutrient supply (e.g. lack of nitrogen, phosphorus), sink saturation (optimum carbon levels reached) and reversibility (carbon sequestration is not necessarily permanent).

Although relatively small in absolute terms, nitrous oxide emissions cannot be ignored due to their high global warming potential (nitrous oxide is 310 times more potent as a GHG than CO_2), the permanency of any reductions and the potential of nitrous oxide emissions annulling carbon sequestration gains. In agriculture, nitrous oxide gas is primarily produced when there are high levels of moisture in the soils leading to anaerobic pockets and when nitrogen in excess of crop needs is applied. The primary consideration for mitigating nitrous oxide emissions from the agricultural sector is to match the supply of mineral nitrogen commensurate with the needs of crops. Nitrous oxide emissions in Australian agricultural systems were below the IPCC default (1.25% of added nitrogen) in cotton and wheat, but well above that level in high rainfall south east Queensland, and extreme in sugar cane on acid sulfate soil.

The increased input of carbon from organic soil amendments (animal manure, compost, crop residues, sewage sludge) is one of the most efficient measures for soil carbon sequestration. **Organic farming systems** prove this through reliance on high organic matter inputs, and on carbon and nutrient cycling to maintain soil quality and productivity. Virtually all comparative trials showed superior carbon sequestration for organic farming systems compared to conventional systems, although maintaining carbon levels on sandy soil can be difficult even for organic farms. Organic farming usually manages to reduce GHG emissions per unit land area, but not necessarily per unit of product.

It is estimated that approximately 3.2 Mt of **animal manures** are generated annually in Australia. Studies over a wide variety of soil textures and climates in diverse agricultural cropping systems have reported increases in soil organic matter following the addition of manure. The rate of increase in soil organic matter depends on temperature, moisture and tillage conditions, as well as the amount of manure added. For all climatic regions except for cold climates, it can be expected that between 5% and 20% of carbon applied with manure is retained and incorporated into the soil carbon pool. **Composted manure** retains considerably higher proportions of applied carbon in soil than does raw manure.

During the digestion of **sewage sludge**, much of the easily decomposable carbon is lost. The organic carbon added to soil in digested sludge is therefore more resistant to decomposition than the carbon in raw sludge. There are dramatic differences between sludges in their ability to release nitrous oxide, but there is no clear explanation of mechanisms controlling these differences.

In broad terms, there are two basic **compost products**:

(i) composted mulch (70% of mass >15mm, applied to soil surface)
(ii) composted soil conditioners, suitable for incorporation into the soil.

Due to a lack of data regarding the use of organic mulches, most compost information in this study relates to compost used as a soil conditioner.

During **composting**, about 50% of carbon contained in the raw materials is lost as CO_2, and 50% is retained, mostly in recalcitrant organic compounds. The rate and extent of **mineralisation of compost products after application to soil** depends on the quantity, type, maturity and particle size distribution of the applied product, as well as on soil properties, environmental conditions, and agricultural management practices. The labile organic compounds contained in compost are degraded relatively quickly, and the recalcitrant fractions remain in the soil. The consecutive use of mature garden/food organics compost for 12 years in a study has shown carbon retention rates of 45% to 50%, while the use of pasturised garden organics compost indicated about 30% carbon retention. However, the latter compost showed the highest carbon retention per dry tonne of compost applied.

No model is currently available that fully describes soil carbon dynamics following the use of compost. In the interim, a **simplified compost carbon sequestration model,** 'CENTURY', is available. The model predicts that new equilibria for soil organic carbon will not be reached before 200 and 300 years for annual compost application rates of 10 and 15 t ha^{-1}, respectively for northern European conditions, and that soil carbon sequestration will be possible for much longer than 20 years.

Using the CENTURY model, it was predicted that 117 kg C (428 kg CO_2-e) is sequestered per dry tonne of applied compost over a 100-year time frame for Australian conditions, and 88 to 93 kg C (321–342 kg CO_2-e) per dry tonne of applied mulch.

The IPCC framework for estimating greenhouse gas fluxes is based on a 100-year time horizon and will only consider compost carbon as 'sequestered' if it remains locked in the soil for at least 100 years. It has been suggested that 2% to 10% of carbon introduced with compost will still be in the soil after 100 years. More recent data suggests a range between 9% and 14%, depending on soil type and crop rotation.

The proportion of compost-derived carbon that becomes part of the stable soil carbon can be considered as 'sequestered' in the realm of the IPCC framework and for the purpose of international carbon trading. Accordingly, if Cinput (kg) is the carbon content in compost and Cbind is the fraction that is or will become 'stable', then the carbon sequestration, expressed as CO_2 ($CO_{2,bind}$, kg) can be calculated as: $CO_{2,bind} = Cinput \times Cbind \times 44/12$.

If carbon levels range between 10.0% and 28.5% (dry matter (DM)) and between 19.1 and 47.0% (DM) for garden and food organics compost respectively, and if between 2% and 14% of compost carbon are sequestered, carbon sequestration can be calculated to be in the order of two to 79 kg CO_2-e t^{-1} for food organics and three to 73 kg CO_2-e t^{-1} for garden organics, assuming mass losses during the composting process of 60% and 30% respectively. When compost is used regularly for four or more years, between 5% and 15% or more of **nitrogen** applied with compost will be utilised by crops annually, which means that between about 20% and 35% of compost-applied nitrogen will support plant growth over a three-year crop cycle. Because only a small proportion of the total nitrogen applied with compost is mineralised and used by plants, continuous compost use increases soil nitrogen levels substantially, providing significantly higher soil nitrogen supply potential.

The use of compost can supply at least some of the crop's nitrogen, as well as most, if not all, of the phosphorous, potassium and trace elements required. Substituting the use of mineral fertiliser through compost use offers the opportunity of reducing GHG emissions generated in the manufacturing and transportation of fertilisers. If 10 t ha^{-1} DM of 'typical' garden organics compost is used continuously, resulting in 40% uptake of nitrogen and 100% of phosphorous and potassium, GHG emissions of approximately 180 kg CO_2-e can be avoided.

When considering the **sequestration of compost-derived carbon** over both the long-term (100 years) and the medium term (20 to 50 years) we can say carbon sequestration resulting from compost use is an important interim climate change mitigation measure, because it provides opportunities for implementing low-cost measures that are immediately available and deliver a wide range of other environmental, agronomic and societal benefits.

Studies indicate that we can assume 45% of carbon applied with compost is retained over a 20-year period, 35% over a 50-year period, and 10% over a 100-year period. Therefore the use of mature garden organics compost as agricultural soil conditioner at a rate of **10 t DM ha^{-1}** will sequester carbon that is equivalent to reducing GHG emissions by:

- **5,046 kg CO_2-e over 20 years**
- **3,532 kg CO_2-e over 50 years**
- **1,009 kg CO_2-e over 100 years.**

If GHG emission savings from fertiliser replacement are added, using **10 t DM ha^{-1}** mature garden organics compost as agricultural soil conditioner can result in GHG emissions savings of:

- **5,224 kg CO_2-e within a 20-year time frame**
- **3,710 kg CO_2-e within a 50-year time frame**
- **1,187 kg CO_2-e within a 100-year time frame.**

In summary, the literature found that using compost as an agricultural and horticultural soil amendment:

- can contribute to mitigating climate change directly and indirectly
- provides opportunities for implementing low-cost measures that are immediately available
- is one of the fastest means of improving soil carbon levels
- is ideally suited as a mitigation measure in productive agricultural soils
- fits easily into the Australian National Carbon Accounting System
- can attract carbon credits
- delivers many agronomic benefits and enhances long-term agricultural productivity and production
- offers environmental and societal benefits.

1 Introduction

This report is a summary of a more extensive and detailed report with the same title: *The benefits of using compost for mitigating climate change.* The full report can be obtained from the Department of Environment, Climate Change and Water NSW.

Facilitating the diversion of organics from landfill has been a primary concern of all government jurisdictions. On the one hand, organic material is responsible for methane generation from landfills but the application of products processed from recycled organics to soil results in a range of important environmental benefits. These environmental benefits include improved soil health, water savings, improved crop productivity, reduced need for synthetic fertiliser and biocidal products, reduced water and wind erosion, improved tilth and, as this literature review demonstrates, enhanced capacity to mitigate climate change by 'locking up' or sequestering carbon in soils, reducing nitrous oxide emissions and reducing agricultural energy use.

Despite the growing body of studies demonstrating the capacity of the diversion of organic material and the application of compost products to mitigating climate change, the Australian National Carbon Accounting System does not currently include compost in its calculations. This report has been commissioned to assist in facilitating the recognition of compost's contribution by climate change governmental bodies, research institutes and agricultural enterprises.

In order to understand the current research that has been conducted in this arena, the Department of Environment Climate Change and Water NSW (DECCW) commissioned Johannes Biala of The Organic Force to review relevant scientific studies.

An extensive literature review entitled 'The benefits of using compost for mitigating climate change' has been prepared for policy makers and researchers. The full literature review can be obtained from DECCW. From this comprehensive survey, this short report draws together a summary of the full review.

The short report targets government officials from all jurisdictions, agricultural research and advisory bodies, agricultural enterprises, bodies involved in carbon quantification, verification and trading, compost processors and compost customers.

This Short Report provides condensed background information on key aspects of agricultural climate change mitigation in croplands (section 2) and the effects on climate change of using organic soil amendments (section 3) as well as conclusions of the literature review and recommendations for further work (section 4).

Note:
In Australia, the application to land of waste derived materials including biosolids and paper mill sludge, is subject to consent of the relevant state or federal authority.

For example, in New South Wales, waste is regulated under the Protection of the Environment Operations (Waste) Regulation 2005. The Government encourages the recovery of resources from waste where this is beneficial and does not harm the environment or human health. A provision for resource recovery exemptions within the legislation enables the reuse of waste or waste derived materials as fill or fertiliser (land applications) that may otherwise go to landfill.

2 Key aspects of agricultural climate change mitigation in croplands

2.1 Greenhouse gas emissions from the waste and agricultural sectors

Greenhouse gas (GHG) emissions from agriculture (excluding emissions caused by 'land use, landuse change and forestry' (LULUCF)) and waste management contributed 15.2% and 2.5% respectively, to Australia's total GHG emissions in 2008. Across Australia an estimated 3.7 million tonnes (Mt) of garden and food organics and a percentage of wood residue were diverted and recycled from landfill in 2007–08, preventing methane generation equivalent to ~ 4.28 Mt CO2-e. Methane is a potent greenhouse gas (21 to 25 times as potent as carbon dioxide) that is normally produced when organic materials break down in landfill. The drop in greenhouse gas emissions from the waste sector has largely been attributed to the rise in landfill gas capture measures (~4.5 Mt CO_2-e in 2007–8); however, what goes unreported is the fact that emissions figures would be almost twice as large if the organics which are currently recycled were to be landfilled instead.

If only 50% of the 9.68 Mt (2007–8 figures) of organic residues sent to landfill was recycled, methane generation of more than 5 Mt CO_2-e per annum could have been prevented. This would have brought methane generation savings through organics recycling activities to around 10 Mt CO_2-e in Australia.

There are also significant opportunities for mitigating greenhouse gas (GHG) emissions through improved agricultural management, particularly of croplands. Greenhouse gas emissions from agriculture (excluding emissions caused by 'Land use, Land-use Change and Forestry') and waste management contributed 15.2% and 2.5%, respectively, to Australia's total GHG emissions in 2008. Of the estimated global technical agricultural mitigation potential by 2030, about 89% is from soil carbon sequestration and about 2% from mitigation of soil nitrous oxide (N_2O) emissions.

According to the federal government's *National Inventory Report*, agricultural soils accounted for 17% and manure management for 3.9% of total agricultural emissions in 2007 (88.1 Mt CO_2-e). Agricultural nitrous oxide (N_20) emissions accounted for 85.9% of the national nitrous oxide emissions. A closer look at the *National Inventory Report* reveals that composting is not considered as one of the manure management options, although it is likely to reduce methane and nitrous oxide emissions associated with alternative manure management systems. A review should be undertaken to determine whether and to what extent composting of manure can reduce methane and nitrous oxide emissions. Composting should then be included and featured as a separate manure management system in future national greenhouse inventory reports.

It is important to understand that agricultural sector GHG emissions take into account only nitrous oxide and methane emissions, while changes in plant and soil carbon pools are dealt with in the 'Land use, Land-use Change and Forestry' sector. All GHG emissions directly allocated to agricultural soil in the Commonwealth's *National Inventory Report* are solely attributed to nitrous oxide emissions.

Nitrous oxide is generated during biochemical nitrogen transformations in soil (mainly nitrification/denitrification). These transformations involve inorganic nitrogen compounds, which can become available from various sources, the major source being the application of inorganic fertilisers. While the Intergovernmental Panel on Climate Change proposes a default nitrous oxide emission factor for nitrogenous mineral fertilisers of 1.25% of applied nitrogen, Australia has developed differentiated factors for different farming systems. These factors are:

- 0.3% for non-irrigated crops
- 0.4% for irrigated and non-irrigated pasture
- 0.5% for cotton
- 1.25% for sugar cane, and
- 2.1% for irrigated and horticultural crops.

An emission factor of 1% of applied nitrogen was adopted in Australia for manure spread on pastures and crops. This was based on the average of the generic class 'organic', sewage sludge and wastes from animal waste management systems.

As mentioned previously, within the Australian National Greenhouse Inventory, soil carbon emissions and stocks are taken into consideration within the Land use, Land-use Change and Forestry sector. Within this sector, croplands comprise emissions and sinks from *Croplands remaining Cropland* (20 million ha) and *Forestland converted to Cropland*, amounting to total net emissions of 23.6 Mt CO_2-e in 2007. Emissions from grasslands, which were estimated to have been more than ten times higher than from croplands, were not considered, as grasslands are relatively unimportant for compost use. Reporting of cropland emissions and sinks within the National Carbon Accounting System is based on estimates using the Full Carbon Accounting Model *(FullCAM)*.

2.2 National Carbon Accounting System

The National Carbon Accounting System (NCAS) was developed to provide a comprehensive system to report on Australia's land-based greenhouse gas emissions and sinks. The analysis and reporting of the NCAS includes all carbon pools (biomass, dead organic matter and soil carbon) and all principal greenhouse gases (CO_2, CH_4 and N_2O), and can be applied at a variety of scales to both forestry and agricultural land uses. The national scale, fine spatial and temporal resolution, and breadth of data (climate, soil type, productivity, land cover and management information) that are combined in FullCAM, provide comprehensive data and a unique modelling capability. By incorporating five sub-models, including the carbon accounting model for cropping and grazing systems *(CAMAg)*, the microbial decomposition model *(GENDEC)*, and the Rothamsted Soil Carbon Model (Roth-C), the terrestrial ecosystem model FullCAM manages to cover biological and management effects on carbon pools, as well as transfers between different pools, including the atmosphere.

The report details how the NCAS estimates changes in soil carbon, and provides information on the data upon which these estimates are based, including soil mapping and inventory, calibration and validation of the Roth-C soil carbon model, and environmental, management and biomass data (e.g. climate, soil type, landuse and management data, crop growth and plant parameter, partitioning of crop residues, carbon content of crops and grass species, crop litter and decomposition rates, crop carbon turnover rates). At present, the NCAS accounts only for crop residues as input into soil carbon pools, but does not account for external inputs that replenish soil carbon pools, such as compost, manure, biosolids or sugarcane residue (mill mud). However, because of the very detailed nature of the NCAS, it should be relatively easy to expand the existing system to account for increase in soil carbon through external organic inputs into farming systems.

2.3 Soil carbon and climate change

The soil organic carbon pool under undisturbed natural vegetation can range widely between about 40 and 400 tonne carbon per hectare (t C ha^{-1}), depending on soil properties, profile characteristics, landscape position, temperature, and rainfall. This carbon pool is rapidly depleted when natural ecosystems are converted to agricultural production systems. The magnitude of depletion in soils of temperate climates is estimated to range between 25% and 50% after 20 to 50 years of agricultural use following deforestation. In the tropics it may be up to 50% or 75% during 5 to 20 years of agriculture after deforestation. Soil organic carbon losses in agricultural systems are caused by accelerated mineralisation, erosion and leaching, and are enhanced by regular ploughing, planting, and harvesting.

This situation is no different in Australia, with estimates that soil carbon stocks have decreased by 30% to 60% in Queensland since native soil was cleared for cropping about 100 years ago, and data from Victoria indicates that soil carbon levels had fallen by 50% after 20 years of continuous annual cropping. Carbon levels often decline for long periods of time after deforestation and landuse change until a new steady state is reached, where the rate of formation of new soil organic carbon from organic residues equals the rate of soil organic carbon decomposition. However, carbon emissions due to landuse change are not confined to historic times and forest-clearing; this is an ongoing process, which can be brought about by intensification of agricultural production systems.

The world-wide decline in soil organic carbon levels has resulted in carbon dioxide emissions, which are thought to have contributed significantly to increased atmospheric carbon dioxide levels.

2.4 Nitrogen and climate change

A number of agricultural activities increase mineral nitrogen availability in soils directly or indirectly, thereby increasing the amount available for nitrification and denitrification, and ultimately the amount of nitrous oxide (N_2O) emitted. Denitrification, which occurs primarily where soil oxygen is depleted, is probably the main source of N_2O, although it can also be generated by nitrification. Both nitrification and denitrification are an integral part of naturally occurring nitrogen transformations. Emissions of nitrous oxide depend primarily on the conditions detailed below.

2.4.1 Moisture and aeration

Maximum N_2O emissions usually occur when soil water content provides for an abundance of aerobic and anaerobic sites. For example, irrigation or rainfall events result in high N_2O emissions, especially when nitrogen fertiliser is applied with irrigation or soon before the irrigation event. Thus soil water content within the range of 55–85% water-filled pore space gives maximum N_2O emissions from denitrification and/or nitrification, while higher water content favours denitrification towards N_2 formation.

2.4.2 Temperature

Nitrification and denitrification, with which N_2O emissions are associated, primarily occurs when temperatures are between (approximately) 25°C and 35°C. N_2O emissions also increase with rising soil temperature, at least up to 35°C.

2.4.3 Soluble and readily decomposable carbon

Generally speaking, the addition of degradable organic materials increases N_2O production in soils that contain nitrate (NO_3-), or that receive either fertiliser NO_3- or materials containing degradable organic nitrogen (e.g. animal and green manures). As easily degradable organic compounds are the source of energy for denitrifying organisms, and because oxygen consumption in soils with high levels of decomposable carbon is normally high, the addition may result in anaerobic microsites.

2.4.4 Nitrogen from soil and fertiliser

The rate of N_2O emission after fertiliser application is influenced by the amount and type of fertiliser nitrogen, soil properties, environmental conditions and how all of these factors interact. Generally, the rate of denitrification increases with increasing NO_3- content in soil under conditions suitable for denitrification (e.g. high moisture). Although, N_2O production often declines within a few hours or days after NO_3- addition, and N_2 production increases, even when relatively high concentrations of NO_3- are still present. Under aerobic soil conditions, fertilisation with ammonium (NH_4^+) can result in marked N_2O emissions.

2.4.5 Soil pH

The optimal pH for both nitrification and denitrifcation is approximately 7–8. Hence, at pH levels above 8.0 and below 5.5 to 6.0, N_2O emissions can be several times higher than in neutral soil.

In the USA, it was estimated that direct emissions account for 82.5% and indirect emissions account for 17.5% of agricultural N_2O emissions, with most emissions (direct and indirect), being generated from cropland (69%), and less from grassland (31%). Emissions from synthetic fertiliser accounted for around 40% of direct N_2O emissions from agricultural cropland (mineral soils). Mineralisation and asymbiotic nitrogen fixation accounted for approximately 46%, and organic amendments for around 8% of emissions. Surface leaching and run-off in cropland accounted for 45% of all indirect N_2O emissions across all landuse types.

2.5 Potential for mitigating climate change in agriculture

The opportunities for mitigating GHG emissions in agriculture comprise:

- reduction of emissions by more efficient management of carbon and nitrogen flows in agricultural ecosystems
- removal of atmospheric carbon by increased biomass production and soil carbon sequestration
- avoidance of emissions by utilising crops and residues from bio-energy production.

Croplands offer many generic opportunities to impose practices that reduce net GHG emissions, including:

- improved agronomic practices that increase yields and generate higher inputs of carbon residue and can lead to increased soil carbon storage
- more carbon intense crops (e.g. perennials and orchards)
- more efficient nitrogen use from all sources that can reduce direct and indirect N_2O emissions and limit the use of nitrogen fertiliser, indirectly reducing GHG emissions associated with the manufacturing of nitrogen fertiliser
- reduced tillage that decreases soil carbon losses due to lower decomposition and erosion, and hence often (but not always) results in soil carbon gains
- improved and expanded irrigation (where water reserves allow) that can enhance carbon storage in soils through enhanced yields and residue returns
- restoration of degraded agricultural land that offers opportunities for carbon sequestration.

However, due to the complex nature of natural processes within agricultural ecosystems, certain management changes often affect more than one gas, through more than one mechanism, sometimes in opposite ways and on different temporal patterns, fully or partly negating other GHG benefits. Hence, the net benefit depends on the combined effects on all gases.

2.5.1 Technical and economic mitigation potential

Of the estimated global technical agricultural mitigation potential by 2030 (~4,500-6,000 Mt CO_2-e yr^{-1}), about 89% is from soil carbon sequestration, about 9% from mitigation of methane and about 2% from mitigation of soil N_2O emissions. As far as management options go, cropland management, grazing land management and restoration of organic soils and degraded land provide the highest potential.

It is recognised that soil carbon levels cannot be returned to their original value while used for agricultural production, with suggestions that no more than 60% to 75% of carbon lost in landuse change can be returned. Based on the premise that 75% of carbon lost after converting native land to cropping can be restored, it was estimated that up to 31.6 Mt C can be sequestered in cropped agricultural soil in Queensland, based on average sequestration potentials of 11 t C ha^{-1} for lower (500–800 mm yr^{-1}) and 14.8 t C ha^{-1} for higher rainfall (800–2,000 mm yr^{-1}) areas. However, merely optimising inputs and reducing losses will deliver considerably lower carbon sequestration gains than when external carbon sources are added.

The marginal abatement curve for agricultural mitigation practices (the amount of mitigation achieved for a given carbon price) showed that cropland management activities are the most cost effective agricultural GHG mitigation measure. These activities achieved around 50% of the technical mitigation potential at modelled carbon prices of up to US$20 t CO_2-e^{-1}. It is clear that land managers are reluctant to adopt GHG mitigation techniques unless they improve profitability and they will favour options that both reduce GHG emissions and increase productivity, over those that reduce emissions alone.

2.5.2 Mitigation by means of sequestering soil carbon

The soil carbon content can be raised by increasing the carbon input, decreasing the output or a combination of both. Soil carbon sequestration is the managed removal of atmospheric CO_2 by higher plants and micro-organisms, and its transfer into the stable soil carbon pool.

While organic matter was previously differentiated mainly into humic substances, soil organic matter is today largely characterised on the basis of physical fractionations. This differentiation is based on size and/or density. These factors better relate to the role organic matter components play in soil structure and soil function, and help to partition organic matter into components that differ in their lifespan chemistry, and origin. The domestic and international terminology for these organic matter fractions has not yet been settled, although the following have been proposed and used in Australia:

- surface plant residue
- buried plant residue
- particulate organic carbon
- humus
- dissolved organic carbon
- resistant organic carbon.

For convenience, the organic soil carbon pools are often grouped into three pools according to the speed of breakdown and replacement:

- ***fast*** - this pool has a short turnover time, with fast decomposition (e.g. daily to annual); also referred to as the *labile* or *active* pool
- ***slow*** - this pool has a longer turnover time, with slower decomposition (e.g. annual to decadal); also referred to as the *stable* or *humus* pool
- ***passive*** - this pool has a much longer turnover time (e.g. decadal to centennial/millennial); also referred to as the *recalcitrant* or *refractory* pool.

In addition to the above three soil carbon pools, the FullCAM model refers also to recalcitrant carbon, but in that case it represents charred carbon.

The proportion of total soil carbon in each pool can vary widely, but is assumed to be in the range of 10% for the fast pool, 40% to 80% for the slow pool, and 10% to 50% for the passive pool. These pools contribute approximately 10%, 40% and one% respectively to soil CO_2 emissions (respiration), with the remaining 50% contributed by live roots. From a carbon sequestration point of view, it is most desirable to increase the amount of organic soil carbon in the slow and passive pools due to their relative stability, although one has to be aware that active carbon responds to management changes more rapidly than total soil carbon. For example, recovery of total organic carbon after landuse change relies primarily on increases in particulate organic carbon with slow recovery of humus, while recalcitrant organic carbon is often unaffected by short to medium term management changes.

2.5.2.1 Effects of agricultural management practices

Land management practices affect carbon sequestration and loss rates, and the degree to which this happens depends on a wide range of variables, including site conditions, climate, and vegetation. Due to the variability in effects, it proved difficult to provide average carbon sequestration yields for certain measures. Work is continuing in this area.

In order to get a better handle on potential soil carbon sequestration levels attributable to different management practices, climatic zones are differentiated into *cool to dry, cool to moist, warm to dry* and *warm to moist*. Accordingly, it is estimated that the following average global carbon sequestration rates can be achieved over a 20-year period:

- agronomy 0.29 (dry) to 0.88 (moist) t CO_2-e ha^{-1} yr^{-1}
- nutrient management 0.26 (dry) to 0.55 (moist) t CO_2-e ha^{-1} yr^{-1}
- tillage/residue management 0.15–0.33 (dry) to 0.51–0.70 (moist) t CO_2-e ha^{-1} yr^{-1}
- water management 1.14 (all climates) t CO_2-e ha^{-1} yr^{-1}
- manure/biosolids use 1.54 (dry) to 2.79 (moist) t CO_2-e ha^{-1} yr^{-1}
- compost 1.1–1.8 t CO_2-e ha^{-1} yr^{-1}.

Although these and other data vary widely and have a high uncertainty level, the addition of external organic soil amendments usually has one of the highest carbon sequestration rates. Use of compost is not considered separately in most cases, except for European Union reports. The rate of sequestration is said to range from negative or zero under arid and hot climates, to approximately 3.67 t CO_2-e ha^{-1} yr^{-1} under humid and temperate climates, with normal rates of soil organic carbon sequestration in agricultural soils set at 1.1 – 1.8 t CO_2-e ha^{-1} yr^{-1}.

In the USA, in particular, it is believed that high sequestration rates are obtained with no-till farming, retaining crop residue as mulch, growing cover crops in the rotation cycle and adopting complex farming systems, as well as integrated nutrient management, including manuring. As minimum and no-till agriculture has never been demonstrated to achieve the same level of carbon sequestration in Europe and Australia as reported from the USA, the promotion of no and minimum-till agriculture as one of the optimal management practices for increasing soil organic carbon in the USA and Canada has recently been questioned.

2.5.2.2 Limits to carbon sequestration

Apart from soil, environmental and management constraints, there are other factors that can limit soil carbon sequestration, including:

- ***nutrient supply*** - the humification process can be severely constrained by the lack of nitrogen, phosphorous, sulphur and other building blocks of soil humus
- ***sink saturation*** - changes in carbon input and/or decomposition rates change soil carbon stocks, moving them towards new 'equilibrium' carbon levels. This is why soil carbon sequestration rates are non-linear and diminish over time, often reaching their maximum soon after a change in land use or land management. While the new equilibrium may not be reached for a long time after such change, carbon sequestration potential may be minimal after 20 to 50 years. Soil carbon sequestration does not, therefore, have limitless potential to offset CO_2 emissions. The potential for carbon sequestration depends on the carbon equilibrium of the current practice relative to the saturation level (optimal carbon level) that can be reached with the new practice at the given soil and environmental conditions
- ***reversibility*** - sequestration of carbon in soils is not necessarily permanent, as changes in agricultural management or land use, can result in declining soil carbon levels. Once equilibrium, or a certain content of carbon, is reached, it is still necessary to add significant amounts of carbon to maintain the attained soil carbon level. For example, cropped soil with 50% clay requires >2.2 t C ha^{-1} annually (or 5.5 t ha^{-1} of biomass of 40% carbon) to maintain a given carbon level, while cropped soil containing 30% clay requires >6.5 t C ha^{-1} (or 16 t ha^{-1} of biomass, including root biomass). In addition, the rate of carbon input has to be higher at higher existing soil carbon levels, in order to maintain soil carbon stock at that level.

2.5.3 Reduction of nitrous oxide emissions

Nitrous oxide emissions cannot be ignored due to their high global warming potential, the permanency of any reductions, and the potential of nitrous oxide emissions to annul carbon sequestration gains. The primary consideration for mitigating nitrous oxide emissions from the agricultural sector is to match the supply of mineral nitrogen and keep it to a minimum, commensurate with the needs of crops. Management practices to minimise nitrous oxide emissions from nitrogen fertilisers and legumes, as well as ways of improving nitrogen use by crops and/or reducing application rates, include the following:

- applying fertiliser nitrogen at optimum rates by taking into account all nitrogen sources available to the crop from soil, legumes, manure or compost
- applying fertiliser nitrogen at the rate and time to meet crop needs
- avoiding fertiliser nitrogen application outside the crop-growing season, and during fallow periods providing guided fertiliser nitrogen application through crop monitoring, yield maps and soil tests
- applying other nutrients if required to balance crop nutrient supply and optimise nitrogen utilisation
- avoiding surface application, incorporate or band so that fertiliser nitrogen losses are minimised and plant utilisation maximised
- incorporating manure immediately after application, and accounting for nitrogen release from organic soil amendments in nutrient budget.

2.5.3.1 Zero/reduced tillage

The effects of conservation tillage practices on nitrous oxide emissions are variable, with some reports demonstrating that emissions increased, while others showed exactly the opposite. It is suggested that the impact of no-till on nitrous oxide emissions is small in well-aerated soils but most often positive in soils where aeration is reduced by restricted drainage.

2.5.3.2 Fertiliser use

There is a marked relationship between nitrous oxide emissions and increased nitrogen application rates, that may be linear or non-linear. Agricultural nitrous oxide fluxes can be reduced, with no or little yield penalty, by reducing nitrogen fertiliser inputs to levels that just satisfy crop needs. Other options for reducing nitrous oxide emissions include avoiding wet soil conditions, applying nitrogen fertiliser as NO_3- or as controlled-release and stabilised nitrogen, and subsurface fertiliser placement.

For some, increasing crop nitrogen-use efficiency by altering nitrogen management practices is the most promising strategy for mitigating agricultural nitrous oxide emissions across a range of cropland systems.

2.5.3.3 Crops

Rotational cropping is already practiced in many areas but optimising crop selection, considering in particular the addition of nitrogen fixing crops, can increase soil carbon storage and reduce mineral fertiliser requirements. While biological nitrogen fixation does not seem to be a measurable source of nitrous oxide *per se*, inclusion of legumes in corn and cotton rotations has increased the level of emissions during the subsequent year of rotation.

2.5.3.4 Australian data

Nitrous oxide emissions in Australian agricultural systems were determined mainly in dryland cropping, cotton, and sugarcane. Nitrous oxide losses in cotton (NSW) and dryland wheat cropping in subtropical south east Queensland accounted for 0.16 and 0.36% of added nitrogen in cotton, and between 0.54–0.91% in wheat cropping, with emissions from conventional tillage being higher than from no-till agriculture. Emissions from pineapple and macadamia in the high rainfall south east Queensland region indicate losses of 2.1–4.7% (0.8–1.4 kg N ha^{-1}) of applied fertiliser nitrogen, and of up to five kg N ha^{-1} from non-fertilised pasture with high soil carbon (> 4% in 0–10 cm layer).

The assessment of emissions from sugarcane in northern NSW and central Queensland vary significantly. A site in northern NSW with acid sulfate soil recorded soil moisture conditions that were near optimal for nitrous oxide production for around five months, resulting in the emission of 45.9 kg N ha^{-1}, or 22% of fertiliser applied nitrogen. Even unfertilised plots emitted 11.3 kg N ha^{-1}. In contrast to most other (sugarcane) soils, acid sulfate soil appeared to be a source of methane, not a sink. The net sequestration of CO_2 by the crop from the atmosphere of 51.4 t ha^{-1} during the growing season was offset by nitrous oxide and methane emissions amounting to 23 t ha^{-1} CO_2-e. On the other hand, nitrous oxide emissions in central Queensland on sandy loam amounted to 3% of the applied nitrogen and no methane emissions were recorded, despite high rainfall and green sugarcane trash blanketing (mulch). The main drivers of nitrous oxide emissions appeared to be the availability of surplus nitrogen and the water content of the near-surface soil.

2.5.4 Outlook

There are significant opportunities for GHG mitigation in agriculture, but for the potential to be realised, numerous barriers must be overcome - including climate and non-climate policy, as well as institutional, social, educational and economic constraints. The mix of agricultural mitigation options that are adopted in the future will also depend upon the price of carbon dioxide equivalents. With appropriate policies, education and incentives, it may be possible for agriculture to make a significant contribution to climate mitigation by 2030, creating offsets of between 5% and 14% of total annual CO_2 emissions, depending on the price for carbon.

3 Effects of using organic soil amendments on climate change

The promotion of increased carbon input from organic amendments (animal manure, compost, crop residues, sewage sludge) is one of the most efficient measures for soil carbon sequestration, according to the European Union Working Group on Sinks Related to Agricultural Soils. Despite this recognition, so far only limited work has been undertaken to investigate the effects of organic soil amendments on climate change, particularly when it comes to the use of compost. Work carried out in Europe related to compost use seems to have focused primarily on carbon sequestration without paying much attention to N_2O emissions. Nevertheless, information regarding the use of animal and green manure is available from Europe and North America, and also from warmer environments such as India, China, and Africa.

There does not seem to be a comprehensive, Australia-wide inventory of potentially available resources, despite increased interest in utilising organic residues and by-products, both for improving soil properties and generating fuel or energy. Nevertheless, it is clear that apart from municipal organics (about 10.6 Mt, of which 3.7 Mt is already converted into recycled organic products), there are large quantities of agricultural and food/fibre processing residues that can be utilised beneficially.

3.1 Organic farming

Organic farming systems rely on high internal or external organic matter inputs, and on carbon and nutrient cycling for maintaining soil quality and productivity. Although it can be assumed that composted organic residues are often used in organic farming systems, reported carbon sequestration, changes in nitrous oxide emissions, and the overall potential for organic farming to reduce GHG emissions are by no means due to compost use alone, but a wide range of farm management practices.

3.1.1 Carbon sequestration

For decades, organic agriculture was at the forefront of systematically developing and optimising the quantity and application of organic manures, based on the principles of integrating crop production and animal husbandry, and the recycling of organic residues. Likewise, organic residue processing techniques were improved to obtain high quality manure, and composting was employed to minimise losses and increase humus production.

Modelling for Northern European conditions suggested that conversion from conventional to organic farming practices increases SOC levels by 100–400 kg ha^{-1} annually for the first 50 years, but actual long-term trials and farm-level comparisons conducted in the USA and Europe that are presented in this report provided diverse results. Virtually all such trials showed superior carbon sequestration for organic farming systems compared to conventional practices, with carbon sequestration rates of 180 kg C ha^{-1} yr^{-1} for an organic farming system with high animal density in Southern Germany (14 years) for example, or 981 kg C ha^{-1} yr^{-1} for a manure-based organic farming system in Pennsylvania, USA (22 years). Despite the fact that not all organic farming system trials showed increasing SOC levels, for example on sandy soil in Germany (18 years), most organic farming systems with that experience still managed to reduce the decline of soil organic carbon in comparison to conventional farming systems.

3.1.2 Nitrous oxide emissions

The highest nitrous oxide efflux in organically managed soil usually occurs after manure application and incorporation of legume (cover) crops. Nevertheless, most studies that compared nitrous oxide emissions from organic and conventional farming systems concluded that organic farming results in lower emissions than conventional production. Comparison of an organic and a conventional farm in southern Germany, for example, showed that nitrous oxide emissions (mainly from soils) contributed about 60% of total GHG emissions in both farming systems, representing, on average, 2.53% of total nitrogen input by synthetic fertilisers, organic fertilisers and crop residues. Mean emissions from organic and conventional crop production (emissions from cropping, fertiliser production, and the consumption of fossil fuels for field management and drying of crops) amounted to 3.2 and 4.4 t CO_2-e per hectare field area respectively, and for winter wheat alone, emissions amounted to 1.48 and 2.5 t CO_2-e ha^{-1}.

Comparison of nitrous oxide emission rates in crop rotations of organic and conventional dairy farms across five locations in Europe showed a significant relationship between total nitrogen inputs and nitrous oxide emissions, with annual average nitrous oxide losses amounting to 1.6 ± 0.2% of total nitrogen inputs. Modelling of GHG emissions from organic and conventional dairy farms in Europe showed that the emissions at farm level could be related to either the farm nitrogen surplus or the farm nitrogen efficiency. The farm nitrogen surplus appeared to be a good proxy for GHG emissions per unit of land area, as GHG emissions increased from 3.0 t CO_2-e ha^{-1} yr^{-1} at a surplus of 56 kg N ha^{-1} yr^{-1} to 15.9 t CO_2-e ha^{-1} yr^{-1} at a surplus of 319 kg N ha^{-1} yr^{-1}. Organic farms showed considerably lower nitrogen surpluses than conventional farms. The GHG emissions per product unit were quite closely related to the farm nitrogen efficiency, and doubling of the nitrogen efficiency from 12.5% to 25% reduced the emissions per product unit by approximately 50%. Farm nitrogen efficiency may therefore be used as a proxy for comparing the efficiencies of farms with respect to supplying products with low GHG emissions. It was shown that organic farming usually manages to reduce emissions per unit land area, but not necessarily per unit of product.

Most research found that, in comparison to conventional agriculture, organic farming resulted in significantly reduced nitrate leaching rates, which reduces the risk of indirect nitrous oxide emissions.

3.1.3 Multilevel assessment of using soil amendments

An evaluation of using mineral fertiliser and different organic soil amendments application on yield, soil quality, soil health, environment and climate change after completion of the first eight-year crop rotation in an organic farming systems trial showed that no product scored high in all of the five chosen agricultural and societal criteria. On the light sandy clay soil, use of farmyard manure resulted in the highest crop yields (in 2006) and, together with compost products, showed a slight increase in SOC levels over the eight-year trial period.

Table 1 Qualitative evaluation of using different soil amendments **(Source: Koopmans *et al.* 2008[*])**

Criteria	Yield	Soil Quality	Soil Health	Environment	Climate Change
Indicator	Fresh matter yield 2006	Nitrogen supply	Parasitic nematodes	P surplus and NO_3 leaching	SOC sequestration
Mineral fertiliser	-	-	-	+	0
Poultry manure	0	0	+	-	0
Farm yard manure	+	+	0	0	+
Cattle slurry	-	+	0	0	-
Household compost + slurry	0	+	+	-	0
Plant compost 1	0	+	-	+	+
Household compost	0	-	0	+	+
Plant compost 2	-	0	0	+	+

- = negative effect, + = positive effect and 0 = no effect

3.2 *Using manures*

It was estimated that approximately 3.22 Mt of animal manures are generated annually in Australia, comprising approximately 1.33 Mt of feedlot manure, 1.56 Mt of poultry manure, and 0.32 Mt of pig manure. These quantities are similar to the amount of municipal organics that was recycled in 2006–07, but are relatively small compared to about 62 Mt of collectable manure produced in the USA in the mid 1990s and some 90 Mt of animal manure being land-applied annually in the UK.

At present, the vast majority of animal manures are used as soil amendment for supply of plant nutrients and improvement of soil properties. Using animal manures for land management purposes can deliver agronomic and environmental benefits but also have certain drawbacks and risks.

3.2.1 Decomposition

Decomposition of animal manures in soil depends upon factors such as type of manure, feeding ration, animal age, living conditions of animals, and how manure is handled, as well as moisture and temperature conditions in the soil. Comparison of 47 animal manures of different origins, physical properties and processing levels, showed that carbon mineralisation varied from 5% to 62% of the organic carbon added during the 224-day incubation. Seventy per cent of the manures induced net nitrogen mineralisation at the end of incubation, ranging from 3% to 51% of organic nitrogen. Some manures induced net soil inorganic nitrogen immobilisation, ranging from -1% to -31% of the organic nitrogen added. Carbon present in the water-soluble fraction at 20°C gave the best indication of carbon mineralisation and the organic nitrogen content or carbon to nitrogen ratio were the best indicators for nitrogen mineralisation.

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[*] Koopmans, C. J., M. Zanen and J. G. Bokhorst (2008). Organic Fertilisers of the MAC Trial and their Impact on Soil Quality, Environment and Climate Change. The 5th International Scientific Conference on Sustainable Farming Systems - ECOMIT, . Slovakia.

3.2.2 Soil carbon

3.2.2.1 Increase in soil organic matter

Research has shown that manure application has a significant effect on chemical, physical and biological soil properties and that most of these effects are due to an increase in soil organic matter. The increase in soil organic matter with manure addition was reported in many studies over a wide variety of soil textures and climates for diverse agricultural cropping systems. The rate of increase in soil organic matter depends on temperature, moisture and tillage conditions, as well as the amount of manure added. Manure needs to be applied for at least two years to see increases in soil organic matter. Higher/longer applications are needed to produce measurable changes in soil properties, particularly physical properties. Where that was not achieved, manure application reduced soil organic matter losses in production systems under tillage.

It appears that long-term use of manure can increase carbon levels in all soil particle size fractions (clay, silt, sand).

Results of long-term field trials are particularly important in evaluating the effects of using manure on soil organic matter levels. The marked and sustained increase in soil organic matter due to long-term use of manure compared to mineral fertiliser in continuous wheat cropping was shown in the Broadbalk Long-Term Experiment (Rothamsted Research Station, UK) where, after 160 years, carbon levels were around 2.5-fold higher in soil that had received manure (35 t ha^{-1} yr^{-1}), compared to soil that received mineral fertiliser or no amendments. As can be expected, the development of soil nitrogen levels in the three treatments followed a very similar pattern to that observed for soil carbon. However, this meant that nitrogen lost in drainage water (1990–98) was also relatively high -approximately 40 and 80 kg N ha^{-1} yr^{-1} where manure and manure + fertiliser (96 kg N ha^{-1} yr^{-1}) were used.

A similar long-term trial at Woburn (also at the Rothamsted Research Station) demonstrated that it is much more difficult to maintain or increase soil organic matter in a sandy loam (8–14% clay) than it is on the silty clay loam (20–40% clay) at Boardbalk. The Rothamsted results were confirmed by a similar trial in Bad Lauchstädt (Germany), where 75 years of unchanged agricultural practices resulted in new soil organic carbon equilibria, which varied between 1.61% carbon for the unfertilised soil and 2.27% carbon for the soil that was amended with manure (15 t ha^{-1} yr^{-1}) plus mineral fertiliser (Figure 1). The reversal of the trial's high input (15 t ha^{-1} yr^{-1} manure + N and no input treatments) after 75 years also clearly demonstrated the reversibility of carbon sequestration.

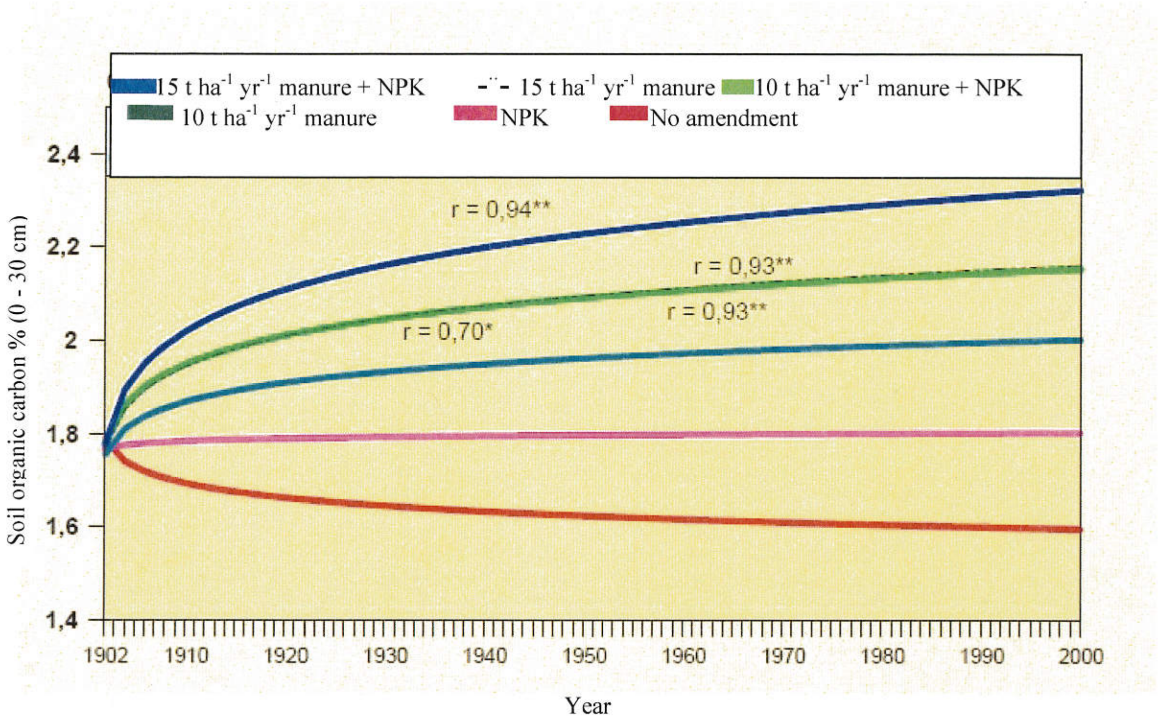

Figure 1 Development of soil organic carbon levels in the main treatments of the static fertilisation experiment Bad Lauchstädt (Source: Körschens 2009*)

A great many short, medium and long-term field trials were conducted in diverse environments, soils and agricultural production systems, in which the effects of manure application on soil carbon pools was determined. The results of many such trials are described in the report, but because of their divergent nature are summarised here for different climatic zones. The average percentage (and range) of carbon applied as manure that was retained in the soil was found to be:

- 23% (± 15%) in temperate or frigid regions
- 7% (± 5%) in thermic regions
- 8% (± 4%) in moist regions
- 11% (± 14%) in dry regions.

Hence, for all regions except for cold climates, it can be expected that between 5% and 20% of carbon applied with manure is retained and incorporated into the soil carbon pool. Over all, the data gathered in the UK indicated that 23% of the carbon applied was retained in the topsoil, which means that soil organic carbon increased by an average of 60 (±: 20) kg ha^{-1} yr^{-1} per tonne of manure DM applied. Comparison of ten long-term agricultural experiments in Germany showed that optimum fertilisation (organic + mineral) resulted in soil organic carbon levels that were between 16% and 76% higher than in soil that had no fertilisation, with increases at most sites ranging between approximately 30% and 50%. Most, but not all trials where organic and mineral fertilisation were combined, reported superior carbon storage, and in most cases increased yield responses were also recorded.

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[*] Körschens, M. (2009). Humus – Voraussetzung für Bodenbildung und Bodenfruchtbarkeit [Humus - a pre-condition for the development and fertility of soils] (PPT). Naturland - Ackerbautagung 2009. Bernburg - Strenzfeld, Germany, Landesanstalt für Landwirtschaft, Forsten und Gartenbau Sachsen-Anhalt.

The analysis and evaluation of many long-term agricultural field experiments, combined with in-depth knowledge about carbon and nitrogen dynamics led senior soil scientist[*] to draw the following conclusions:

- improved soil organic matter levels, and hence improved soil properties, contribute up to 10% of crop yield on sandy soils, and up to 5% on loamy soils
- the highest yields are only attainable in an environmentally acceptable way if organic and mineral fertilisation are combined
- changes to the soil organic carbon equilibrium due to management changes occur very slowly and affect almost exclusively the decomposable carbon fraction. Whether management changes (including fertilisation) result in increasing or decreasing SOC levels depends on the initial soil carbon level
- hot water extractable carbon has proved to be an appropriate criterion for the characterisation of the decomposable carbon
- considering texture, environmental conditions and agricultural practices, it is possible to determine optimum soil organic carbon levels in agricultural soils
- the ranges of optimum soil organic carbon and nitrogen levels are relatively narrow. Under conditions in Central Europe (mean annual temperatures 6-10°C, annual precipitation 450-800 mm), the optimum content of mineralisable carbon ranges between 0.2% and 0.6% of total carbon, and optimum nitrogen levels range between 0.02% and 0.06%
- soils with lower carbon and nitrogen levels are prone to encounter soil fertility problems, experience yield depression, and plant biomass will absorb less CO_2, while soils with higher levels might pose environmental risks (nutrient leaching)
- use of optimum soil organic carbon levels (standard values) together with methodology for developing 'humus balances' allows management of soil organic matter content in arable soils that ensure high yields and prevent detrimental environmental effects
- increase in soil organic carbon by 0.1% results in improved soil physical properties that approximately amount to:
  - an increase in hygroscopicity of 0.06% to 0.08% (w/w)
  - an increase in water holding capacity of 0.4% to 0.6% (w/w)
  - a decrease of dry matter density of 0.004 to 0.005 g/cm^3
  - a decrease of bulk density of 0.006 to 0.008 g/cm^3
- due to increased stability of its carbon compounds, compost is suitable to increase the SOC level of sand and clay soils and hence to improve their physical characteristics.

3.2.3 Composting of manure to enhance carbon sequestration

Composting of animal manures reduces volume, limits odours, stabilises nutrients, kills weed seeds and pathogens, and reduces volatile organic compounds. Composting of manure also induces chemical changes that affect carbon cycling processes in the soil. Composting of cattle manure for 100 days, for example, doubled the proportion of humic substances from 35% before composting to 70% at the end of composting. Overall, it can be said that carbon in manures is lost during composting and transformed into more stable carbon forms, and because of this, the remaining carbon is less decomposable when applied to land. Unfortunately, considerable proportions of nitrogen are also frequently lost during the composting of manures. For example, composting of feedlot cattle manure resulted in 51±9% loss of carbon, and 31±12% loss of nitrogen, while composting of poultry litter for 65 days resulted in 16% loss of carbon, and 49% loss of nitrogen.

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[*] M. Körschens

Various incubation tests and field trials have shown that the use of composted manure results in retention of considerably higher proportions of applied carbon in soil than when raw manure is used, although there are also reports where no difference was observed. Several results showed that soil containing composted manure retained about three times as much carbon as soil with added raw manure.

In addition to aiding soil carbon sequestration, composting of manure yielded a more predictable and reliable source of mineralisable nitrogen than fresh manure, and composting of liquid pig manure (plus straw) was found to reduce N_2O emissions compared to using liquid pig manure.

3.2.4 Nitrous oxide and methane emissions

Laboratory incubation of soil to which different dairy manures (n=107) were added showed that denitrification accounted for approximately 5% of the added manure nitrogen. Potential denitrification is strongly influenced by mineralisable nitrogen and the water-soluble organic carbon content of manures. Slurries often contain greater concentrations of these important components than stacked or composted manures. Another incubation trial showed that sand with low cation exchange capacity emitted more GHGs (N_2O, CO_2 and CH_4) than clay with high cation exchange capacity.

Field trials with manure confirmed that the nitrous oxide flux significantly correlates with the mineral nitrogen and water-filled pore spaces in soil, and is also affected by the soil's redox potential. Most of the nitrous oxide emissions occur within a few weeks of manure application, but might also peak when rain or irrigation results in high water-filled pore space. It is suggested that the depletion of readily decomposable carbon within a few weeks of manure application is responsible for decreasing nitrous oxide emissions.

Depending on circumstances, trials showed higher, equal or lower nitrous oxide emissions when manure or slurry was used, compared with mineral fertiliser. For example, nitrous oxide losses from fertiliser use amounted to 3.3% and 0.2% respectively in sandy and heavy clay soil, while it accounted for 2.7% and 1.8% respectively of applied slurry nitrogen for sand and clay. In another trial, the cumulative nitrous oxide loss was 0.6% of total nitrogen applied as mineral fertiliser, 1.2% of total nitrogen applied with 60 t ha^{-1} yr^{-1} of swine slurry, and 1.7% of total nitrogen applied with 120 t ha^{-1} yr^{-1} of swine slurry.

3.3 Using biosolids and paper mill sludge

3.3.1 Biosolids

Long and short-term biosolids application trials in the UK showed average soil organic carbon increases of 130 (90–170*) kg ha^{-1} yr^{-1} per tonne of raw solids DM applied, and of 180 (130–230*) kg ha^{-1} yr^{-1} per tonne of digested solids DM applied. A higher proportion of carbon added with digested sludge (ca. 56%) was retained in the topsoil than with raw sludge (ca. 36%), and this supports the notion that, similar to composting, much of the easily decomposable carbon is lost during the digestion process, so that the organic carbon added with digested sludge is more resistant to decomposition than raw sludge carbon. Partitioning of carbon in composted sewage sludge and cattle manure into two components differing in ease of decomposition, showed that the labile component accounted for 16% to 20% of total carbon in both products, with a decomposition rate constant of 2.4×10^{-2} d^{-1} for labile carbon, and 1.2 to 1.4×10^{-4} d^{-1} for the resistant pool.

Strip mined land was reclaimed over 34 years with rotational biosolids application, at cumulative loading rates between 455–1,654 t ha^{-1} DM (over 8–23 years). The mean net soil carbon sequestration was 1.73 (0.54–3.05) t C ha^{-1} yr^{-1} in biosolids-amended fields, compared with -0.07–0.17 t C ha^{-1} yr^{-1} in fertiliser controls. Soil carbon

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* at 95% confidence interval (CI)

sequestration was significantly correlated with biosolids application rate. The equation for this correlation can be expressed as: $y = 0.064x - 0.11$, in which y is the annual net soil carbon sequestration (t C ha^{-1} yr^{-1}) and x is the annual biosolids dry matter application (t DM ha^{-1} yr^{-1}).

Characterisation and incubation of over 40 sludges of diverse properties, representing different treatment and stabilization processes, showed dramatic differences between sludges in their ability to release greenhouse gases (N_2O and CO_2), but failed to deliver a clear explanation of mechanisms controlling these differences. Some biosolids of similar properties represented entirely different emission behaviour. Other work showed that increasing application rates resulted in both increased nitrous oxide emission rates and higher cumulative emissions, but had mixed results in comparing nitrous oxide emissions from biosolids (processed and unprocessed) and mineral fertiliser.

3.3.2 Paper mill sludge

Around 700,000 tonnes of paper mill sludge was applied to agricultural land in England and Wales, but; there is no equivalent quantitative data available for Australia. With a 'typical' carbon content of around 30% DM, and a proportion of recalcitrant (lignin) and readily decomposable carbon comparable to farm manures, paper mill sludge clearly has the potential to increase soil organic carbon levels in the range of manures; i.e. by ca. 60 kg ha^{-1} yr^{-1} per tonne paper mill sludge DM applied. Nevertheless, paper mill sludges seem to vary somewhat, at least as far as nitrogen immobilisation is concerned, and, by extension, their various effects on nitrous oxide emissions.

3.4 Using compost

Composting is a partially controlled and accelerated bio-oxidative process through which highly diverse and heterogeneous organic material is decomposed and transformed into humified material. Composition of the raw materials to be composted determines CO_2 evolution during composting, and residual carbon levels in finished compost. Readily degradable components (e.g. sugar, starch, protein) are the most important source of carbon during the intensive composting phase, and are primarily broken down during this period. Lignin is the compound most resistant to biodegradation and strongly influences degradability of organic materials. Most labile organic compounds are broken down during the composting process, and are recorded as 'carbon loss', i.e. transformation of solid into gaseous (CO_2) carbon. The literature provides the following examples of carbon losses during composting:

• ~46% loss	open windrow composting of mixed food and garden organics (biowaste), average over 10 years
• ~50% loss	40–60 day open windrow composting of animal manure
• ~83% loss	9 months covered windrow composting of corn silage
• ~41–45% loss	9 months covered windrow composting of two-phase olive waste.

This means that around 50% of carbon contained in the raw materials is retained and found in compost, mostly in recalcitrant organic compounds such as humic substances. The formation and increasing presence of humic substances during the composting process has been documented on various occasions. It has also been shown that humic acids found in compost consist of a stable fraction and a labile fraction, the relative contents of which depend on the duration of the composting process.

In very broad terms, there are two basic compost products defined by their end use:

- composted mulch (70% of mass >15mm, applied to soil surface)
- composted soil conditioners, suitable for incorporation into the soil.

Due to a lack of data regarding the use of organic mulches, most compost related information in the following sections concerns use of compost as a soil conditioner.

The variability of the organic matter undergoing composting, the diversity of composting conditions, the levels of stability and maturity reached during composting, and the degree of product refining makes compost research a great challenge. To limit the variability in input materials, this section will deal primarily with composted soil conditioning products made from municipal and commercial residues other than manures, paper sludge and biosolids.

3.4.1 Fate of compost carbon after soil application

Compost is a complex mixture of inorganic and organic components, ranging from labile to resistant in terms of degradability, as well as dissolved organic carbon, which may be degraded or leached. As with other organic soil amendments, the rate and extent of mineralisation of compost products after application to soil depends on the quantity, type, maturity and particle size distribution of the applied product, as well as on soil properties, environmental conditions, and agricultural management practices.

3.4.1.1 Degradation

Distinct differences in the level of carbon degradation in soil were observed between different organic residues, and between composted and uncomposted materials. The longer organic materials were composted, the smaller the proportion of organic matter degraded after soil application. Again, the labile organic compounds contained in compost are degraded relatively quickly, and the recalcitrant fractions remain in the soil.

Research by means of incubation trials reported the following carbon losses after incorporating compost into soil:

• 5.4%	in 70 days at 25/20°C	tree prunings + yeast composted for 25 wks
• 21%	in 32 wks at 25°C	two sources of two-phase olive mill residue composted for 34 wks
• 14%	in 24 wks	20 samples of manure and plant based composts
• 5%	in 60 days	MSW + biosolids compost in two soils
• 8% and 14%	in 60 days	biosolids compost in two soils
• 5%	in 54 days	biowaste compost (food and organics)
• 17%	in 52 wks at 14°C	biowaste compost in mine site rehabilitation
• <20%	in 78 wks at 5 and 14°C	biowaste compost in mine site rehabilitation and agricultural soil.

Incubation of fresh (C:N ratio = 15.4) and mature (C:N ratio = 9.2) biowaste compost in sand for 112 days at five, ten, 15, 20, and 25°C showed cumulative carbon losses for fresh compost that ranged from 36% of the initial carbon content at 5°C to 54% at 25°C, while carbon losses were small for mature compost, ranging from 1% to 6% within the assessed temperature range of 5 to 25 °C (Figure 2).

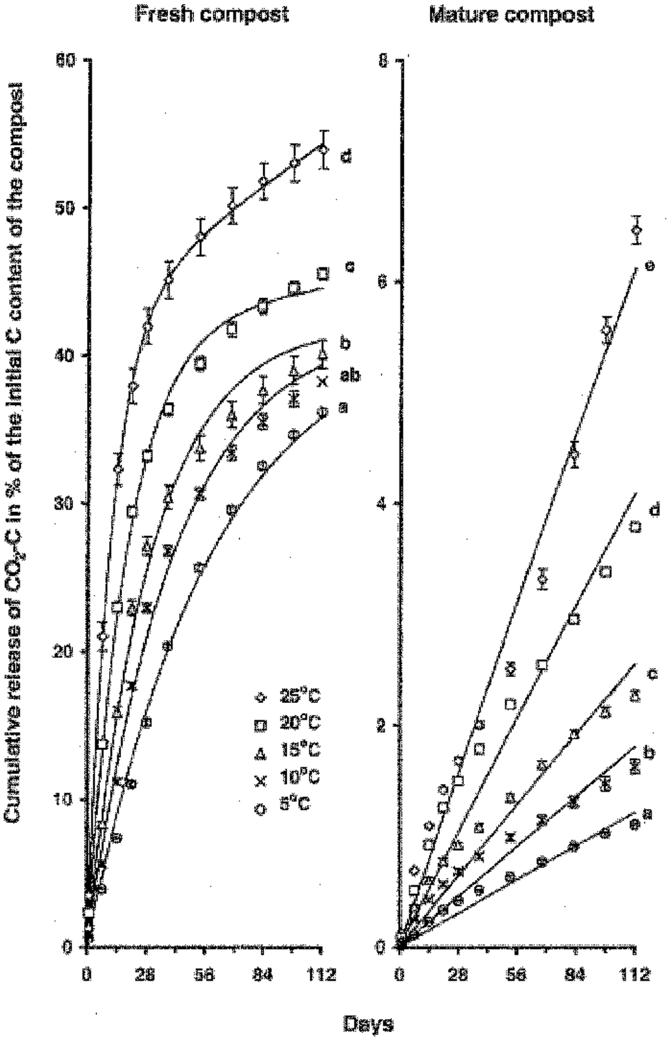

Figure 2 Cumulative CO_2 emissions from fresh and mature biowaste compost in sand during 112 days at various temperatures. Treatments with the same letter assigned are not significantly different ($p<0.05$) **(Source: Chodak et al. 2001)*.**

3.4.1.2 Sequestration

Scientific trials have established the application of compost to arable soils can increase SOC levels. This has been demonstrated in Europe (UK and Germany), North America and Australia as shown below.

For the **United Kingdom** it was estimated that, over a 20-year time frame, carbon retention is greatest from the addition of digested biosolids cake (56%) and composted garden organics (43%), and least from manure (23%) and straw (7%). These materials deliver between 370 and 1,500 kg ha^{-1} additional carbon annually, if the organic materials are applied at rates of 250 kg ha^{-1} total nitrogen. Estimates for garden organics compost are based on four medium-term (5-8 yrs) compost experiments at four locations, at which 50 t ha^{-1} of compost was applied annually or 100 t ha^{-1} bi-annually, with or without addition of inorganic nitrogen fertiliser.

On average, soil organic carbon increased by 60 (36–84) kg ha^{-1} yr^{-1} per tonne of compost DM applied. While this figure is similar to estimates for farm manure when expressed as a proportion of the organic carbon applied, soil carbon retention following compost use (43%) was almost double that from farm manures (about 23%), since composts have a lower carbon content (per tonne of DM) than farm manures. The proportion of applied carbon that remained in the soil varied between 28% and 60% for the three locations where compost was used for five years, with the eight-year trial showing 54% carbon retention. However, it is stressed that these increases in soil carbon can be regarded only as the initial rate of increase (up to about 20 years), as soil organic carbon accumulation rates decline with time.

Another large-scale, unreplicated trial (ten fields on four farms), in which garden organics compost was applied at rates of 0, 25, 50 and 75 t ha^{-1} to a range of different crops, showed that the two highest application rates arrested the decline in SOM between 2001–05. This was surprisingly high (30%) in unamended soil (average across all sites). Extension of the trial by two years (five fields on two farms) demonstrated that the longer compost is used, the more obvious its carbon sequestration benefits become. The lowest application rate (25 t ha^{-1}) did not increase SOM at the selected trial sites. This is a reflection of the significant decline in SOM at the trial sites and evidence that the lowest compost application rate halted the decline of SOM. Nevertheless, higher application rates increased SOM levels by about 20% (50 t ha^{-1}) and 30% (75 t ha^{-1}) over a seven year period.

Compost application trials that were conducted over a period of 9 and 12 years at five locations in **Germany** provide the most comprehensive data set to date regarding the medium-term effects of compost use in agriculture. Continuous compost application for 9 and 12 years resulted in organic matter level increases across all five sites by approximately 0.3 percentage points for every 5 t DM ha^{-1} of compost applied annually, and even by approximately 0.42 percentage points at the three cooler sites. Assessment of sites with a 12-year compost application history revealed a linear correlation between compost application rate and soil organic matter content. Averaged over all five trial sites, the addition of 8–9 t DM ha^{-1} of organic matter from compost over 9 and 12 years increased soil organic matter levels by approximately 0.1 percentage points. Annual compost application rates of 5 t DM ha^{-1} delivered between 20 and 35 t DM ha^{-1} of compost organic matter over the trial period, and this demonstrates that even low compost application rates can affect soil organic matter levels positively.

Annual compost application rates of 6.7 t DM ha^{-1} added 2.4–2.8 t DM ha^{-1} of organic matter, or around 1.5–2.2 t ha^{-1} of carbon to soil. Where mature compost was applied, it was estimated that around 50% of the added carbon, i.e. 0.75–1.1 t ha^{-1}, contributed to humus production, at least in the short term. These were quite sizable additions to the humus carbon pool, considering that in order to maintain soil humus levels, around 0.3 t ha^{-1} of carbon has to be added annually to the humus carbon pool of average soils in the region where the trials were conducted.

The capacity of compost products to increase soil humus levels has been studied over a 12-year period. The main findings are presented below:

- proportion of organic matter added with compost retained in the topsoil (0–30 cm):
    - mature garden organics compost (used on silty loam) 50.6%
    - mature biowaste compost (used on loamy sand) 45.7%
    - pasteurised biowaste compost (used on silty clay loam) 29.7%.
- the annual increases in soil organic carbon recorded at compost application rates of 10 t DM ha^{-1} yr^{-1}:
    - mature garden organics compost (used on silty loam) 0.82 t C ha^{-1} yr^{-1}
    - mature biowaste compost (used on loamy sand) 0.97 t C ha^{-1} yr^{-1}
    - pasteurised biowaste compost (used on silty clay loam) 1.13 t C ha^{-1} yr^{-1}.
- the application of compost at rates of 5, 10 and 20 t DM ha^{-1} yr^{-1} over 12 years did not result in the establishment of new organic matter equilibria in the soil. On the contrary, the increase of soil organic matter due to compost use observed in the 12th year was still linear
- long-term fertilisation field trials suggest that, as an average for Germany, around 0.4 t ha^{-1} of carbon has to be added annually to the humus carbon pool of soils to maintain current levels, if no crop residues are added. Hence, in order to maintain current soil organic matter levels, the following compost quantities need to be applied annually:
    - mature garden organics compost (used on silty loam) 4.8 t DM ha^{-1} yr^{-1}
    - mature biowaste compost (used on loamy sand) 3.8 t DM ha^{-1} yr^{-1}
    - pasteurised biowaste compost (used on silty clay loam) 2.6 t DM ha^{-1} yr^{-1}.

This medium-term compost application trial showed that the regular and agronomically appropriate use of compost (20–30 t DM ha^{-1} yr^{-1} over three years) is a suitable means for increasing and optimising soil organic matter levels.

Carbon storage on loam soil in **The Netherlands** over **almost 30 years** amounted to 6% for green manure, 18% for composted farmyard manure and 'Municipal solid waste (MSW) compost after preselection of the organic fraction'; 22% for composted sewage sludge and 40% for peat moss. In neighbouring **Belgium**, 20.1% of applied organic matter was recovered after applying 22.5 t ha^{-1} of biowaste compost annually for four years to silage corn monoculture grown in sandy loam.

Two long-term compost field trials were conducted in **Austria**. During a 13-year trial on silty loam (3.4% organic matter) the use of biowaste compost (8, 15 and 22 t ha^{-1} yr^{-1} on average) on its own or in combination with mineral fertiliser stored between 1,900 and 6,500 kg ha^{-1} organic carbon, which represents 10–19% of the organic carbon applied with compost. Results of a 12-year trial in which four types of compost (biowaste, garden organics, cattle manure, sewage sludge) were applied at annual rates of 175 kg N ha^{-1} yr^{-1} with and without additional nitrogen fertiliser (80 kg N ha^{-1} yr^{-1}) on a loamy silt (1.9% organic matter) showed that using compost made from garden organics and sewage sludge was most effective in raising soil organic carbon levels, effecting an increase of about 20%. The application of garden organics compost at 175 kg N ha^{-1} for 12 years, (ca. 11 t DM ha^{-1} yr^{-1}), sequestered around 780 kg C ha^{-1} annually, or 70 kg C ha^{-1} per tonne of compost DM applied per year.

Around 76% of carbon added with a single application of 50 and 85 t ha^{-1} of biowaste compost in **Italy** was recovered after four months. In a similar trial, in which composted food and ligno-cellulosic residues were applied at 50 and 85 t ha^{-1} to corn in silty-clay, it was found that, after five months, around 62% of compost applied carbon was retained in the soil for both application rates. The amount of carbon sequestered was similar to the content of recalcitrant carbon in compost, which amounted to 58.6% of total organic carbon in the compost.

The amount of compost needed to maintain soil organic matter levels in north-western **Spain** over a 25 year period at current mean soil temperatures (13.5°C) was estimated at 4.0 to 7.2 t DM ha^{-1} of compost. If mean soil temperatures increased by 2°C due to climate change, the amount of compost needed to maintain soil organic carbon levels would increase by ca. 10%.

In **Canada**, the application of compost (50 kg N ha^{-1}) over a three year period in snap beans resulted in the retention of 43% of added carbon. Tripling the application rate to 13 t C ha^{-1} yr^{-1}, reduced carbon retention to 27.7%. Elsewhere, composted corn silage (composted for nine months), dairy manure and sewage sludge were applied to a temperate grassland soil. In the first year after application, 95% (composted corn silage), 75% (dairy manure) and 88% (sewage sludge) of applied carbon remained in the soil. In the second year after application 89%, 63% and 42% of applied carbon remained in the soil for the respective treatments.

A nine-year trial in the **USA**, in which two composts, raw manure and mineral fertiliser, in combination with legume cover crops in all treatments, were used in a corn-vegetable-wheat rotation in a 2 ha field with silt loam, demonstrated that continuous compost use increased soil carbon levels by between 16% and 27%. Annual soil carbon sequestration over the nine-year trial period amounted to 1,261 kg C ha^{-1} for chicken litter and leaf compost and to 2,363 kg C ha^{-1} for dairy manure and leaf compost, while the use of fresh manure accrued only 312 kg C ha^{-1}. Mineral fertiliser reduced soil carbon levels by 317 kg C ha^{-1} yr^{-1}, although green manure crops were used also in this treatment.

The application of mature compost made from garden organics and chicken litter (~10% v/v) in vegetable cropping on sandy soil (0.6% C) in **Western Australia** showed that after the fifth compost application, i.e. around 18 months after compost was first applied, residual compost carbon levels amounted to 42.7%. The addition of clay (200 t ha^{-1}) to the sandy soil demonstrated the important role of clay minerals in preserving compost carbon, as 78.7% of compost carbon was still present in the clay amended sand 18 months after applications began.

The diversity of compost products, as well as the environmental and soil conditions in which they are used, combined with the complexity of related dynamics (turnover), has so far prevented development and adoption of a model that describes soil carbon dynamics following compost use. However, in the interim, a **simplified compost carbon sequestration model** can be used. The model assumes that the application of compost results in readily available organic matter being mineralised at y%. X% of the readily available organic carbon is converted to stable organic matter and z% of this stable organic matter is mineralised. The example in Figure 3 predicts the expected development of SOM levels associated with compost use at different application rates (0, 5, 10 and 15 t ha^{-1} yr^{-1}) in northern European conditions (over a 300-year period). This example uses the following mineralisation and transformation rates: x= 30%, y=15% and z=1% ($t_{1/2}$ of some 68 years). This model predicts that a new equilibrium for SOC will only be reached within a time frame of probably 200 and 300 years for annual compost application rates of 10 and 15 t ha^{-1} respectively. Annual application rates between 2.5–5.0 t ha^{-1} are expected to prevent further decline in soil carbon levels.

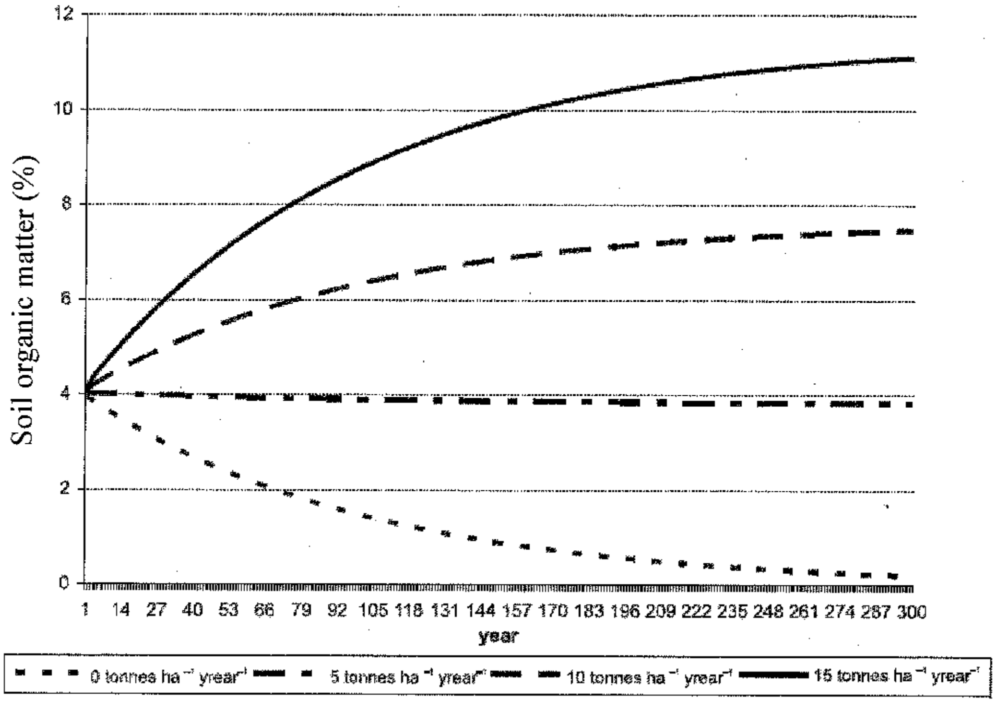

Figure 3 A simplified carbon sequestration model for compost use (Source: Favoino and Hogg 2008*).

In another simplified model using European data ('3-pool carbon sequestration model'), compost carbon sequestration was modelled and calculated over a 50-year time frame. It was calculated that, at $t_{\frac{1}{2}}$ = 20 years for the active soil organic matter pool, annual application of 22.8 t DM ha^{-1} biowaste compost (23% C) over 50 years results in the sequestration of 110 t C ha^{-1}, which represents 42% of carbon added with compost. Therefore, using compost at 22.8 t DM ha^{-1} for 50 years could achieve GHG emission abatements of 403 t CO$_2$-e ha^{-1}, 350 kg CO$_2$-e t^{-1} applied compost dry matter, and 1,537 kg CO$_2$-e t^{-1} applied compost carbon.

Based on a number of assumptions, the CENTURY model predicted the following carbon sequestration rates per tonne of dry matter compost in the USA:

- initial three years of compost application: 305 to 400 kg carbon, equivalent to between 1,122 and 1,467 kg CO$_2$-e
- initial ten years of compost application: 226 to 259 kg carbon, equivalent to between 829 to 950 kg CO$_2$-e
- after 34 years, with compost application for the initial 10 years only: between 85 and 155 kg carbon, equivalent to between 311 and 570 kg CO$_2$-e
- after 100 years: between 141 and 198kg carbon, equivalent to between 518 and 725 kg CO$_2$-e.

Using the same CENTURY model base data, a sequestration rate of 70 kg C t^{-1} fresh matter composted product (surface applied mulch and incorporated soil conditioner) was used in Australia. Considering different moisture contents, it is expected that 117 kg C (428 kg CO$_2$-e) is sequestered per dry tonne of applied soil conditioner over a 100-year time frame, and 88 to 93 kg C (321–342 kg CO$_2$-e) per dry tonne of applied mulch.

Not withstanding the significant short to medium-term carbon sequestration gains following compost use, the IPCC framework for estimating greenhouse gas fluxes is based on a 100-year time horizon and will only consider compost carbon as 'sequestered' if it remains locked in the soil for at least 100 years. As indicated previously, turnover times of different organic matter compounds can differ widely, and the proportion and estimated half-life times for different

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* Favoino, E. and D. Hogg (2008). "The potential role of compost in reducing greenhouse gases." Waste Management & Research **26**(1): 61 - 69.

compost carbon pools largely determine long-term carbon sequestration rates. In 2001, a major UK study adopted a compost carbon turnover time in soil of 40 years, and calculated the decay of the carbon added each year (a) according to the equation $y = a^{-kt}$, where y is the amount of carbon remaining after t years and k is the first-order decay rate constant, i.e. 1/turnover time.

With reference to manure-based data, the carbon still bound to soil after 100 years has been estimated to represent 2% to 10% of the input in compost, with recent suggestions that between 9% and 14% of compost carbon is sequestered, depending on soil type and crop rotation. As stable organic matter has a turnover time of 100 to 1000 years the proportion of compost-derived carbon that becomes part of the stable soil carbon fraction can be considered as 'sequestered' in the realm of the IPCC framework and for the purpose of international carbon trading, and be credited either as avoided downstream CO_2 emission to the waste management system, or as carbon sequestration gain to the land manager. Accordingly, if C_{input} (kg) is the carbon content in compost and C_{bind} is the fraction that is or will become 'stable', then the carbon sequestration, expressed as CO_2 ($CO_{2,bind}$, kg) can be calculated as:

$$CO_{2,bind} = C_{input} \times C_{bind} \times 44/12$$

If the carbon content in compost is in the order of 100–285 kg t^{-1} DM for garden organics compost, and 191–470 kg t^{-1} DM for food organics compost, and if between 2% and 14% of compost carbon are sequestered, between two and 66 kg C t^{-1} DM compost could be bound in soil, equivalent to 7–242 kg CO_2-e t^{-1} DM compost. Carbon sequestration is thus in the order of 2 to 79 kg CO_2-e t^{-1} for food organics and 3 to 73 kg CO_2-e t^{-1} for garden organics, assuming mass losses during the composting process of 60% and 30% respectively.

3.4.2 Mineralisation and use of nitrogen

The supply of mineral nitrogen through compost use can affect the level of nitrous oxide emissions from soil, the potential for nitrous oxide emissions due to nitrate being leached into groundwater, and emissions associated with fertiliser production when compost partially or totally substitutes for mineral nitrogen fertiliser. Apart from soil characteristics, soil moisture and temperature regimes, the supply of nitrogen from compost is primarily governed by its total nitrogen and carbon content and its C:N ratio, while compost maturity is of lesser importance.

When considering nitrogen mineralisation and nitrogen use efficiency data associated with compost use, it is helpful to note that:

- nitrogen availability measured with plants might be higher than in incubation tests without plants
- the presence of plants can modify nitrogen availability in soil
- nitrogen availability of commercial composts might be higher than for composts made on a smaller scale
- small differences in nitrogen release might not be detected when plants were used as indicators.

3.4.2.1 Nitrogen mineralisation

Although available publications often show similar nitrogen mineralisation effects and trends, results are difficult to compare because of divergent trial conditions, the high number of possible variables, or inadequate descriptions of trial conditions. In addition, laboratory incubation test results cannot be transferred directly to field conditions.

In any case, the following generic conclusions regarding nitrogen mineralisation following compost use can be drawn from the available information:

- use of immature compost often results in nitrogen immobilisation shortly after compost use
- total nitrogen mineralisation from immature compost is often lower than from mature compost
- composting of manure results in markedly reduced nitrogen mineralisation
- the average minimum and maximum mineralisation rates ranged between 4.6per cent and 19.5per cent of added total nitrogen in the most favourable temperature and moisture conditions
- mineralisation rates were higher in sandy soils than in clay soils.

Incubation of fresh and mature biowaste compost at temperatures between five and 25°C showed the influence temperature has on nitrogen mineralisation (Figure 4). Cumulative net release of NO_3 from fresh compost ranged from 1.8% of the initial nitrogen content at 5°C to 14.3% at 25°C during 112 days. However, net nitrogen release occurred only after a time lag of between 14 days at 25°C and 84 days at 5°C, when approximately 30% of compost carbon was mineralised. In contrast, in mature compost around 10% of the initial nitrogen content was released as NO_3 after 14 days, regardless of temperature, most likely representing the mineral nitrogen content in the mature compost. Cumulative net release of NO_3 from mature compost ranged between 9.8% at 5°C and 16% at 25°C.

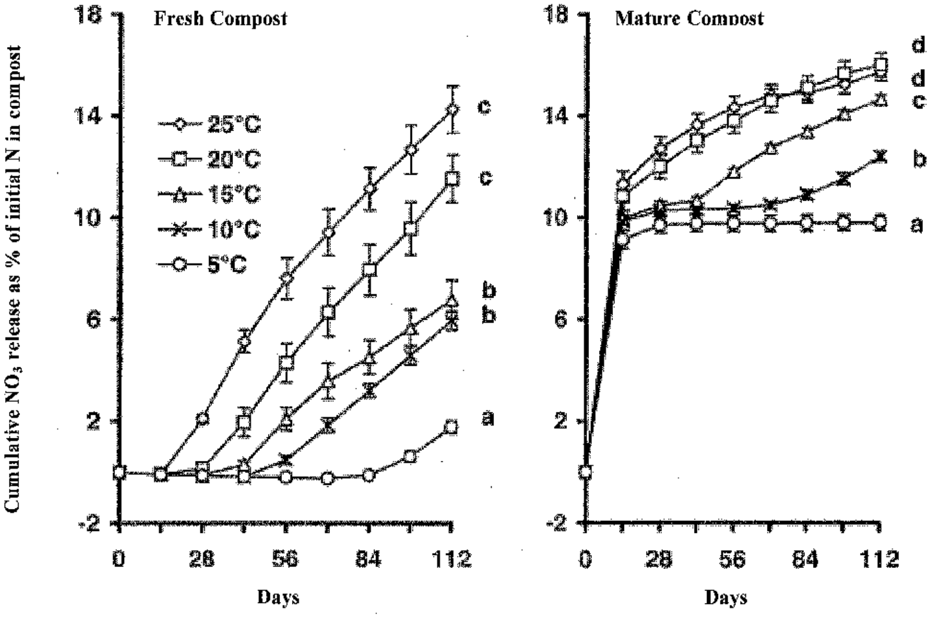

Figure 4 Cumulative release of NO_3-N from fresh (left) and mature (right) biowaste compost at various temperatures. Treatments with the same letter assigned are not significantly different ($p<0.05$) (Source: Chodak et al. 2001)*.

The report provides various examples where uncomposted material, along with composted products, resulted in nitrogen immobilisation, at least for the first few weeks following soil amendment. Although nitrogen immobilisation following the use of recycled organic products is usually an undesirable outcome, this effect can be employed as a means of 'mopping up' excess nitrogen from high nitrogen crop residues, and hence prevent nitrogen loss and nitrate leaching. Net nitrogen mineralisation from sugar beet and brassica leaves, for example, was reduced by up to 90% when soil was amended with paper waste that had a C:N ratio of 520:1. When leek residues were incubated with straw, garden organics composts (high and low C:N ratio) and tannic acid, all added materials (except for the compost with low C:N ratio) resulted in significant immobilisation of the residual nitrogen released from the vegetable residues. The addition of molasses after 15 weeks, which aimed at enhancing mineralisation, resulted in 73% of nitrogen that was initially immobilised by compost with high C:N ratio, to be remineralised.

3.4.2.2 Nitrogen use efficiency

Not all mineral nitrogen that becomes available in the soil from fertiliser, organic soil amendments or plant residues is absorbed and utilised by plants. Hence, not all compost-derived mineral nitrogen will be absorbed and utilised by plants, resulting in relatively low nitrogen use efficiency.

The following generic observations were made mainly in European climates about plant uptake of nitrogen derived from compost use:

- trial results must be differentiated between those that used mineral fertiliser as a supplementary source of nitrogen, and those that did not
- use of mineral nitrogen fertiliser usually reduces the efficiency of plants in utilising compost-derived nitrogen
- rotations that contain crops with high nitrogen demand can increase nitrogen use efficiency
- the longer the growth period of crops (e.g. corn versus wheat or leafy vegetables), the higher the nitrogen use efficiency
- nitrogen use efficiency in field trials ranged from 2.6% to 10.7% (average of minimum and maximum values respectively) during the first year after compost application
- subsequent (three or more years after compost application) nitrogen availability depends largely on site and production-specific conditions but is generally in the range of 2–3% of added compost nitrogen
- continuous compost use will increase nitrogen use efficiency in the initial year after compost application, with a maximum of 40% measured after 21 years of compost use
- mineralisation from mature composts usually occurs faster than from fresh composts, and fresh compost as well as those with high C/N ratio usually incur temporary nitrogen drawdown.

Results obtained during 9 and 12 year compost application trials at 5 sites in Germany confirm that, as an average across all sites, years and compost products used, only around 5% to 10% of nitrogen added with compost is utilised by crops. The long-term trials with three or four cropping cycles showed very clearly that nitrogen use efficiency increased from less than 5% during the first three-year crop cycle, to levels between 5% and 15% during subsequent cropping cycles. These medium-term trials confirmed the above generic observations that:

- the highest nitrogen use efficiency is achieved when no additional mineral nitrogen is used (plant uptake approximately 5% in the first crop cycle, and between 8% and 15% in subsequent crop cycles)
- increasing humus levels and microbial activity with continued compost use result in increased mineralisation and growing release and use of compost nitrogen
- a higher nitrogen use efficiency can be achieved when compost is used in crop rotations that have high nitrogen demand (average compost nitrogen use efficiency was 13–16% for high nitrogen demand rotations, and 7–10% for crop cycles with lower nitrogen demand, when no mineral fertiliser was used).

Nitrogen supply from occasional applications of moderate compost quantities often need not be accounted for in nitrogen budgets unless N immobilisation or optimum mineralisation conditions occur. When compost is used regularly for four or more years between 5–12% or possibly even 15% or more of nitrogen applied with compost, will be utilised by crops annually. Over a three-year crop cycle, approximately 20–35% of compost applied nitrogen will support plant growth. Such high nitrogen supply from compost use has to be accounted for in nitrogen budgets of individual fields and farms, so that monetary savings can be made, and agronomic and environmental problems prevented. Conversely, when garden organics composts and immature composts are used, nitrogen use efficiency should not be overestimated (about 5%). This will prevent unjustified fertiliser reduction that might adversely impact on crop yield or quality. In addition to estimating nitrogen release from compost, mineral soil nitrogen levels should also be assessed regularly.

A long-term (13 yrs) field trial in Austria showed relatively low nitrogen use efficiency. On average, approximately 6.5%, 3.7% and 3.4% of the total nitrogen applied through biowaste compost at annual application rates of 8, 15 and 22 t FM ha^{-1} respectively was used. Nitrogen use efficiency for compost remained unchanged (3–6%) when nitrogenous fertiliser (26–57 kg N ha^{-1}) was applied in addition to compost.

Trials on sands of low natural fertility in Australia confirmed that very little nitrogen initially supplied with compost (garden organics+10% chicken manure) was available to plants. However, at the conclusion of two trials, around 18% of nitrogen applied with compost was either utilised by crops or leached. However, application of unmatured compost for the seventh time resulted in the additional release of approximately 300 kg N ha^{-1}, presumably due to a priming effect that stimulated mineralisation of soil nitrogen reserves. This effect highlighted the potential risk of compost contributing to nitrate leaching, particularly on sandy soils, and in the early phases of compost use before the soil microbiology has had time to fully adapt to elevated compost inputs.

As only a relatively small proportion of the total nitrogen applied with compost is mineralised and used by plants over the first few years after application, continuous compost use leads to substantial increases in soil nitrogen levels. This has been confirmed by most compost related research, since, as soil organic matter has a C:N ratio of around 10:1, it is inevitable that increasing organic matter levels also increases soil nitrogen levels. If it is assumed that, on average, 10% of total nitrogen applied with compost is mineralised during the first year after application, 4% in the second year and 2% in the following years, it is possible to model both the potential soil nitrogen accumulation following annual compost application, as well as the increasing nitrogen mineralisation potential. If, for example, 8 t DM ha^{-1} of compost containing 1.6% N (in dry matter (DM)) is applied annually, and nitrogen losses are disregarded, annual nitrogen mineralisation from compost use (ca. 130 kg N ha^{-1}) will match the amount of total nitrogen added with compost after about 100 years (Figure 5). This equilibrium can be reached considerably faster in environments where higher mineralisation rates can be expected.

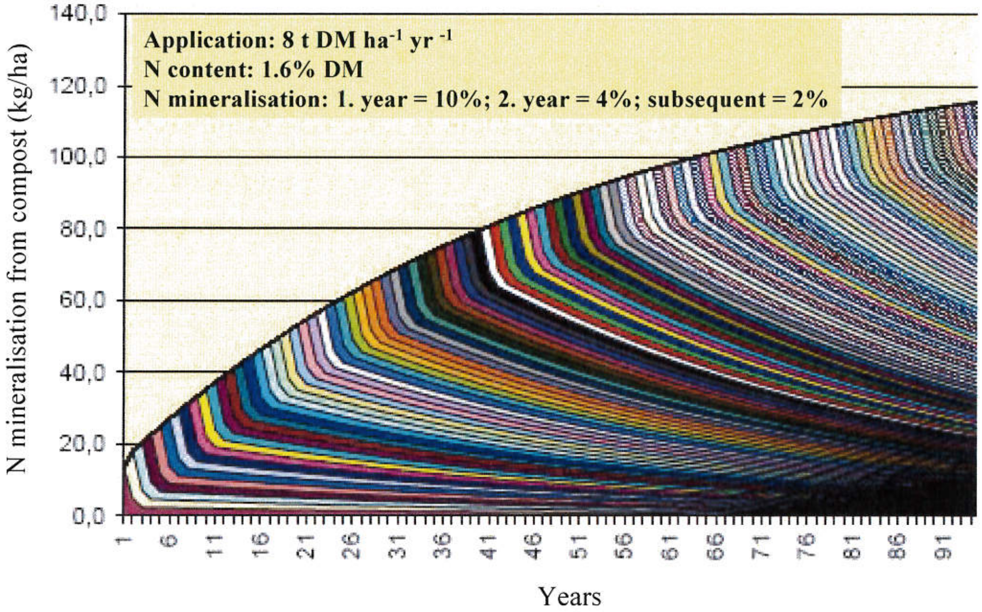

Figure 5 Expected increasing nitrogen mineralisation potential following long-term annual compost application at 8 t DM ha-1 (Source: Amlinger et al. 2003b*).

3.4.2.3 The risk of nitrate leaching from and for compost use

Untimely mineralisation and release of nitrogen is often raised as a potential problem associated with compost use, despite the fact that most nitrogen applied with compost is organically bound and only a small proportion is immediately available, or becomes available gradually after application. Concerns have also been expressed about nitrate leaching, which has been seen as a potential threat to groundwater quality. Quite a number of trials have shown that compost use does not pose a risk to groundwater quality through nitrate leaching, or reduces such risks compared to land application of uncomposted materials. However, there are also occasions where compost use on sandy soils can cause unexpected high release of nitrogen, probably due to the long-term build-up in soil nitrogen from previous compost and manure applications contributing to the surge of nitrates. However it is reported that if compost is applied regularly, it will take between 40 and 100 years before nitrogen supply through compost use will be balanced with plant nitrogen uptake. Problems may be caused if nitrogen supply through compost use is not accounted for in farm nutrient budgets, and that mineral fertiliser application rates are therefore not reduced, despite additional nitrogen inputs.

Many countries have regulations to protect the quality of ground and surface water, and to restrict the use of organic soil amendments, as well as overall nutrient applications. These regulations can cover the use of compost, even though it is only responsible for only a very small part of agricultural non-point source pollution of ground and surface water. The way in which the European Union's Nitrates Directive has been (and still is) implemented is presented in the report to demonstrate and exemplify potential implications for farmers, and for the use of organic soil amendments.

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* Amlinger, F., S. Peyer and P. Dreher (2003b). Kenntnisstand zur Frage des Stickstoffaustrags in Kompostduengungssystemen (Current Knowledge Regarding Leaching of Nitrogen in Agricultural Systems that Use Compost). E. a. W. M. Federal Ministry for Agriculture and Forestry, Austria. Vienna: 91.

3.4.3 GHG emission savings through fertiliser replacement

Compost use can supply at least some of the mineral nitrogen that would otherwise have to be provided through mineral fertilisers, as well as most, if not all, of the crop's phosphorous, potassium and trace element requirements. Substituting the use of mineral fertiliser through compost use offers the opportunity of reducing GHG emissions caused by the manufacturing and transportation of fertilisers. In order to estimate the potential for reducing GHG emissions, the typical nutrient contents of compost has to be established, as well as the level of GHG emissions that are associated with the manufacturing and transportation of various fertilisers.

Based on limited information available, it was estimated that 'typical' garden organics compost in Australia contains 11 kg N t^{-1} DM, 2 kg P t^{-1} DM, 5.5 kg K t^{-1} DM, and 275 kg C t^{-1} DM. GHG emissions associated with the production of mineral fertilisers vary greatly, but in considering dominant fertiliser use practices in Australia, GHG emission factors that are equivalent to 3,500 kg CO_2-e per tonne of nitrogen, 350 kg CO_2-e per tonne of phosphorous, and 300 kg CO_2-e per tonne of potassium were used for estimating potential GHG savings by replacing mineral fertilisers with nutrients supplied in compost in Australia. Consequently, if 10 t ha^{-1} DM of garden organics compost is used continuously and 40% of the applied nitrogen is utilised by crops (replacing the use of urea and single superphosphate) GHG emissions of approximately 180 kg CO_2-e can be avoided (Table 2). GHG savings are higher if compost with higher nutrient density is used, or if mineral fertilisers with higher carbon intensity are replaced. The 50% or less nitrogen use efficiency of mineral fertilisers, which requires the application of at least twice the amount of nitrogen that is replaced by compost use, has not been taken into account.

Table 2 Potential for avoiding GHG emissions associated with the production of mineral fertiliser if compost displaces their use

Nutrient element	Nutrient content (kg t^{-1} DM)	Crop uptake	Fertiliser nutrients replaced (kg t^{-1} DM)	Fertiliser GHG emissions (kg CO_2-e t^{-1})	Avoided GHG emissions (kg CO_2-e t^{-1} DM compost)
Garden organics compost					
Nitrogen (N)	11	40%	4.4	3,500	15.4
Phosphorous (P)	2	100%	2.0	350	0.7
Potassium (K)	5.5	100%	5.5	300	1.7
Total					**17.8**
Biowaste compost					
Nitrogen (N)	15	40%	6.0	3,500	21.0
Phosphorous (P)	3	100%	3.0	350	1.1
Potassium (K)	8	100%	8.0	300	2.4
Total					**24.5**

3.4.4 Nitrous oxide emissions following compost use

Although a considerable number of experiments sought to determine the effects of organic soil amendments use on (i) nitrous oxide emissions, (ii) the mechanisms that govern emissions and (iii) the magnitude of emissions following the use of different products, no conclusive and universally applicable results have yet emerged in this area. Although authors in the UK have put forward an emissions factor for garden organics compost of less than 0.04% of total nitrogen applied, and have calculated net reduction in nitrous oxide nitrogen emissions of 0.1 kg ha^{-1} (49 kg CO_2-e ha^{-1}) when applied at a rate that delivers 250 kg N ha^{-1}, most research conducted so far has delivered inconclusive and conflicting results. In any case, at this point in time it appears impossible to derive and use a generic nitrous oxide emission rate following the use of garden organics and biowaste compost.

3.4.5 Potential additional GHG savings following compost use

Apart from sequestering carbon, substituting the use of mineral fertiliser, and possibly reducing nitrous oxide emissions, the use of compost offers additional opportunities for reducing GHG emissions directly and indirectly. These include:

- reducing/avoiding the use of micronutrients
- reducing/avoiding the use of agricultural lime
- reducing/avoiding the use of gypsum
- avoiding the use of humic substances on their own as soil improvement treatment
- reducing need for irrigation and the associated use of fuel/electricity
- reducing the use of tractor fuel due to improved tilth
- reducing erosion and loss of soil and nutrients
- reducing the use of biocides
- increasing yields, and hence improving the efficiency per unit of input.

3.4.6 Combined effects of using compost on climate change mitigation

Both the medium (20 and 50 yrs) and the long-term (100 yrs) time frames need to be considered in regards to the sequestration of compost-derived carbon. From a global warming potential (GWP) point of view, it makes relatively little difference if compost use is considered under a 20 or 100-year time frame, as:

- GWP of CO_2 (i.e. degradation of biogenic carbon in soil) equals one across all time horizons
- GWP of methane (72 over 20 years and 25 over 100 years) is not relevant as methane is not emitted from soils, rather the opposite
- GWP of N_2O differs very little between the 20 year time frame (289) and the 100 year time frame (298).

Interim climate change mitigation measures, such as carbon sequestration following compost use, are of particular importance. They provide opportunities to implement low-cost measures that are available immediately and deliver a wide range of other environmental, agronomic and societal benefits.

Consequently, it was assumed that 45% of carbon applied with compost is retained over a 20-year period, 35% over a 50-year period and 10% over a 100-year period. Hence, the use of mature garden organics compost as agricultural soil conditioner at a rate of **10 t DM ha^{-1}** will sequester carbon that is equivalent to reducing GHG emissions by:

- **5,046 kg CO_2-e over 20 years**
- **3,532 kg CO_2-e over 50 years**
- **1,009 kg CO_2-e over 100 years.**

The use of **10 t DM ha^{-1}** of mature garden organics compost as agricultural soil conditioner is expected to replace the use of mineral fertiliser (N, P, K), which saves **178 kg CO_2-e** of GHG emissions.

By combining the two GHG emission savings options, mature garden organics compost is used at **10 t DM ha^{-1}** as agricultural soil conditioner:

- **around 5,224 kg CO_2-e can be saved in GHG emissions within a 20-year time frame**
- **around 3,710 kg CO_2-e can be saved in GHG emissions within a 50-year time frame**
- **around 1,187 kg CO_2-e can be saved in GHG emissions within a 100-year time frame.**

An economic compost evaluation model, which included potential GHG emission savings through continuous use of biowaste compost demonstrated that GHG emission savings increase from year to year as compost use continues, reaching a level of 474 kg CO_2-e per hectare on an arable farm after seven years, at an application rate of 10 t DM ha^{-1}. This equated to a monetary value of $18.06* per hectare in the seventh year, based on marginal costs for avoiding GHG emissions in Germany (without carbon trading), which at the time were €24.00 or $38.09* per tonne CO_2-e.

Although increasing compost application rates result in increased GHG emission savings, the increase in GHG emission savings was not proportional to the increase in application rates. This resulted in declining marginal monetary benefits of saved GHG emissions per tonne of compost:

5 t DM ha^{-1}	€1.23 or $1.95* t^{-1} DM compost
10 t DM ha^{-1}	€1.14 or $1.81* t^{-1} DM compost
20 t DM ha^{-1}	€0.90 or $1.43* t^{-1} DM compost.

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*Converted at exchange rate as of 1.7.2005: $1.00 = €0.63 (see http://www.rba.gov.au/statistics/hist-exchange-rates/2003-2006.xls)

4 Findings, conclusions and recommendations for further work

4.1 Findings

This report, which has assessed whether and to what extent the use of compost can assist in mitigating climate change, has clearly demonstrated that using compost saves GHG emissions by the following direct and indirect means:

- reducing methane emissions from landfill
- reducing GHG emissions through improved manure management
- sequestering biogenic, compost derived carbon in the soil
- replacing the use of mineral fertilisers
- reducing methane emissions from soil, or increasing soil methane absorption
- reducing nitrous oxide emissions from soil
- improving plant biomass production, resulting in increased sequestration of plant carbon
- supplying auxiliary GHG emission savings (e.g. reduced need for irrigation, reduced erosion, reduced liming, reduced nitrate leaching).

Consequently, using compost is an important tool in reducing both waste management and agricultural GHG emissions, and in helping to mitigate climate change.

More precisely, the following key findings were made:

- **Reduction of methane from landfill**
  The Australian organics recycling industry diverted at least 3.7 Mt of organic residues from landfill in 2007–08, preventing the generation of methane that is equivalent to ~4.28 Mt CO_2-e. The saving in methane when organic materials are recycled is nearly equivalent to the amount of methane recovered from landfill (4.5 Mt CO_2-e).
- **Reduction of methane emissions through improved manure management**
  Composting improves manure management practices, as it reduces methane emissions compared to other manure management options (stockpiling, lagoons), and the use of composted manures has higher climate change mitigation effects than the use of raw manures.
- **Sequestration of compost derived carbon**
  Based mainly on European and North American data, it is estimated that the use of garden and biowaste compost as soil conditioner will result in the following carbon sequestration rates:

○ 20-year accounting time frame:	40–55% of carbon applied with compost
○ 50-year accounting time frame:	30–50% of carbon applied with compost
○ 100-year accounting time frame:	0–14% of carbon applied with compost.

Using characteristics of Australian garden organics composts, it is assumed that 45% of carbon applied with compost will be retained over a 20-year period, 35% over a 50-year period, and 10% over a 100-year period. The use of mature garden organics compost as agricultural soil conditioner at a rate of 10 t DM ha^{-1} will sequester carbon that is equivalent to reducing GHG emissions by 5,046 kg CO_2-e over 20 years; 3,532 kg CO_2-e over 50 years; or 1,009 kg CO_2-e over 100 years.

Reduction of mineral fertiliser use

Again, based on the characteristics of Australian garden organics composts it is estimated that the use of mature compost as agricultural soil conditioner at a rate of 10 t DM ha^{-1} will replace the use of mineral fertiliser (N, P, K) to the extent that saves 178 kg CO_2-e of GHG emissions.

Combined GHG emission savings

When mature garden organics compost is used at 10 t DM ha^{-1} as agricultural soil conditioner, the following GHG emissions can be saved:

- ~ 5,224 kg CO_2-e within a 20-year accounting time frame
- ~ 3,710 kg CO_2-e within a 50-year accounting time frame
- ~ 1,187 kg CO_2-e within a 100-year accounting time frame.

National Carbon Accounting System

The National Carbon Accounting System (NCAS) has been developed by the Australian government to report on land-based GHG emissions and sinks. The NCAS reports on all carbon pools (biomass, dead organic matter and soil carbon) and all principal greenhouse gases by making use of integrated ecosystem modelling combined with remotely sensed data. The terrestrial ecosystem model (Full Carbon Accounting Model) is based on 25 m grid resolution data and incorporates various sub-models, one of which is Roth-C soil carbon model. The NCAS provides an ideal framework for integrating the use of compost and other organic soil amendments into the National Carbon Inventory. Including the use of compost into the National Carbon Inventory will reduce Australia's net GHG emissions.

4.2 Conclusions

Using compost as an agricultural and horticultural soil amendment:

- can contribute to mitigating climate change directly and indirectly
- provides opportunities for implementing low-cost measures that are immediately available
- is one of the fastest means of improving soil carbon levels
- is ideally suited as a mitigation measure in productive agricultural soils
- fits easily into the Australian NCAS
- can attract carbon credits
- delivers many agronomic benefits and enhances long-term agricultural production
- also delivers environmental and societal benefits.

4.3 Further work

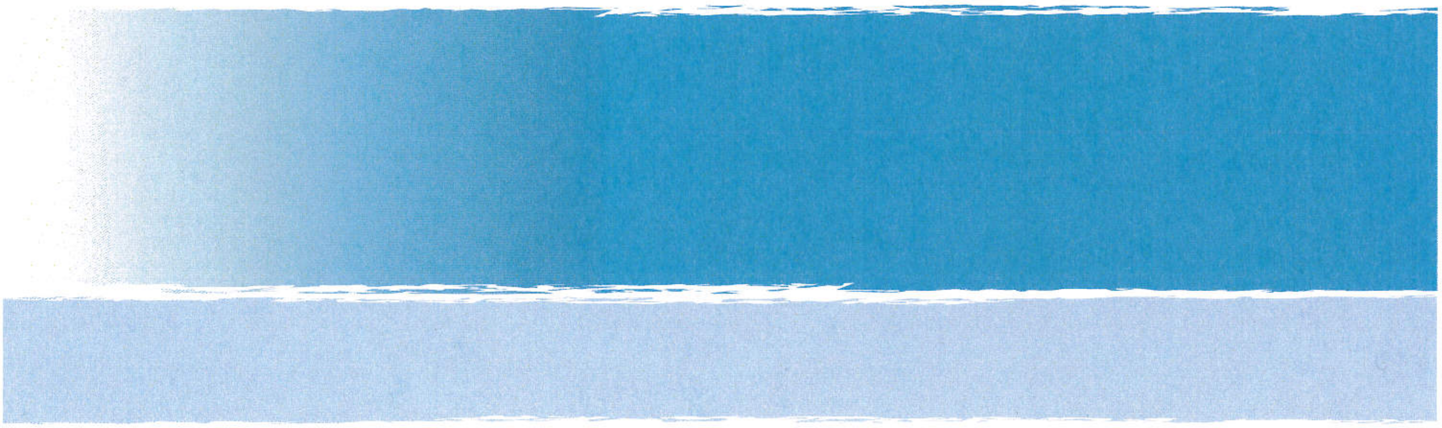
Based on the findings of this report, the following work should be carried out to maximise the use of compost with the associated GHG savings, and also to provide farmers with a tool for gauging how management practices impact on soil humus levels:

- adjust available data from Europe and North America to reflect Australian environmental and farming conditions
- incorporate compost use into the Australian National Carbon Accounting System
- introduce the concept and use of 'Humus Balance'.

Additional recommendations for future work are documented in the main report.

4.4 References

For a complete list of references please see the full report *The benefits of using compost for mitigating climate change,* available at the Department of Environment, Climate Change and Water NSW website.



Life Cycle Inventory and Life Cycle Assessment for Windrow Composting Systems

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- Dr Sven Lundie and Mr Paul Beavis, Centre for Waste and Water Technology, UNSW.
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The Recycled Organics Unit also thanks Dr Frank Valzano for his contribution to the construction of the LCIA calculator as a tool (Excel 2000) for modelling environmental impacts using life cycle inventory data established for windrow composting systems.

Life Cycle Inventory and Life Cycle Assessment of Windrow Composting Systems

The NSW Department of Environment and Conservation has undertaken a study on the life cycle assessment (LCA) of windrow composting systems to inform the management of organic materials diverted from landfill in NSW.

The Department's Sustainability Program Division delivers programs, which support the achievement of waste reduction and resource recovery outcomes identified in the *NSW Waste Avoidance and Resource Recovery Strategy.*

LCA is a tool that can assist in evaluating the environmental impacts of different resource recovery options. It measures the environmental impact of a product or process by assessing individual impacts at each individual stage of its life cycle. For the recycling of municipally generated organic materials, this typically involves source separation by the householder, collection of organics from the kerbside, transport of material to a composting facility and processing. The composted products are sold to a range of markets for use in improving soils, plant growth and the environment.

One of the barriers preventing an assessment of the full environmental benefit of composting in NSW has been a lack of life cycle inventory data, that is, measures of impacts and benefits at each stage of the life cycle described above. In particular, data has been lacking on the actual benefits composted products provide to soils and plants. It is known that the use of such products can reduce the need for fertilisers, watering and herbicides, and can improve plant growth, but these benefits have been difficult to quantify / calculate. Previous LCA studies have therefore underestimated the full environmental benefit of composting, because the 'post application benefits' have not been fully accounted for.

This is the first study internationally that has attempted to fully assess the environmental benefit of source separated collection, composting and application of the resulting recycled product to the environment. The study clearly demonstrates that this approach offers substantial benefits to the environment. Much of this benefit is achieved when quality composted products are applied to soils to enhance soil health, fertility and productivity.

The study confirms the environmental benefits associated with recycling of organic materials diverted from landfill, being an important service provided to the community by numerous local councils across NSW. The report also provides valuable support data for local councils considering and assessing options for improving the recovery of organics from landfill in NSW.

Executive summary

Abstract

Numerous local government, industry and state government agencies have expressed a need for Life Cycle Assessment to inform environmental decision making in relation to the streaming and management of solid waste. However, comparative studies to date have not in any significant manner addressed the impacts resulting from the use of recycled organics products such as composts once such products have been applied. As a result, previous comparative studies have tended to exclude such post application impacts from the analysis, effectively excluding a range of significant environmental benefits and reducing the relative environmental value of biological treatment systems.

This study provides a comprehensive Life Cycle Inventory (LCI) for commercial composting systems in Australia, and models the environmental impacts of the commercial composting systems in NSW using rigorous Life Cycle Assessment (LCA) modelling. This study is the first time, internationally, that LCI data for the post-application impacts of composting systems has been developed in any significant or comprehensive manner. The study has been extensively reviewed by relevant technical experts in Life Cycle Assessment, and by relevant agricultural and environmental scientists in NSW Agriculture and the CSIRO.

The results of this study indicate significant environmental benefits arise from the commercial composting system, including net greenhouse benefits, even where composts are transported significant distances (in this study 600 km) for agricultural application.

Whilst this LCA study is valuable in its own right in identifying and quantifying the environmental externalities of the composting system, the LCI data in this study is also transparently developed and documented to allow for application in future comparative LCA studies.

Background and scope of the study

The *Waste Avoidance and Resource Recovery Act* 2001 requires consideration of life cycle assessment as a tool to assist in the assessment of new technologies and to inform government policy and strategy.

As a result, the Department of Environment and Conservation (DEC) has commissioned the Recycled Organics Unit of UNSW to conduct a study on Life Cycle Assessment for Windrow Composting Systems, given the importance that large scale composting plays in recovering a significant proportion of organic material diverted from the waste stream in NSW.

Life cycle assessment (LCA) provides a structured framework and methods for analysing and assessing the environmental issues and the potential impacts (both beneficial and detrimental) of a material, product or service over the entire period of its life cycle. Life cycle inventory analysis (LCI) is a technical process that identifies and quantifies energy and resource consumption, and environmental releases to air, water and land throughout the life cycle of a product or system (AS/NZS ISO 14041, 1999). Life cycle impact assessment (LCIA) characterises and assesses the affects of resource consumption and environmental loadings identified in the inventory stage on environmental impacts.

The Goal of this LCA study is to quantify and evaluate the total environmental impacts of windrow composting systems, including the production of mature compost products, transport, and application of composted products to agriculture as a option for managing compostable organics materials generated in NSW. The boundaries of windrow composting system considered in this study include receival of municipally collected source separated garden organics materials, processing of these materials, transportation and subsequent agricultural application of composted products (Figure 1).

Importance of assessing post-application environmental benefits

Significant environmental impacts result from the use of recycled organics products such as composts once such products have been applied. Such post application impacts have been developed in this study for two categories of compost product, and two agricultural contexts. The study has focused on composted mulches and composted soil conditioners that comply with the specification documented in Australian Standard AS 4454 (2003). In the broadacre agriculture context, the study has modelled the application of relevant application rates of composted soil conditioner to irrigated cotton production. In the intensive agriculture context, the study has modelled the application of relevant application rates of composted mulch to irrigated grapevine production. The environmental impact categories associated with windrow composting systems that have been considered in this study include energy consumption, greenhouse effect, photochemical oxidant formation, human and eco-toxicity, resource depletion, eutrophication, and land use. This has enabled a full environmental assessment of the windrow composting system, which has not previously been possible.

The LCI data for the production of quality compost products is derived from literature, site visits and interviews. The LCI data for the post application of quality compost products is derived from extensive review of scientific literature.

The post application LCI data has been developed from a comprehensive literature review and analysis for agricultural production systems using a novel line of best fit method. This method analysed the results of compost applications of varying rates to establish the relationships between compost application rates and environmental impacts arising from the application of composted products in agriculture. This report contains the supporting technical data from published journal papers, reports, and books.

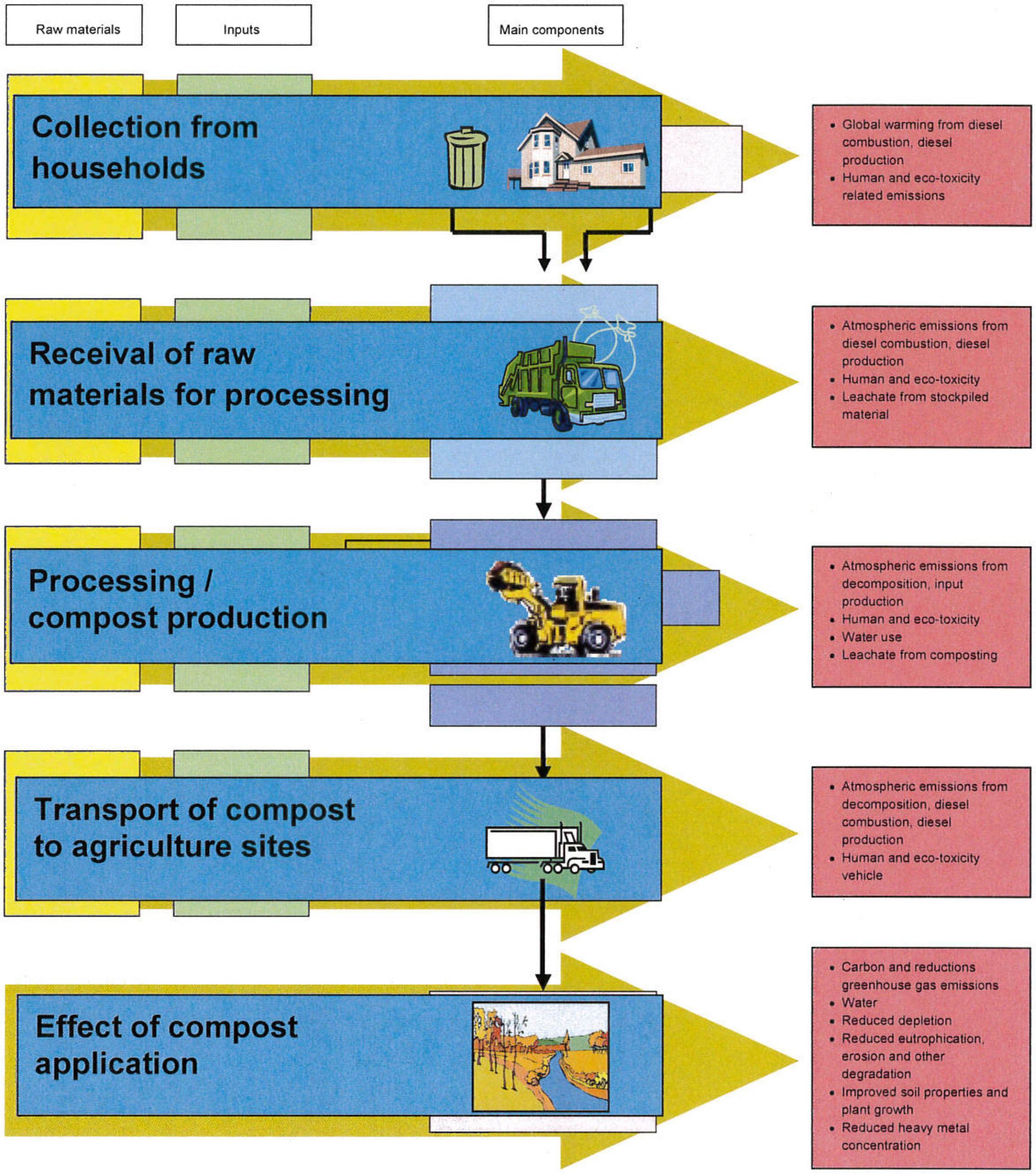
The LCI data was then used to model, via LCIA, a range of application rates and defined scenarios to establish environmental impacts associated with the commercial composting system. The application of composted products has been modelled in 7 modelling scenarios, and results reported in the context of these impact categories. The major variables considered in the LCIA modelling per application area included:

- compost product type;
- application rate;
- transport distance to application sites; and
- the backloading or empty return (non-backloading) of vehicles transporting composted products to application sites.

System boundaries

The system boundary for the life cycle assessment of the windrow composting system included the production of composted products, transport, application to agricultural production systems and post-application impacts. The system model is graphically represented in Figure 1.

Figure 1. Graphical overview of the system model used in this study. The system model is explained in more detail in Section 4.2 of this report. Note: Dotted lines represent the system boundaries of this LCA study.



Summary results

a) Life cycle inventory

Life cycle inventory of windrow composting systems, including post application impacts has identified relevant environmental issues (impact categories), and quantified data for production and application of composted products. Life cycle inventory data suggests that most environmental issues arising from production and transportation to application sites and application on agricultural land are related to production and consumption of diesel fuel and electricity required during all stages of these processes (Tables 1 and 2).

Table 1 Fuel, electricity and water consumption during a composting operation.

Operation	Consumption (per tonne of feedstock)		
	Diesel (litres)	Electricity (kW)	Water (litres)
Receival	0.48	0	0
Composting process (including shredding, windrow formation and screening)	5.02 (3.65+0.58+0.79)	0.13	0
Dispatch	0.03	0	0
Total	5.53	0.13	

Table 2 Fuel consumption during transportation and spreading of composted products at application sites.

Operation	Product type	Distance travelled (km)	Backloading/non-backloading	Fuel consumption per tonne of composted product (litres)
Transport	Composted mulch	175	Backloading	0.74
Transport	Composted mulch	275	Backloading	1.15
Handling and spreading	Composted mulch	Not applicable	Not applicable	0.66
Transport	Composted soil conditioner	600	Backloading	2.47
Handling and spreading	Composted soil conditioner	Not applicable	Not applicable	1.06

In contrast, LCI data relating to post application of compost relate to a wider range of issues including: reduction in use of irrigation water, electricity, fertilisers, and herbicides for agriculture (Table 3). In addition, LCI data relating to the post application impacts has identified a reduction and/or improvement in processes responsible for land degradation, including:

- reduced soil erosion;
- improved carbon sequestration in the soil;
- improved soil structure; and
- the potential to reclaim saline-sodic soils (Table 3).

Note that land degradation has been identified the most severe environmental problem in NSW State of Environment Report (NSW EPA, 1997).

Table 3 Summary of post-application impacts resulting from application of composted soil conditioner (25-50 t ha^{-1}) to cotton and composted mulch (10 cm deep) to grapevine. (Note: Results in this table refer to a single application).

Impact	**Tangible (potential) benefits**	**Tangible (potential) benefits**
	Composted soil conditioner to cotton	Composted mulch to grapevine
Water use (Section 7.2.8)	• Increasing water holding capacity of top 0-15 cm soil layer by 2.4-3.0 %. • Savings of 0.13-0.16 ML of water per hectare per season in irrigated cotton • Savings of 37,006-45,370 ML of water to cotton industry per season of irrigated cotton.	• Increasing moisture retention of top 0- 15 cm soil layer by 9.82%. • Savings of 0.95 ML of water per hectare per season in irrigated viticulture, if it is available to plants. • Savings of 23,750 ML of water per season of irrigated viticulture.
Fertiliser use (Section 7.3.4)	• Savings of 34-68 kg of N, 29-57 kg of P, and 24-48 kg of K per hectare during the first year of application • Savings of 90-180 kg of N, 75-150 kg of P, and 30-60 kg of K per hectare during 3-5 years.	• Savings of 27-40 kg of N, 46-68 kg of P, and 72-108 kg of K per hectare during the first year of application • Savings of 72-108 kg of N, 120-180 kg of P, and 90-135 kg of K per hectare during 3-5 years after application.
Herbicide use (Section 7.4.4)	• Soil conditioners do not suppress weeds.	• Replacing the total herbicide requirement in a vineyard. i.e 2-6 L of glyphosate per hectare.
Biocide use (Section 7.5.3)	• Potential to control Fusarium wilt in cotton, which is spreading at epidemic levels in NSW. There is no known biocide that can be replaced with compost.	• No data available
Carbon sequestration (Section 7.6.2)	• Sequestering about 2.9-5.9 tons of carbon per hectare after 10 years. This figure may be taken for carbon accounting process. • Sequestering about 1.8-3.5 tons of carbon per hectare after 100 years. This figure is relevant to LCI studies.	• Sequestering about 11.56 tons of carbon per hectare after 10 years. This figure may be taken for carbon accounting process. • Sequestering about 7.056 tons of carbon per hectare after 100 years. This figure is relevant to LCI studies.
Sodicity (Section 7.7.3)	• Potential savings of 2-5 tonnes of gypsum per hectare in affected lands. No such data available yet.	• No data available.
Erosion (Section 7.8.3)	• Preventing a soil loss of 2.3 to 4.2 tonnes per hectare annually.	• Preventing a soil loss of 17.5 tonnes per hectare annually.
Soil structure (Bulk density) Section 7.9.3	• Bulk density decreased by 4.1 to 7.6%	• No data available.
Yield response (Section 7.9.1)	• Increasing the cotton yield by 19.5 to 21.5%	• Increasing the grape yield by 27%.

b) Life cycle impact assessment

Life cycle impact assessment of environmental issues arising from production and application of composted products (Table 4) has shown a net environmental benefit for impact categories including global warming potential (GWP), human toxicity potential (HTP), ecotoxicity potential (ETP) (terrestrial and freshwater) and eutrophication potential (EP) (in case of soil conditioners only). The beneficial results are largely due to the reduction in use of fertilisers, herbicides, water, and electricity resulting from compost applications, and therefore reducing release of GHGs, nutrients and toxic chemicals to environment (air, water, and soil) during production and use of these avoided inputs. These beneficial impacts offset the GHGs, nutrients and toxic substances released into environment during production and use of diesel and electricity required for production and application of composted products.

Life cycle impact assessment however has shown either a negligible or a small net detrimental environmental impact (Table 4) from the composting system for impact categories such as ecotoxicity potential (ETP) (marine water), photochemical oxidation potential (POCP) and abiotic resource depletion potential (ARDP). The oxides of nitrogen, sulphur, and carbon (that produce photochemical oxidants) released during production and consumption of diesel fuel and electricity used in compost production and transport are currently calculated to be greater than the reduction in release of these oxides during production and use of fertilisers, herbicides, and electricity resulting from post compost application to agriculture. This study could not however include a number of reductions in agricultural inputs (such as fertilisers and water) and improvements in soil conditions due to absence of characterisation/equivalency factors required for LCIA, and the absence of a range of data, which would be expected to result in improved environmental performance of the composting system in these detrimental impact categories. Note characterisation/equivalency factors are derived from characterisation model relevant to environmental mechanism of each impact category, which are applied to convert the assigned LCI results to the common unit of the category indicator (AS/NZS ISO 14042:2001).

Composted mulches cause detrimental environmental impact with respect to EP (Table 4). Mulches, being lower nutrient products compared to soil conditioners, often require normal quantities of fertiliser to support plant growth. Thus, the impacts arising from the use of fertiliser are not offset from the use of mulch. This highlights the importance of considering nutrient budgeting in specifying suitable composts for agriculture. Where compost products are selected that provide maximum nutrient value to a crop (over the life of the application), this will result in maximum environmental benefit arising from reduced fertiliser use. This principle applies also to avoided irrigation, herbicide and biocide applications. It should be pointed out that eutrophication potential from diesel production will be mainly confined to coastal water resources and not to inland water resources (river systems) because refineries that produce diesel are generally located in coastal areas associated with significant port cities.

The results of the LCIA with respect to ETP have shown beneficial environmental impacts for terrestrial and freshwater ecosystems but a negligible detrimental environmental impact for ETP (marine water) (Table 4), suggesting that ecotoxicity, like eutrophication, will be mainly confined to coastal water resources and not to inland water resources (river systems) because production of diesel fuel occurs near coastal areas releasing toxic chemicals in marine water thereby causing a minor net detrimental environmental impact with respect to ETP (marine water).

Table 4 Summary of LCIA results for production and application of total composted products on 100 ha of land (▤-Beneficial impact; ×-Detrimental impact; **Note**- General comment).

Impact Categories	**Environmental impacts**	
Global warming potential (t CO_2 eq)	▤	Beneficial environmental impact for composted mulches and soil conditioners.
	Note	Application rates of composted products had greater influence on GWP than transport distance, and whether or not backloading was practiced.
Eutrophication potential (t O_2 eq)	▤	Beneficial environmental impact for soil conditioners.
	×	Detrimental environmental impact for composted mulches. Note this impact will be mainly confined to coastal water resources and not to inland river systems.
	Note	Application rates of composted products had greater influence on EP than varying transport distance, and whether or not backloading was practiced.
Human toxicity potential (t DCB eq)	▤	Beneficial environmental impact for composted mulches and soil conditioners.
	Note	Application rates of composted products had greater influence on HTP than transport distance, and whether or not backloading was practiced.
Ecotoxity potential- (t DCB eq)	▤	Beneficial environmental impact for composted mulches and soil conditioners with respect to ETP for terrestrial and freshwater environment.
	×	Negligible detrimental environmental impact for composted mulches and soil conditioners with respect to ETP (marine water) compared to beneficial environmental impact of ETP for terrestrial and freshwater ecosystems.
	Note	Application rates of composted products had greater influence on ETP than variation in transport distance, and whether or not backloading was practiced.
Photochemical oxidation potential (t ethylene eq)	×	Small detrimental environmental impact for composted mulches and soil conditioners.
	Note	Application rates of composted products had greater influence on HTP than transport distance, and whether or not backloading was practiced.
Abiotic Resource depletion potential (t antimony eq)	×	Small detrimental environmental impact for composted mulches and soil conditioners. Note a number of reductions in agricultural inputs and improvements in land conditions could not be included for this category in LCIA due to absence of a range of data.
	Note	Application rates of composted products had greater influence on ARDP than transport distance, and whether or not backloading was practiced.
Land use	**Note**	Data not available
Avoided loads		
Irrigation water (ML)	▤	Net environmental benefit from composted mulches and soil conditioners.
	Note	Mulches produced greater benefit than soil conditioners.
Fertilisers (tonnes)	▤	Net environmental benefit from composted mulches and soil conditioners.
	Note	Soil conditioners produced greater benefit than mulches.
Herbicides/pesticides savings (litres)	▤	Net environmental benefit from composted mulches.
	Note	Soil conditioners do not produce this benefit.
Reduced erosion (tonnes)	▤	Net environmental benefit from composted mulches and soil conditioners.
	Note	Mulches produced greater benefit than soil conditioners.
Carbon sequestration (tonnes)	▤	Net environmental benefit from composted mulches and soil conditioners.
	Note	Mulches produced greater benefit than soil conditioners.
	Note	Soil conditioners produce this benefit faster than mulches.
Soil structure improvement (% decrease in bulk density)	▤	Net environmental benefit from composted mulches and soil conditioners.
	Note	Mulches produced greater benefit than soil conditioners.
	Note	Soil conditioners produce this benefit faster than mulches.
Increased plant productivity (% increase in yield)	▤	Net environmental benefit from composted mulches and soil conditioners.

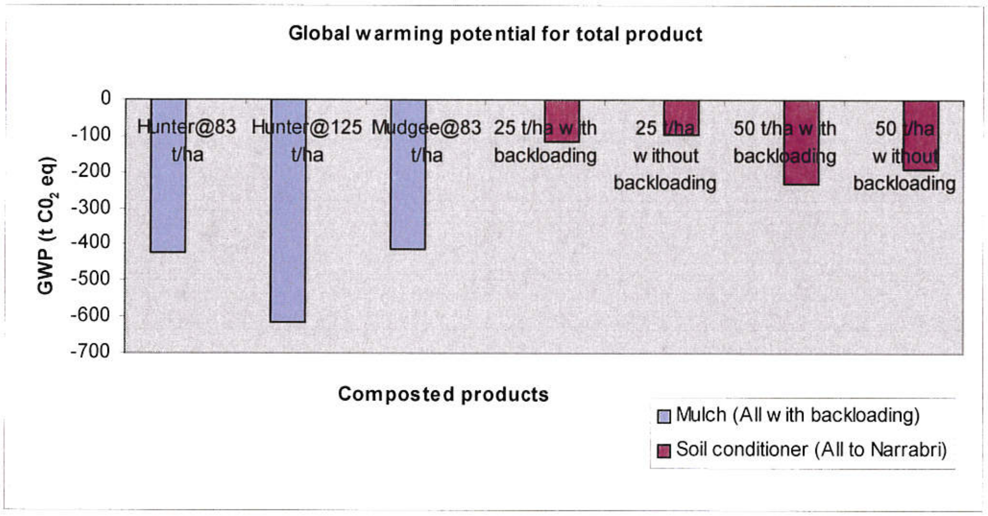
Higher application rates of composted products produce greater net environmental benefits (Figure 2), as higher application rates reduce the need for other farm inputs. Higher application rates per unit area produce higher post application benefit, and greater net environmental benefit.

However, it should be mentioned that post application benefits do not increase linearly with increasing application rates. At certain application rates, most post application benefits will tend to maximise. Excessive application rates may cause detrimental affects suggesting a limit for net environmental benefits. Therefore, to achieve maximum environmental benefit, composts should be applied at a rate so as to maximise:

- Weed suppression,
- Soil moisture available for plant use,
- Manufactured to suppress crop/region specific pest and diseases, and
- Selected to maximise contribution to crop nutrient requirements when applied at the appropriate application rate.

Varying the transport distance by 100 km produced negligible impact on all impact categories, in particular GWP (Figure 2). However, there will be a limit for transport distance difference beyond which impact from transport difference outweighs benefits of compost applications. Note, however, that transport of soil conditioner to the most distant area of the state still resulted in a net environmental benefit including with regard to GWP.

Figure 2. Global warming potential for production and application of total composted product on 100 ha of agricultural land (Note: Negative numbers represent a net environmental benefit. More negative results represent a greater environmental benefit).



Backloading or without backloading scenarios hardly have any impact on all impact categories particularly GWP (Figure 2). The reason for this is that greenhouse gases released during backloading journeys have not been accounted in this system because these are not considered part of this system. However, greenhouse gases released during non-backloading journeys have been included as a part of this system.

Limitations and gaps of this study

The main limitation of the LCA in this study is that LCIA does not present total environmental impact assessment as it could not include a number of reductions in agricultural inputs (such as fertilisers and water) and improvements in soil conditions due to absence of characterisation/equivalency factors required for LCIA. This includes an absence of LCI data, which would be expected to result in improved environmental performance of the composting system for a number of impact categories.

The main limitation of post application impacts is that the majority of data for these impacts have been obtained from overseas studies. Also, performance does not necessarily reflect the level of performance that could be realized from appropriate application rate of products that are specifically tailored to the specific agricultural application context. Though relationships and trends identified for these impacts in this study will most likely be similar for Australian agriculture, quantitative impacts may not be entirely appropriate for Australian conditions. Australian agriculture (particularly soil and climate conditions) is different from where (Europe and North America) most of the data in this study has been obtained.

The establishment of performance data from applications relevant to the needs of the crop, soil and region, and applied at a rate to achieve maximum benefit will markedly improve the quality of LCI data, and provide a more accurate indication of the associated post-application environmental impact.

The gaps in LCI data for post application impacts and LCIA identified in this study are listed below:

- Life cycle inventory data in this study could not be quantified for Australian gypsum production; biocide used in cotton and grapevine cultivation in NSW that can be replaced to control diseases suppressed by compost application; and gypsum that can be replaced with composted soil conditioners to ameliorate sodic soils.

In the case of composted mulches, post application performance benefit relationships have been derived for mulch application depth without considering mulch application width because benefits from mulch application in literature are cited mainly for application depth. Though mulch application depth is responsible for producing post application benefits, nevertheless changing mulch application width also changes mulch application rate, which is likely to change post application benefits. This study also could not identify the impacts of composted mulch application on soil bulk density and sodicity.

The relationships could not be determined from existing LCI data between post application benefits to plant available water in the case of composted mulches and to ecological integrity and biodiversity for composted products.

- In relation to LCIA, characterisation/equivalency factors were not available to assess ARDP in relation to fertilisers, water, and land use.

Options for further work

Basic recommendations of this study are:

- It is recommended that an economic assessment should be conducted to calculate dollar values associated with identified environmental benefits to inform and justify future policies and strategies for organics recovery, management, and agricultural compost use.
- That this report to be distributed to ensure that LCI data and LCA results is available for future studies, and for related policy and strategy development across Australia.
- That the results and identified gaps of this study should be integrated into other NSW Department of Environment and Conservation projects, including field research trials to address gaps identified in LCI data for post application impacts of composted products and life cycle impact assessment.
- That future LCA studies should be commissioned in a manner such that LCI data is transparently documented and is made available in a form suitable for use in future LCA studies.
- Maximising beneficial environmental impact of composts to a particular crop across the life of a compost application are significant in terms of maximising the nutritional, irrigation efficiency and disease/weed suppression value. It is therefore recommended that soil testing, nutrient budgeting and other related crop requirements be integrated into the development, specification and selection of compost products for crop specific applications.
- That future compost production application trials should assess the performance of products that are specifically designed to best meet the crop and environment performance requirements of the particular crop and growing region.
- That the NSW EPA Alternative Waste Technology Selector and Handbook should be updated to incorporate data and information arising from this study.

Section 1 About this report

1.1 Background and objectives

With the current national environmental focus on greenhouse gas issues, a tendency has developed to compare technologies on this basis, and provide financial incentives for technologies that claim superior greenhouse performance.

However, a policy based on a single environmental impact is flawed, as there are many other environmental impacts, which could be more detrimental to the environment. It can be argued that the current implementation of greenhouse gas programs and incentives is distorting priorities and compromising ecologically sustainable development (ESD) in the resource recovery sector.

Two primary principles of ESD are the 'conservation of biological diversity and ecological integrity', and 'intergenerational equity'. Soil and water resources are finite, and need to be protected for future generations. Currently the majority of compostable organic materials generated in urban areas in Australia are sent to landfill disposal sites. Such practices diminish soil and water resources, and are inconsistent with the principles of ESD. The *Waste Avoidance and Resource Recovery* Act (2001) defines an explicit objective of encouraging the most efficient use of resources and to reduce environmental harm in accordance with the principles of ESD.

Composting of organic materials (e.g. food organics and garden organics) is one method of managing these valuable materials, while at the same time adhering to the ideals of ESD. However, the impacts (beneficial and detrimental) of composting need to be evaluated and compared with alternative methods of organics management (e.g. waste to energy, landfill disposal, etc.) in order to understand and compare the total environmental risks and impacts of these different management systems.

At the *Recycled Organics: Expanding the Horizons* conference in Queensland (November, 1999), participants (particularly those from NSW state government agencies and local government) expressed a strong desire for Life Cycle Assessment (LCA) modelling to be conducted to enable informed comparison of the environmental impacts of various organics management systems. Participants felt a strong need for objective informing upon which priorities could be argued, established and defended.

Life cycle assessment is a valuable tool for the assessment of environmental impacts, giving quantitative and qualitative information on resource consumption and environmental emissions of the system investigated. Life cycle assessment provides a basis for making informed decisions with regard to resource recovery priorities for the management of compostable organic materials. In the context of commercial composting, meaningful cost-benefit analysis on a triple bottom line basis cannot be provided until such time as comprehensive and relevant LCA studies have been conducted.

In response to expressed need, in 2000, the Recycled Organics Unit commissioned an LCA of organic waste management systems (Shrestha, 2001). This study found that meaningful and relevant Life Cycle Inventory (LCI) data for composting as a system had not been developed and was therefore not available to inform environmental impact modelling. The conclusions drawn from this study are that previous LCA studies involving composting systems should be considered inadequate, and that the development of relevant and meaningful LCI data must occur prior to the conduct of future LCA studies in this area.

Further in the past, LCA studies comparing composting systems with alternative waste management technologies have not considered the post-application effects of adding compost to soil. Although some studies (e.g. US EPA, 1998; NSW Agriculture, 2002) have attempted to quantify carbon sequestration effects, other beneficial effects of applying compost to soil have not been addressed due to the complexity of these post-application effects. However, exclusion of total environmental impact criteria from LCA devalues the key environmental benefits resulting from compost application, and unfairly disadvantages composting systems in comparison to alternative technologies.

In the recent review of the *Waste Minimisation and Management* Act 1995 (NSW EPA, 2001), one of the responsibilities identified for NSW Department of Environment and Conservation is *developing a way to assess new waste technologies.*

In the subsequent presentation of the *Waste Avoidance and Resource Recovery Bill 2001* to the NSW Legislative Assembly, the current NSW Minister for the Environment, the Honourable Mr Debus, stated (in relation to the identification of resource recovery options) that:

> "*...priority will depend on analytical tools, including cost-benefit and life cycle analysis*" (Hon. R. Debus, 2001)

To overcome identified obstacles and to implement the explicit NSW government objective of establishing waste management priorities, this study aims to develop valid LCI data, and to conduct a LCA study to inform government strategy for resource recovery.

Clearly there is now recognition that LCA has an important contribution to make to informed decision making in terms of government strategy and priorities. LCA may also be fundamental to developing a consistent whole of government approach to environment management and ESD.

The summary objectives of this study are therefore to

- Develop an LCA model for windrow composting systems to provide outcomes relevant to decision making across NSW.
- To develop a comprehensive Life Cycle Inventory (LCI) for licensed windrow composting systems in NSW, including the post-application impacts resulting from agricultural application of recycled organics products, specifically *composted mulch* and *composted soil conditioner.*
- Conduct a transparent Life Cycle Assessment (LCA) of windrow composting systems for specified application scenarios.

1.2 Who is the report for?

This report is intended for following groups:

- NSW government agencies responsible for the development of resource recovery policy and strategy, primarily Department of Environment and Conservation NSW, NSW Department of Primary Industries and Department of Energy, Utilities and Sustainability;
- Other government agencies responsible for natural resources management;
- Local government;
- Commercial composting industry, who stand to benefit from the quantification of the environmental benefits of their operations; and
- Related researchers in areas of organics management, sustainable agriculture, natural resource management and life cycle assessment.

1.3 Scope of the report

Life cycle inventory data for post-application scenario has been obtained from the studies that have used a range of organic soil amendments and mulches including composted products, biosolids, manures, straw and other agricultural residues. All these materials are a source of organic carbon/organic matter and when added to agriculture land can produce similar long term impact on agricultural production systems as would composted products manufactured as per AS 4454 (2003).

This report covers the review of national and international studies, which always may not be directly relevant to NSW soils and environmental conditions, but which provide overall insight into a range of impacts of land application of organic materials and products.

1.4 How to use the report

This report documents LCI data for windrow composting systems, including use of composted products in agriculture. It analyses the various aspects of environmental impacts arising from the production, transportation, application and post-application in step-by-step procedure.

This report provides quantified environmental impacts associated with the production, transportation, and application including post-application of specific composted products to specific agricultural scenarios using LCA methodology.

This report is complemented by a Life Cycle Impact Assessment (LCIA) calculator (MS Excel) that can be applied to modelling of composts of varying characteristics and different application scenarios.

The quantified environment benefits can be used to; establish associated dollar values related to the environmental impacts of the system for the purpose of comparative analysis of different waste management/resource recovery systems; and to inform future industry and government policy and strategy.

Appendices and a glossary annexed at the end of the report provide an overview of LCA methodology and terms to further enhance comprehension of the report.

1.5 Terminology

Terms used throughout this report have been officially adopted by NSW Department of Environment and Conservation in July 2000 as documented in the Recycled Organics *Dictionary and Thesaurus: Standard terminology for the recycled organics industry* (Recycled Organics Unit, 2002b). This document is freely downloadable from http://www.rolibrary.com.

Life cycle assessment terminology used in this report is nationally accepted and derived from the relevant AS/NZS ISO 14040 series of international standards defining LCA methodology. The definitions of LCA terminology are given in the Glossary of LCA Terms (Appendix 3).

A range of recycled organic products are described in this report including as composted soil conditioner and composted mulch, these products are as defined in the relevant Australian Standard AS 4454 (Standards Australia, 2003).

Section 2 Introduction

2.1 Land degradation in NSW

Land degradation is one of the important environmental impacts associated with agriculture. Significant reduction of soil organic matter (SOM) in most Australian soils due to continuous cultivation is a major contributor in land degradation. This low level of organic matter is often the limiting factor affecting soil fertility, productivity and stability. A level of 2% organic matter is considered desirable for maintaining good soil structure for agricultural activities (Greenland, 1971), but many NSW surface soils have organic carbon levels less than 1% (Spain et al., 1983).

Total arable land suitable for crops, sown pastures and grasses was about 104,241 km^2 or 13% of total NSW land area (ABS, 1996). This represents a comparatively small and non-renewable resource base compared to European countries.

The 1997 NSW State of the Environment report has specifically identified land degradation as the most serious land use issue affecting the state of the environment in NSW (NSW EPA, 1997). More than 70% of the state is affected by at least one form of land degradation. Of this area, 29% is severely to very severely affected by land degradation (NSW EPA, 1997). Note, the 2000 State of the Environment report (NSW EPA, 2000) did not provide any such explicit statement on the significance of land degradation in NSW.

Land degradation was responsible for at least $700 million a year in lost production in NSW (NSW Agriculture, 1998). It is widely recognised that if steps are not taken to reverse the degradation of these finite soil reserves, the long-term viability of many primary production industries will be severely threatened, as will many aesthetic and biodiversity values in the state (NSW EPA, 1997).

In this context, application of composted products to agricultural lands can make a beneficial contribution to address a range of land degradation problems. It can help to reverse the rapid rate of soil organic matter and nutrient decline across the state by providing organic carbon and nutrients to the soil, which is essential to redress these critical deficiencies. Soil erosion (various forms), structural decline, nutrient decline, salinity, sodicity and acidity are the main categories of land degradation, which may be addressed by the application of composted products.

Addition of compost increases SOM and electrolyte concentration, which improves soil aggregation, stability of soil aggregates, reduces physical disintegration of soil aggregates by raindrop impact and chemical dispersion (Agassi *et al.,* 1990). Due to the stability of soil aggregates, which improves soil structure, the application of compost prevents surface sealing, improves water infiltration and the water holding capacity thus reducing runoff generation and soil erodibility (Bresson *et al.,* 2001; Albaladejo *et al.,* 2000; Stocking and Albaladejo, 1994).

An increase in soil organic matter content also reduces the effect of soil sodicity by removing excess sodium due to complexing sodium (Na) with soluble organic compounds and by decreasing precipitation of calcium (Ca) resulting in a decrease in exchangeable sodium percentage (Kaur *et al.,* 2002; Slatterly *et al.,* 2002; Wahid *et al.,* 1998; Churchman *et al.*, 1993; Sekhon and Bajwa, 1993). Application of organic matter also enhances migration of Na to lower soil layers due to the increase in porosity resulting from improved water stability of aggregates (Sekhon and Bajwa, 1993; Wahid *et al.*, 1998). Therefore, organic matter application effectively modifies the effect of exchangeable sodium in soils.

The use of compost has also been shown to ameliorate soil acidity due to its proton consumption capacity and ability to complex aluminium (Al) ions (Mokolobate and Haynes, 2002; Van den Berghe and Hue, 1999; Pocknee and Sumner, 1997).

Other positive effects of using composted products on agricultural soils include improvement in biological activity (Broken *et al.,* 2002), cation exchange capacity (Shiralipour *et al.,* 1992), macro and micronutrients availability (Movahedi and Cook, 2000), and the suppression of soil borne diseases (Alvarez *et al.,* 1995;Tilston *et al.,* 2002).

The negative effects of using some composted products on agricultural soils can include an increase in heavy metal contents and contamination with organic compounds such as polychlorate biphenyl, phenols, aromatics and organic acids (Gallardo-Lara and Nogales, 1987). However, this report addresses only composts that comply with the requirements of the relevant Australian Standard AS 4454 *Composts, soil conditioners and mulches* (Standards Australia, 2003) and are considered suitable for unrestricted application (NSW EPA, 1997a), therefore heavy metals and other chemical compounds referred to here are not considered relevant to this study.

2.2 Windrow composting systems

Turned windrow composting systems are the dominant form of composting in Australia, particularly for *garden organics* material. However, at least eight different forms of composting systems are available for processing a wide range of compostable organic materials. Nevertheless, when only the composting process itself is considered, most systems are variations of a common theme (Recycled Organics Unit, 2002a).

The composting process is a complex process of biological transformation governed by the activity of naturally occurring microorganisms, and involving highly variable heterogenic substrates (materials). In the process, organic materials provide the substrate to produce fully mineralised products such as carbon dioxide (CO_2), water (H_2O), ammonium (NH_4^+), and stabilized compost products (organic matter dominated by humic substances) that are heavily populated with competitive microbial biomass and ash (Chefetz *et al.,* 1996). Hence, the aim of all composting systems is to efficiently manage this process of biological transformation by manipulating temperature, oxygen and moisture of the composting mass through a variety of mechanisms.

Windrow composting is a process for *aerobic* (aerated) and *thermophilic* (hot) biological transformation of organic materials. The process generates heat that destroys pathogens and produces stabilized compost products for use as a mulch, soil conditioner and topsoil additive. It involves the formation of horizontally extended piles of organic materials by a front-end loader, and the periodic turning of the piles by loader or purpose built windrow turner to mix materials and reduce the density of the composting mass, thereby allowing increased passive air movement (via convection and gaseous diffusion) into the composting mass. Extended piles are generally 1.5 to 3 m in height, 3 to 6 m wide, and up to any length, length being limited by the scale of composting facility (Recycled Organics Unit, 2002a). The mass of organic material (windrow) is then monitored, and managed only by watering and periodic mechanical turning for aeration. This method is simple, non-intensive, has relatively low capital cost, and is commonly used by farmers, municipalities, and waste processing centres in Australia and around the world. The generic stages in a windrow composting operation are identified in Figure 2.1.

Plate 2.1 Garden organics (a) can be processed in windrows (b) to make composted products.

(a)

(b)

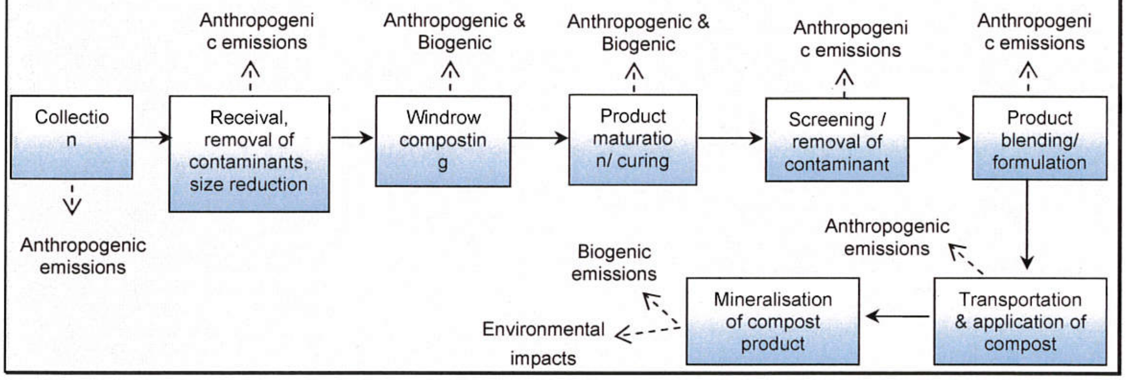
Plate 2.2 Clean garden organics are unloaded (a) and shredded by a tub grinder (b) before composting.

(a)

(b)

Windrow composting can be used to process residual garden organics, food organics (food waste), paper and cardboard, agricultural residuals and biosolids (sewage sludge). The process of composting begins with collecting, receiving, sorting, and storing compostable organic materials (raw materials). These steps are then followed by size reduction, mixing and windrow formation. The compostable organic materials must be screened or hand picked for non-biodegradable materials (contaminants), and then chipped, ground, or shredded into acceptable particle size to increase the rate of decomposition. The high-carbon, dry wood and paper materials are usually mixed in equal proportion with high-nitrogen, high-moisture grass clippings and food materials to provide balanced nutrition for the organisms of decomposition. Materials are mixed using a front-end loader, or paddle-blade mixer to distribute the carbon and nitrogen rich materials evenly throughout the composting mass. The material is then formed into windrows to decompose.

Figure 2.1 Windrow based aerobic composting flow chart (modified from Recycled Organics Unit, 2001).



Note: Dotted lines indicate a by-product of the main process

Windrows can be placed directly on soil or paved area, according to requirements of the applicable licensing regime. The land requirement for a windrow composting facility depends on the volume of material processed. Generally, all the materials handling and windrow formation can be accomplished with a front-end (wheel) loader. Purpose built windrow turners can provide additional size reduction function and operational efficiencies.

Plate 2.3 Making (a) and turning (b) of windrows by specialized purpose built windrow turners.


(a)

(b)

The microbial decomposition of organic waste is controlled by environmental factors affecting microbial activity within the windrow piles. Aeration and moisture are two very important factors influencing the microbial activity. Temperature and aeration control is managed by physically turning the composting mass by either a front-end loader or specialised windrow turner.

Oxygen levels and temperature are maintained within a specific range to provide optimum conditions for the microorganisms that are responsible for decomposition. The temperature must be high enough (50-55^{0}C) to kill pathogens and weed seeds, but not so high as to kill the beneficial organisms. The decomposition process produces heat, and the insulation properties of the organic material results in heat being generated at a greater rate than the rate of heat loss from the windrow. Well-aerated and properly mixed compost piles formed from an appropriate mixture of organic materials do not produce unpleasant odours.

As the microorganisms start decomposing the organic fraction, they generate heat resulting in a rise in temperature of the pile. The temperature in the middle of a windrow composting system can reach as high as 70°C depending on the size of the pile or system, its moisture content, aeration and availability of substrate for microbes. If the temperature reaches to 65–70°C, the microorganisms die off and the decomposition slows down. Eventually the temperature of the pile drops due to slowing of the composting process when microorganisms use (consume) up most of the readily decomposable materials ("food"). The highest rate of decomposition occurs at temperatures between 50–55°C. Therefore, aeration (by turning) is important to maintain the temperature around this range to maximise the rate of decomposition and to ensure pasteurisation (killing of weed seeds and pathogens). Oxygen level of 10–14%, moisture content of 50–60% and carbon to nitrogen (C:N) ratio of the composting mixture within an acceptable range are the other requirements to maintain a windrow pile under optimal composting conditions (Recycled Organics Unit, 2002a).

Windrows are aerated by natural or passive air movement (convection and gaseous diffusion) between turnings. The rate of air exchange depends on the porosity of the windrow. Therefore, the size of a windrow that can be effectively aerated is determined by its porosity. If a windrow is too large, anaerobic zones may form near its centre, which release odours when the windrow is turned (Rynk, 1992).

Turning procedures used in mechanically turned windrows help to:

- Distribute materials more evenly throughout the composting mass,
- Mix materials,
- Rebuild the porosity of the windrow,
- Release trapped heat, water vapour and gases, and
- Replenish oxygen levels.

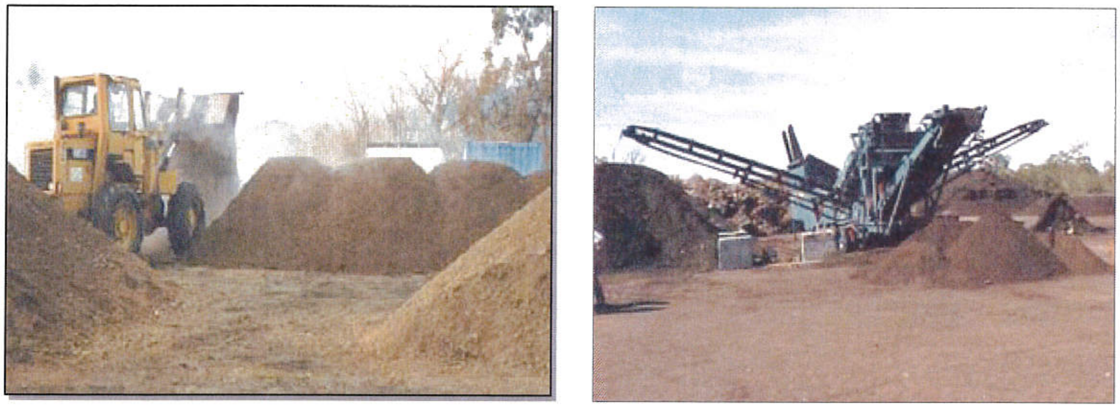
Equipment used in windrow composting operations may include front-end loaders and/or windrow turning machines, shredding and screening equipment, and bunded pads for windrow placement. Each of these mechanical components of the system consumes fuel and contributes to the greenhouse gas (GHG) emissions from a facility, either during their construction and/or while they are operated in the management of the composting facility. The equipment used in the turning process, area available and the nature of the compostable organic material being processed determines the size, shape and spacing of windrows (Jackson and Line, 1998; Rynk, 1992).

Following the composting period, the windrows are broken down and reconstructed into curing piles for additional aging and drying of the material as required. Curing compost improves stability and prevents odours or other nuisances developing while the material is stored. The curing phase requires significantly less management than the active composting phase. Minimal odour generation occurs during this phase.

After curing, the compost can be screened to a range of specific particle size gradings (soil conditioner, fine mulch, coarse mulch) and/or blended with a range of amendments to produce value added products such as potting mixes, depending on the requirement of the relevant market. The scope of this study does not extend to the blending of value added products, this study addresses the production of composted soil conditioners and mulches, and therefore only the process of screening for particle size grading is included in relation to value adding processes.

Plate 5.3 Front-end loader used for turning windrows (a) and screening of end product (b).

(a) (b)



2.2.1 Land application of composted materials

The windrow composting system does not end with the production of the compost, but continues with the application of composted material to the soil, and the ongoing impacts that result from such application. Compost has many beneficial post-application impacts, which are discussed in detail in Section 7. These benefits primarily arise from the addition of organic matter to the soil and the reduced need for soil chemical and other additives.

Application rates of compost vary, but in general more than 10 tonnes per ha of composted soil conditioner is required to have some influence on soil properties.

Plate 6.2 Tractor drawn manure spreader used for the application of compost.

2.3 Composition of composted products in NSW

The composition of composted products varies with the composition of the raw materials from which composts are produced. Since a variety of organic materials can be used to make compost (residual garden organics, food organics, biosolids, wood and timber, paper products, agricultural organics, etc.), knowledge of the physical and chemical characteristics of the specific compost products, and their respective application rates is critical to the study their post-application impact on environment.

This study identifies the impacts/avoided impacts of the agricultural application of the following composted products:

1. *Composted mulch*: a relatively low nutrient composted mulch compliant with specification in Australian Standard AS 4454 (Standards Australia, 2003), and produced from 100% garden organics.
2. *Composted soil conditioner*: a relatively high nutrient composted soil conditioner compliant with specification in Australian Standard AS 4454 (Standards Australia, 2003), and produced from either:
   - A mixture of garden organics and food organics; or
   - A mixture of garden organics and biosolids.

Table 2.1 shows the physical and chemical characteristics of the composted products considered in this study. The composition of composted products for the purpose of this study is based on commercially available mature compost products that are produced at facilities with strict quality assurance procedures (Table 2.1), and that comply with Australian Standard AS 4454 (Standards Australia, 2003).

Table 2.1 Physical and chemical properties of composted products.

Product types	Composted soil conditioner	Composted mulch
Particle grading (mm)	0 – 15	15 – 35
pH	6.5 – 8.0	5.5 – 6.5
EC (dS m^{-1})	1.0 – 3.5	1.2 – 2.0
Bulk density (kg/m^3)	950	480 – 540
Moisture content (%)	40	20 – 25
Water holding capacity (%)	50 – 60	10 – 20
Organic matter (%dry matter)	55 – 75	75 – 95
Total N (% dry matter)	1.0 – 2.0	0.2 – 0.4
Total P (%dry matter)	0.1 – 0.9	0.1 – 0.3
Total K (%dry matter)	0.1 – 0.3	0.1 – 0.2
Total Ca (%dry matter)	0.3 – 0.9	0.1 – 0.4
Total Mg (%dry matter)	0.08 – 0.1	0.08 – 0.1

Source: Australian Native Landscapes Pty. Ltd. (NSW), and Natural Recovery Systems (VIC).

2.4 Compost application scenarios

Vegetable, fruit, meat, grain, oil and fibre products exported from the rural sector for consumption in urban and metropolitan areas are produced through the management of finite soils reserves, with the injection of fertiliser, water and other inputs. Return of these organic resources, particularly nutrients and organic matter, back to agricultural soils is vital if the very serious issue of widespread land degradation and losses in farm productivity are to be reversed in NSW.

Recycled organic products categories such as surface mulches and soil conditioners have been identified having high demand potential in agriculture (NSW Waste Boards, 1999). Utilization of available residual organic materials in agriculture therefore provides an option to reduce fertilizer use, promote soil remediation, improve ecological integrity and biological diversity, and avoid pollution. In this context, composting processes should be recognised for their potential to convert organic materials into compost products that can make a significant contribution to resolving some of the problems facing agricultural and environmental management sectors in NSW.

The land degradation issues such as soil erosion (various forms), structural decline, nutrient decline and salinity and sodicity are more pronounced along the Murray-Darling river basin of NSW due to the continuous cultivation over a long period. This study therefore recognises the Murray-Darling river basin of NSW as a potential beneficiary of the application of compost products generated in urban areas of NSW.

The current situation (such as impacts of agricultural production systems on soils and landscape, availability of crop residues, land area under cultivation, the value of the crop, and gross margins) and potential of compost application in major agricultural production systems in NSW has been analysed to prioritise agricultural production systems for this study (Table 2.2).

This analysis has identified that the application of composted soil conditioner in cotton production, and the application of composted mulches in grape production systems as the most relevant context for this study. This analysis has been based on the criteria provided below:

- Affordability (i.e. cost of production, crop value per hectare, gross margin);
- Area under cultivation (particularly irrigated cultivation);
- Soil remediation value to the farming system (i.e. status of soil, irrigation inefficiency and related soil problems);
- Logistics of transport (back loading potential) and low risk associated with potential transport cross contamination; and
- Availability of relevant research findings.

The main criteria for selecting cotton and grape production systems were high value crops, high gross margin, limited crop residues and variation in transport distances. In addition, previous studies have identified viticulture as having the greatest market potential for RO products in intensive agriculture (NSW Waste Boards, 1999). In relation to broadacre agriculture, the high gross margin, high level of production inputs, limited relative availability of crop residues, and variation in transport distance (relative to viticulture) supports the selection of cotton production for the purpose of this study. This study therefore will focus on two agricultural production systems and two product categories as provided below. Many of the findings from this study can be extended to other agricultural systems (i.e. cereals, oil crops, etc.).

1. The soil incorporation of 12 t ha^{-1} per year of *composted soil conditioner* to broadacre agriculture cotton production systems.

2. The surface application of a 10 cm deep and about 50 cm wide (equivalent to about 75 t ha^{-1}) *composted mulch* approximately every 3 years to intensive agriculture grapevine production systems.

These application rates have been selected after direct consultation with key researchers and composting enterprises directly involved in land application of compost products in agriculture, and on the basis of the review of literature in subsequent sections of this report. At this point, it is neither possible, nor critical that the application rate be absolutely correct, simply that it be within a reasonable and viable range as the resulting LCA modelling will allow for variation in application rate.

Table 2.2 Summary of application scenarios considered for this study, as relevant to criteria for prioritising application (Source: Farm Enterprise Budget (2001/2002), NSW Agriculture and ABS, 2001).

Agricultural Practice	Current situation	Viability of RO product application
Cereals	- Significant impacts on soils and landscape - Availability of crop residuals from stubble retention - Low value crop with relatively low input system - Large area under cultivation - Area (NSW) - 4,543,000 ha - Gross margin - 100 – 800 $/ha	Unlikely
Oil crops and legumes	- Mostly included in crop rotations with cereals - Availability of crop residuals from previous crops - Low value crop with low input - Area (NSW) - 773,000 ha - Gross margin - 100 – 600 $/ha	Unlikely
Fibre crops (cotton)	- High impact on soil and landscape - Depreciation of soil organic mater at high rates - Low irrigation efficiency with salinity and sodicity trends - High value crop under irrigated conditions - Rising world prices - High possibility of backloading - Less cross contamination during backloading - Cotton products are exposed to further processing - Area (NSW) - 268,000 ha (irrigated), 66,000 (rainfed) - Gross Margin - 700 – 1,250 $/ha under irrigation 100 – 350 $/ha in dryland production	Possible
Pasture & grasses	- Generally used as a part of any crop rotation - Usually have significant levels of organic matter if managed - Low value crop - Area (NSW) - 6,400,000 ha - Gross margin – 100 – 350 $/ha	Unlikely
Sugarcane	- High impact on soils and landscape - Availability of organic matter through green cane - Harvesting and thrash mulching - Low value crop - Area (NSW) - 20,000 ha - Gross margin - 150 - 350 $/ha	Unlikely
Grapes	- Prone to soil erosion - Chemical weed control and irrigation inefficiencies - High possibility of backloading - High value crop - Area (NSW) – 32,269 ha - Gross margin – 6,000 – 14,000 $/ha	Possible

2.4.1 Cotton production

The cotton growing areas of NSW are scattered between the Queensland border and Lachlan Valley. Most cotton is irrigated (80% of the total area under cotton). Little or no crop residue is returned to the soils under irrigated cotton, though some measures have been recently taken in this regard.

Cotton has poor tolerance of waterlogging, therefore cotton soil needs to have good porosity for infiltration and internal drainage to allow adequate water entry, and to encourage root exploration by quickly re-establishing aeration after irrigation and rainfall (NSW Agriculture, 1998). The alluvial soil types, black earths and the better structured grey and brown clays with their extensive cracking enabling vigorous root growth and provide favourable conditions (NSW Agriculture, 1998).

Most cotton growing areas in Australia are dominated by clay soil (black earths and grey and brown clays). Cracking clay soils are resilient and regenerate their structure by shrinking and swelling: a phenomenon called "self mulching" due to their high smectite content, while alluvial soils maintain the soil structure due to their high organic matter content (NSW Agriculture, 1998). In the Macquarie Valley, and to a lesser extent in the Namoi and Gwydir Valleys, cotton is grown on red brown earth soils (NSW Agriculture, 1998). Red brown earths are difficult to repair due to structural decline and limited inherent capacity to regenerate and these soils commonly develop hard setting surfaces and saline and sodic conditions.

This study focuses its findings on the application of composted soil conditioner to red brown earth soils under irrigated cotton cultivation for the development of relevant life cycle inventory data. It provides a very useful case study context that is relevant to the objectives of this study, to the development of agricultural markets for compost, and is widely relevant to the broader Australian agricultural context in terms of the application of composted soil conditioner to broad acre cropping scenarios. The findings of the study therefore may be readily extended to other soil types under cotton cultivation, since problems such as structural decline, soil compaction, salinity and sodicity are common to all cotton growing soils to a certain extent, and beyond that to other broadacre cropping systems in Australia, particularly in the Murray-Darling basin.

2.4.2 Grape production

This study also considers the application of composted mulch on vineyards on the basis of affordability (Table 2.1), soil remediation value, logistics of transport (back loading potential), and availability of relevant research findings.

Grapes are grown throughout the state of NSW from Moree to Dayton, Tenterfield to Tumbarumba, and Port Macquarie to Mudgee (Anthony Somers, District Horticulturist, NSW Agriculture, Tocal). Total area occupied by viticulture in NSW was 34,559 hectares that included grapes planted after the 2000 harvest and were not fruit bearing at the time of survey (ABS, 2001).

Grapevines perform best in terms of fruit, productivity and ease of management on red soils (Browne, 1994). The ideal soil is red loam to clay loam, over well-structured red clay, but any well-drained subsoil is suitable (NSW Agriculture, 1995).

Numerous recent studies have identified viticulture production in the intensive agriculture sector as having high demand potential for application of composted mulches (Recycled Organics Unit, 2002c), and increasing use of composts in the viticulture sector is currently a key project area for NSW Department of Environment and Conservation.

Section 3 Life cycle assessment (LCA)

3.1 LCA background

Life cycle assessment (LCA) provides a framework and methods for analysing and assessing the environmental aspects and the potential impact of a material, product or service over the entire period of its life cycle. The scope of the assessment encompasses extraction and processing of raw materials, manufacturing and assembly processes, product distribution, use, re-use, maintenance, recycling and final disposal (Kniel *et al.*, 1996; Nash and Stoughton, 1994). Life cycle assessment is a 'cradle to grave' approach that reviews the environmental effect of the aforementioned processes in a holistic manner (Barton, 1996).

The origin of the LCA methodology can be traced to the late 1960 (Miettinen and Hamalainen, 1997). Initial studies were simple and generally restricted to calculating energy requirements and solid wastes, with little attention given to evaluating potential environmental effects. The general framework for the LCA methodology changed over the years. Since 1990, attempts have been made to develop and standardise the LCA methodology under the coordination of the Society of Environmental Toxicology and Chemistry (SETAC) (Udo de Haes, 1993). In 1993, SETAC published a 'Code of Practice', which presents general principles and a framework for the conduct, review, presentation and use of LCA findings (SETAC, 1993). An international standard for LCA put together by the International Standardisation Organisation (ISO) has recently emerged and is undergoing evaluation and revision (Burgess and Brennan, 2001). The methodology framework for ISO is similar to that for SETAC with some differences for the interpretation phase, where ISO has included further analysis and sensitivity studies. The ISO standards, recently produced are

- ISO 14040 covering LCA within environmental management,
- ISO 14041 covering inventory analysis,
- ISO 14042 covering impact assessment, and
- ISO 14043 covering interpretation.

The life cycle assessment framework identified in these ISO series of international standards forms the basis of this study.

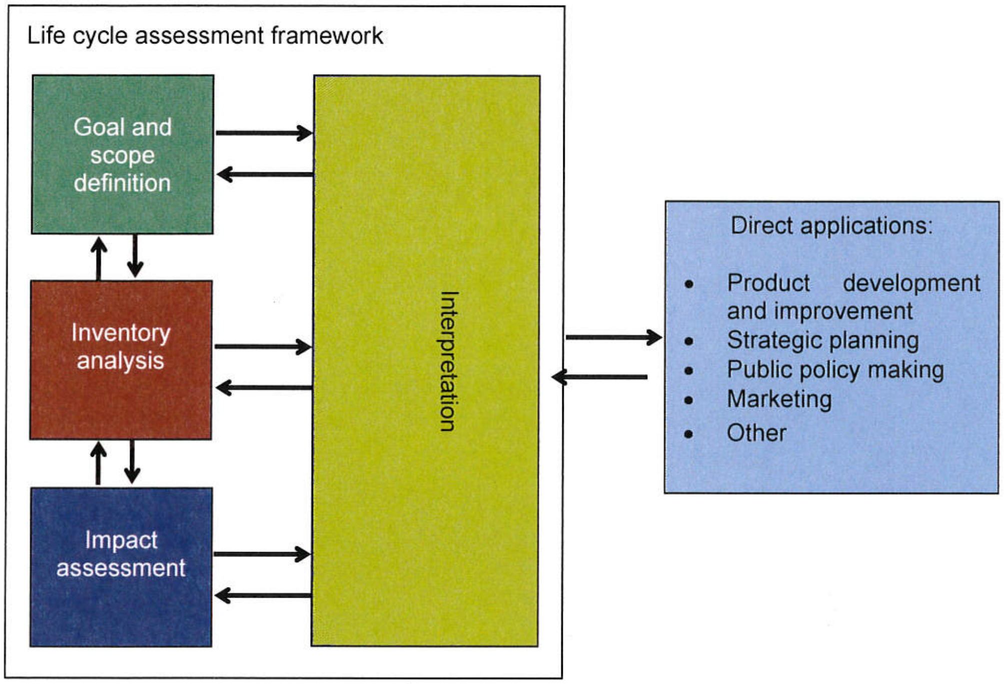
3.2 LCA methodology

The technical framework of AS/NZS ISO 14040 (1998) has been used to conduct this LCA study, which is shown in Figure 3.1. The major components of LCA are:

- Goal and scope definition,
- Life cycle inventory analysis (LCI),
- Life cycle impact assessment (LCIA), and
- Interpretation phase.

These components have been widely used in the evaluation of environmental performance of products or systems. A system is defined as a collection of operations that together performs some clearly defined function. Phases in an LCA are interdependent, as the development and completion of LCA studies is iterative and requires refinement at all levels as new information is generated and/or processed.

Figure 3.1. Phases of an LCA (AS/NZS ISO 14040, 1998)



3.2.1 Goal and scope definition

The development of a clearly defined goal and scope is the first phase of an LCA. It is necessary that the intended application of the study, the reasons for carrying out the study and the intended audience be stated beforehand. During this phase, the context and depth of assessment of the study is identified. This initial step also determines the boundary of the study. An accurate description of the system boundary is very important because data collection is contingent on proper understanding of where each stage of the life cycle begins and ends. Furthermore, a reference unit, to which all the environmental impacts are related, has to be defined. According to the LCA terminology this reference unit is called a functional unit. A functional unit is the function or service that a system provides – for use as a reference point to make comparisons of environmental impacts. An appropriate functional unit for composting processes is the treatment of a specified amount of compostable organics over a specified period.

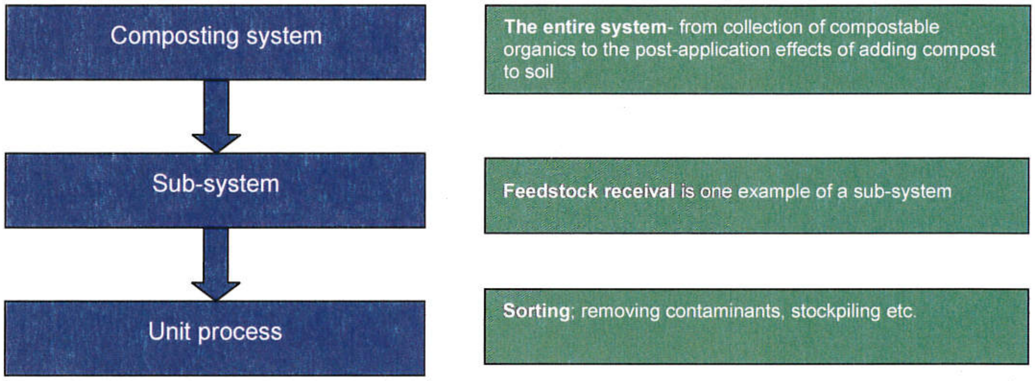
The goal of the LCA in this study is to quantify and to evaluate the environmental impact of windrow composting systems. Receival, processing, transportation and the application of composted products are the functions of the product system considered here. All the environmental impacts are related to the processing of municipally collected, source separated garden organics materials, and the subsequent land application of the composted products as listed below:

1. *Composted mulch*: a relatively low nutrient composted mulch compliant with specification in Australian Standard AS 4454 (Standards Australia, 2003), and produced from 100% garden organics.
2. *Composted soil conditioner*: a relatively high nutrient composted soil conditioner compliant with specification in Australian Standard AS 4454 (Standards Australia, 2003), and produced from either:
    - A mixture of garden organics and food organics; or
    - A mixture of garden organics and biosolids

One product tonne (Mg) of municipally collected, source separated garden organics materials used in the production of above composted products is considered as the functional unit for this analysis. Equally, the resulting LCI data may be used for modelling using a functional unit of one tonne of composted product. The reference system and its boundaries for this analysis are depicted in the Figure 4.1 of Section 4. This study follows the guidelines given in AS/NZS ISO 14040 (1998) for defining the goal and scope.

When describing components in the composting system, a number of levels/scales are considered. These are identified in Figure 3.2. This method of classifying components in the system is used throughout the study.

Figure 3.2 Levels/scales in the composting model.



A clear understanding of the following aspects is very important in the goal and scope definition phase of a LCA (AS/NZS ISO 14040, 1998):

- The functions of the product system (i.e. receival, processing, application),
- The functional unit (i.e. one tonne of source separated organic mixture),
- The product system to be studied (the summary of the system as a whole),
- The product system boundaries (what is included and excluded),
- Allocation procedures (methods of allocating data to different processes in the system),
- Types of impact and methodology of impact assessment (the main environmental impacts considered in the study and how they are assessed),
- Data requirements (what data is required to identify the impacts of the system),
- Assumptions (assumptions made when creating system boundaries, using data sources/types etc.), and
- Limitations (any limitation associated with the overall approach used to identify the total environmental impact of a system).

3.2.2 Life cycle inventory analysis (LCI)

Life cycle inventory analysis (LCI) is a technical process that identifies and quantifies energy and resource consumption, and environmental releases to air, water and land throughout the life cycle of a product or system (AS/NZS ISO 14041, 1999). The following description of LCI is based on AS/NZS ISO 14041 (1999).

In LCI, the energy and resource use and emissions are considered for various processes in a system, including:

- Acquisition of raw material from earth,
- Processing and transformation of raw materials to final products,

- Production and consumption of intermediate products,
- Transportation of raw and finished products, and
- Final disposal of any waste produced during the processing period and at the end of the life of the product.

An iterative process is used throughout the LCI, allowing for the refinement of system boundaries and life cycle stages or unit processes. This will limit further data handling to significant data only (increasing efficiency).

There are several phases to the LCI process:

- Initial preparation,
- Data collection, and
- Calculation procedures.

The initial preparation phase requires that a very clear understanding of the studied system be established. For the purpose of data collection, it is appropriate to view the system as a series of sub-systems or unit processes. Each unit process has energy and resource requirements. Likewise each unit process contributes to environmental emissions through different activities. These need to be clearly identified and documented.

Once the components in the system have been identified, it is necessary to describe data collection techniques for each unit process. At this point, any special issues or irregularities in data requirements are documented. Inventory data for the system can be obtained from a number of sources including:

- Literature,
- Specific studies/reports,
- Laboratory investigations,
- Industry organisations, and
- Government bodies.

Depending upon study data requirements (influenced by system boundary) and the accessibility of appropriate information, collecting data comprises a major part of the LCI process. The steps required for data collection may vary because of differences between individual unit processes. Therefore, LCI requires clear documentation of procedures used and associated reasons for their use.

Data manipulation/calculation is a required step used to create inventory data during the LCI phase. Often data for different components or unit processes is obtained from a range of sources that may not be compatible with the functional unit(s) used in a study. Consequently, data needs to be modified to suit the purpose of the study. A simple example would be having to change the units of measure of data – if data is initially on a per kg basis, it may be necessary to convert it to a per tonne basis, as may be used for the functional unit of the study. Similarly, measures of electricity may have to be converted from kWh to MJ or vice versa depending upon the standard measures used throughout the study. Data may also be aggregated where unit processes or functions result in similar impacts. The Sections 2.1 to 2.6 (Appendix 1) give detail descriptions of phases involved in LCI.

3.2.3 Life Cycle Impact Assessment (LCIA)

Life cycle impact assessment (LCIA) is used to characterise and assess the effects of resource consumption and environmental loadings identified in the inventory stage. The following description on LCIA is based on the AS/NZS ISO 14042 (2001).

Impact assessment is carried out in three different phases:

- Classification,
- Characterisation, and
- Valuation.

The classification phase requires the allocation of all resource inputs and environmental emissions (identified in the LCI) to a range of impact categories depending upon what type of environmental issues they contribute to. Internationally accepted impact categories typically include global warming, ozone depletion, photo-oxidants formation, eco-toxicity, eutrophication, energy, acidification, and human toxicity (Environment Australia, 2001). Where possible impact categories are also based upon indicators of environmental health, as identified in the Environment Australia, "State of the Environment Reporting" document (Environment Australia, 2001). Environmental indicators are "physical, chemical, biological or socio-economic measures that best represent key elements of complex ecosystems or environmental issues" (Hamblin, 1998). The use of nationally accepted environmental indicators and associated terminology will facilitate comprehension of issues presented in the LCA.

Allocation procedures need to correspond with the goals and scope of a study. In some instances LCI emission data can be allocated to more than one impact category if it is required by the goal and scope of the study. For example, nitrous oxide contributes to global warming as well as acidification and eutrophication.

The characterisation phase is a quantitative process where the contribution of each type of emission and resource consumption to different impact categories is determined. The calculation involves the conversion of LCI results to common units and the aggregation of the converted results within the impact category. This conversion uses characterisation/equivalency factors. Characterisation/equivalency factors are developed from characterisation model relevant to environmental mechanism of each impact category. Characterisation/equivalency factors have been developed within the LCA framework to identify how much a substance contributes to a particular environmental impact category compared with a reference substance. For example, nitrous oxide contributes 310 times more to global warming than carbon dioxide. Thus the quantity of nitrous oxide released by a system is multiplied by 310 to derive a carbon dioxide equivalent global warming figure. Carbon dioxide is used as a reference substance for GWP. In some instances these converted results are 'normalised' to better understand their relative magnitude. Normalisation involves the transformation of a result by dividing it by a selected reference value. For example, total emissions or resource use for a given area (global, regional or local), total emissions or resource use for a given area on a per capita basis etc.

Valuation or weighting is the last stage of LCIA phase and involves converting impact category results using numerical factors based on value choices. Weighting is achieved by:

- Converting impact category results or normalised results with selected weighting factors; or
- Aggregating these converted results or normalised results across impact categories.

Weighting is used to facilitate decision-making processes by aiding with comparisons of overall impacts of products or systems studied. The weighting process is, however, not widely used in LCA methodology due to problems associated with weighting factors. Assigning weighting factors is a subjective process and varies with geographical location, and the extent and type of environmental problem. A detailed description of phases involved in LCIA is given in Sections 3.1 to 3.10 (Appendix 1).

3.2.4 Interpretation phase

The objectives of the life cycle interpretation phase are to analyse results, reach conclusions, explain limitations and provide recommendations based on the findings of the preceding phases and to report the results of the life cycle interpretation in a transparent manner (AS/NZS ISO 14043, 2001).

The following steps are usually taken to determine and to enhance the confidence and the reliability of the results of the study including any significant issues identified (AS/NZS ISO 14043, 2001):

- Completeness check – ensures that all relevant information and data needed for the interpretation are available and complete.
- Sensitivity check – assesses the reliability of the final results and conclusions by determining whether they are affected by uncertainties in the data, allocation methods or calculations of impact category results.
- Consistency check – determines whether assumptions, methods and data are consistent with the goal and scope of the study

By addressing data quality and other issues (via the above checks), the following is achieved (AS/NZS ISO 14043, 2001):

- Identification of the significant issues.
- Drawing of conclusions and checking that they are consistent with the requirements of the goal and scope of the study.
- Making recommendations based upon the findings of the study.

A detailed description of the interpretation phase is given in Sections 4.1 to 4.4 (Appendix 1).

3.3 Selected impact categories, category indicators and models

There are a number of impact categories associated with windrow composting systems that are considered in this study and include:

- Global warming,
- Human and eco-toxicity,
- Resource depletion,
- Eutrophication, and
- Land use.

Impacts of post compost applications that could not be categorised under the above stated impact categories have been presented as avoided loads under the following headings:

- Reduction in requirements of irrigation water,
- Reduction in requirements of fertilisers, and pesticides,
- Reduction in soil erosion,
- Carbon sequestration,
- Improvement in soil health (soil structure), and
- Remediation of saline and sodic soils.

Life cycle inventory data have been developed and classified under each of these category using existing environmental indicator classifications, as described in the State of the Environment Report by Environment Australia (Environment Australia, 2001) and provided in Appendix 2. The use of consistent national terminology will facilitate comprehension of impacts discussed, and support future comparative analysis of environmental impacts of technologies and systems in an Australian context.

This study quantifies the environmental impacts associated with the composting process from the point at which compostable organic materials are delivered to a commercial composting facility, to the agricultural application of the resulting composted product, including post-application impacts and benefits. If quantification is not possible, the relevant environmental impact is addressed qualitatively.

3.3.1 Energy consumption

Modern agricultural production is largely dependent on non-renewable energy and imported resources. Direct energies (i.e. energy carriers) are essentially used for their energy properties and thereby assessed on the basis of their energy content. In contrast, indirect energies (such as fertilizers, pesticides and machines) are assessed on the basis of the energy required for their use according to the production inventory. Energy could be defined into different categories such as process energy, inherent energy, and production and delivery energy (Audsley *et al.,* 1997). Process energy is the energy input required and consumed by the considered process to operate within the process phase. Inherent energy (feedstock energy) is the extracted energy that remains in the product after its production and delivery to its site of use. Production and delivery energy is the energy input into the processes, which extract, process, refine and deliver energy and material inputs to a process.

In LCA studies, energy analysis takes into consideration the energy consumed and energy gained in the system per functional unit. The inherent energy of commonly used energy carriers is listed in Table 3.1.

Table 3.1 Energy content in energy carriers.

Energy carrier	Inherent energy (MJ kg^{-1})
Diesel	42.8
Hard coal	27.1
Natural gas	46.1
Oil	42.7
Electricity	1 MJ MJ^{-1}

Source: Audsley *et al.* (1997)

3.3.2 Greenhouse effect

A build up of greenhouse gases (GHG), mainly carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O), in the atmosphere is causing temperature to rise due to the absorption of long-wave radiation re-transmitted by the earth surface by these gases (Recycled Organics Unit, 2001b).

International protocol has established carbon dioxide as the reference gas for the measurement of the heat-trapping potential [*global warming potential* (GWP)] of GHGs. Global Warming Potentials are expressed as CO_2 equivalents (relative to the GWP of unit mass of CO_2). The GWP of carbon dioxide, methane and nitrous oxide are summarised in Table 3.2. Emissions of these gases are also converted to their 'carbon equivalents' on the basis of carbon fraction in carbon dioxide (C = 12/44 CO_2) (Recycled Organics Unit, 2001b).

The GWP of gas emissions may be expressed in terms of delayed time (20, 100 or 500 years) after initial emission. In most instances the gas impact subsides as time is projected – it is also discounted, as future impacts are less certain.

Carbon sequestration is the opposite of GHG emissions. Carbon is removed from the carbon cycle (or from the atmosphere) and added to a carbon sink. A carbon sink is a point in the carbon cycle where carbon is stored for a long period of time. While carbon is stored, it is not in the atmosphere contributing to the greenhouse effect. Examples of carbon sinks include soils, forests and oceans (Recycled Organics Unit, 2001b).

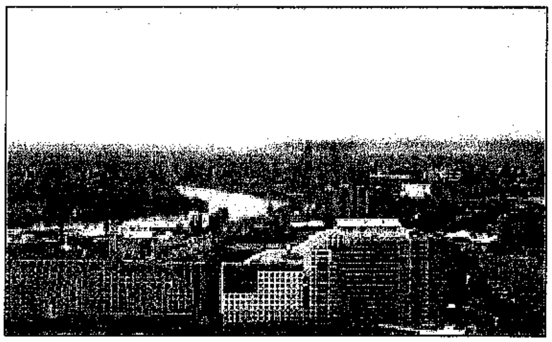
The United Nations has established a Framework Convention on Climate Change. Only those greenhouse gas emissions resulting from *anthropogenic* sources (from human activity) are accounted for in this framework. Emissions that are generated from *biogenic* processes (emissions that occur naturally, or would have happened during the natural decomposition process) are excluded from the analysis (Recycled Organics Unit, 2001b).

Table 3.2 Global warming potential and related carbon equivalents of GHGs.

Greenhouse gas	**Quantity** (kg)	**Global warming potential** (CO_2 eq.)	**Carbon equivalent** (kg of carbon)
Carbon dioxide	1	1	0.27
Methane	1	21	5.67
Nitrous oxide	1	310	83.7

Source: US EPA (1998)

3.3.3 Photochemical oxidant formation



Photochemical oxidants are trace species that are formed during the photo-oxidation of volatile organic compounds (VOCs), carbon monoxide (CO) and oxides of nitrogen (NO_x). Examples include ozone (O_3), hydrogen peroxide (H_2O_2) and peroxy acetyl nitrate ($CH_3C(O)OONO_2$, PAN) (Kley *et al.,* 1999). Among these, the pervasive surface level of O_3 is considered to be the most important (McKee, 1994; National Research Council, 1992). Ozone is photochemically produced in the troposphere through oxidation of CO and VOCs in the presence of NO_x (Kley *et al.,* 1999). The gas phase chemistry of photooxidant formation is difficult, because many VOCs of anthropogenic and biogenic origin are emitted into the atmosphere (Kley *et al.,* 1999).

The prevalence of tropospheric photochemical oxidants is of major international concern, because of their adverse effects on human health and the environment (US EPA, 1996). They are respiratory irritants that cause physiological effects, lung inflammation and disruption or death of cells along the respiratory tract (Kley *et al.,* 1999). They also reduce the plant growth and productivity (Kley *et al.,* 1999).

In urban and industrial areas, many VOCs, CO and NO_x are emitted from anthropogenic sources, such as transportation, fossil fuel-burning power plants, chemical plants, petroleum refineries, certain construction activities, solid waste disposal and slash burning (Arya, 1999; Davis and Otson, 1996). In addition to the anthropogenic sources, many VOCs are produced naturally by vegetation (Arya, 1999).

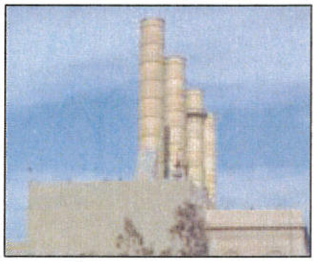
The concept of Photochemical Ozone Creation Potential (POCP) is used to quantify the contribution of each photochemical oxidant to ozone formation (Andersson-Skold *et al.,* 1992). Photochemical Ozone Creation Potentials are expressed as ethylene-equivalents (ozone formation due to unit mass of photochemical oxidant relative to the ozone formation by unit mass of ethylene). Table 3.3 shows POCPs of some common emissions. As a result of changing spatial patterns of the ozone photochemistry, POCPs are not fundamental, geophysical quantities but are derived quantities depending on rate coefficients, solar actinic radiation, dry deposition velocities and environmental variables (Derwent *et al.,* 1996; Lindfors *et al.,* 1995).

Table 3.3 POCPs for characterising photo-oxidant forming releases to air.

Substance (kg)	**POCP at high NO_x background** (ethylene eq.)
Carbon monoxide	0.027
Nitrogen dioxide	0.028
Sulphur dioxide	0.048
Ethylene	1.0
Methane	0.006

Source: Guinée *et al.* (2001)

3.3.4 Human and Eco-toxicity



Toxic substances released during production and application of compost, fertilizers, pesticides, biocides etc. may be toxic to humans and the environment. These releases may be gaseous, liquid or solid. Human exposure to these chemicals through food, air, water and soil causes health problems. These chemicals also have ecotoxicological impacts on aquatic, terrestrial and sediment ecosystems and could decrease biodiversity. A variety of characterization methods are available to measure the human and eco-toxicity impacts of these substances. These impacts are generally referred to human toxicity potential (HTP) and eco-toxicity potential (ETP). Human Toxicity Potentials and ETPs are usually based on the impact of a reference chemical on human and ecosystems. The toxic potentials are substance-specific, quantitative representations of potential impacts per unit emission of a substance that can be used as weighing factors in aggregation of emissions coming from life cycle inventories (Huijbregts *et al.,* 2000). For example, USES 2.0 (RIVM *et al.,* 1998) uses 1,4-dichlorobenzene as the reference chemical and impacts are measured related to 1,4-dichlorobenzene equivalents.

The toxic potentials suitable for Western European conditions provided in the USES-LCA model and modified for Australian conditions by Huibregts et al. (2001) have been used in this study to estimate the human and eco-toxicity impacts of compost production and application. The adapted model, called USES-LCA$^{\text{Australia}}$, has been used to calculate toxicity potentials for 181 substances and the substance group carcinogenic Polycyclic Aromatic Hydrocarbons using an infinite time horizon. The modified model has found that ecotoxicity potentials for the majority of substances did not differ for Australia and Western Europe. However human toxicity potentials were lower for Australia compared to Western Europe due to the smaller population in Australia and the Southern hemisphere compared to the Northern hemisphere.

3.3.5 Resource depletion

Most of the environmental crises confronted today are direct consequences of resource use pattern and depletion. Over the last century, the exploitation of material resources has grown enormously in parallel to the growth of economic activities, resulting in depletion of limited reserves of high-grade resources. For example, phosphorus is concentrated in relatively few large deposits, which are expected to be depleted within the next 100–200 years. In addition, mining of other minerals results in depletion of other resources such as land, vegetation, etc. and causes environmental pollution. The above photo shows an area that has been mined and is awaiting reclamation. Resource depletion is an important issue in life cycle impact assessment studies to compare different production systems.

Resource depletion can be either abiotic (non-biological) or biotic (biological). Abiotic resources are natural resources such as phosphate ore, crude oil and wind energy, which are non living while biotic resources, are living material resources, e.g. rainforest (Guinée *et al.,* 2001).

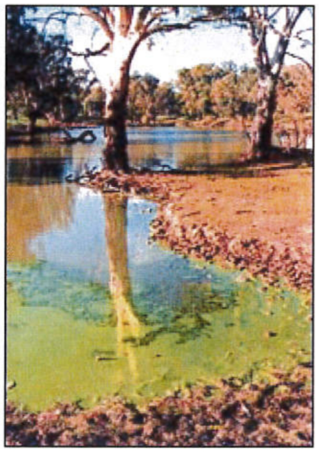
Resource depletion is one of the most frequently discussed impact categories and there are consequently a wide variety of methods available for characterising contribution to this category. In this study, only abiotic resource depletions are considered. Aggregation and assessment of the impact associated with the resource depletion are measured in terms of their exergy content (Ayres *et al.,* 1996; Finnveden, 1996). Exergy is the energy content that can be used for work (available energy). Factors to characterise some resource depletions are given in Table 3.4.

Table 3.4 Factors for characterising abiotic resource depletion based on exergy content.

Material	**Factor** (MJ kg^{-1})
Gypsum (calcium sulphate)	1.485392
Sulphuric acid	16.0259452
Calcium oxide	6.1799058
HCl	3.0846006
NaCl	0.835692
Water	0.1711425
Urea	115.01732
Amnmonia	5.7547749
Nitrous Oxide	4.7049897
Ammonium Chloride	17.7218996
Calcium chloride	9.7556694
Calcium hydroxide	3.9789015
Fluorine	17.7180011

Source: CML LCA Operational Guide to ISO Standards, May 2001, CML Leiden University

3.3.6 Eutrophication



Eutrophication refers to the enrichment of ecosystems of nutrients via water and air and may cause an undesirable shift in species composition and elevated biomass production in both aquatic and terrestrial ecosystems. In addition, high nutrient concentration may also render surface and ground water unacceptable as a source of drinking water. In aquatic ecosystems, increased biomass production may lead to depressed oxygen levels, because of additional consumption of oxygen in biomass decomposition (increased biological oxygen demand). Eutrophication covers all of these potential impacts due to excessively high environmental levels of macronutrients, the most important of which are nitrogen (N) and phosphorus (P). As emissions of degradable organic matter have similar impact, such emissions are also treated under this impact category.

Table 3.5 EP for characterising eutrophying releases to water.

Substance (g)	EP (g 0_2 depletion) P-limited	EP (g 0_2 depletion) N-limited
Ammonia (air)	3.8	19.8
Ammonium (water)	3.6	18.6
Nitrate (water)	0	4.4
COD (water)	1	1
Nitrogen dioxide	0.13	-
Nitrogen monoxide	0.2	-
Nitrogen oxides (air)	0	6
Phosphorus (water)	140	0
Phosphorus (V) oxide P_2O_5	1.34	-

Source: Karrman and Jonsson (2001)

In this study eutrophication impact category is estimated from the potential oxygen demand of emissions to water. This impact category contains the impact of water emissions on the total oxygen consumption in the receiving water. In this study, the EPs are calculated for P-limited and N-limited emissions and the EPs of some emissions are shown in Table 3.5.

3.3.7 Land use

Land use has received increasing attention in life cycle assessment studies. The category impact of land use covers range of consequences of human land use. It is a relatively new topic in life cycle impact assessment (LCIA) and still being debated and developed. Land use impacts in LCIA are related to the area of land use, generally in combination with the time required to produce a certain output. Two aspects can be distinguished under the land use impact category. The associated changes in the quality of the land (transformation) typically expressed in biodiversity and/or life support functions and the length of time for the land is used (occupation) (Guinée *et al.*, 2001).

Land transformation is the process of changing aspects of biodiversity and life support functions. i.e. flora, fauna, soil acidity, sodicity, salinity, erosion, productivity etc. Transformation impact is expressed in units of quality multiplied by area. Occupation refers to the time period during which the land is unavailable for other uses. The occupation impact represents the temporary changes in the quantity of area of land and expressed in area multiplied by time (m^2 x years).

In this study, the following straightforward expressions to assess the land use impact given by Lindeijer (2000) are used. Quality differences are measured in terms of productivity (in $), i.e. an application of compost may increase the productivity by increasing nutrient availability.

Land occupation impacts = area x time x quality
Land transformation impact = area x quality difference

3.4 Challenges encountered in LCA

It is often reported in the literature that LCA has not yet matured into a well-defined tool (Burgess and Brennan, 2001; Huppes, 1996). Discussion on basic principles and how methodological limitations should be approached are ongoing. These limitations and difficulties restrict the practical application of LCA. However, methodological proposals are never going to suit all case studies, and opportunities exist for LCA practitioners to adapt the methodology according to their application (Burgess and Brennan, 2001).

Despite being the more objective component of LCA, inventory analysis still lacks agreement on a number of critical points (Udo de Haes, 1993). Furthermore, inventory results from studies with similar systems can still vary significantly (Keoleian, 1993). Allocation of resource consumption, emissions and waste streams into different systems sometimes leads to confusion as emission or waste leaving a system may be used in other systems. For example, when two or more products being studied are saleable or leave the system for beneficial use in other systems, allowance must be made for their share of environmental burden and hence of the inventory data. This problem of co-product allocation to multiple output streams is the subject of international discussion (Burgess and Brennan, 2001). The problem of allocation can actually be avoided in some cases by extending the system boundaries to include more than one product (Stromberg *et al.,* 1997; Lindfors *et al.,* 1995). However, this method may require substantial additional data collection. Furthermore, the results of such a procedure are often more difficult to interpret, with the risk of information on the life cycle under study becoming blurred (Stromberg *et al.,* 1997).

In defining the system boundaries, assumptions are necessary to maintain the manageability, with a weighing of completeness on the one hand against practical feasibility on the other. The analysis becomes meaningless and the assessment will underestimate environmental effects if boundaries are not properly demarcated. In some studies, important life cycle stages are excluded due to pragmatic limitations in the data collection process (Burgess and Brennan, 2001). It is often asserted in the literature on LCA that many of the apparent contradictions between studies of similar systems arise because the systems are defined inadequately (Clift, 1998). Furthermore, the determination of system boundaries is dependent on individual goals and the scope of the study (Stromberg *et al.*, 1997). For example, processes can be excluded from system boundaries when they are common to all alternatives compared. Lee *et al.* (1995) suggested that a potential method of determining the system boundary is to evaluate the effect of a particular process or activity on the overall LCA results, and if the effect is significant, then include it in the analysis. Therefore, it is extremely important that a precise definition of the system boundaries is stated as part of the final results in any LCA.

The question of whether to include capital goods in LCA needs to be addressed in LCAs. Heijungs (1992) believes that if the cost of maintenance and depreciation are a substantial part of the product price, then the environmental impacts associated with the production, maintenance and disposal of equipment should not be excluded from the analysis. Tillman *et al.* (1994) suggested that capital goods would need to be included in the life cycle analysis if it were necessary to compare the production and operation of new equipment with continued use of existing equipment.

The quality of the data used in LCA is a very important issue. There is a tendency to use low quality data with the progress of the LCA methodology. Until recently, LCA practitioners have been forced to concentrate on finding the data rather on the quality (Lindfors *et al.,* 1995). Even though all inventories will have data variability, uncertainties and gaps, it is important that these issues are considered in relation to the goal of the study (Burgess and Brennan 2001). A complete record of all data and the methods used to obtain it is important when presenting LCA results (Burgess and Brennan, 2001). Some analyses are based on confidential information, which is not publicly available.

These studies are inclined to be controversial due to the inability of validating the accuracy and reliability of such data (Hendrickson *et al.,* 1998). The lack of public data is one of the reasons why LCA is such a time consuming and costly procedure (Udo de Haes, 1993). The normalization step is often frustrated by lack of such data.

In LCA methodology, temporal and spatial variations of impacts are usually ignored in the impact assessment step (Burgess and Brennan, 2001). These variations are important, as most processes are not at steady state with the time and space. Therefore, it is important to specify the temporal and spatial representativeness of the system at goal definition stage of an LCA (Burgess and Brennan, 2001).

While the tools needed to carry out the inventory analysis are well developed, interpretation of results causes difficulties (Burgess and Brennan, 2001). Furthermore, the lack of a standard impact assessment methodology compounds the problem. Assigning relative weights to each impact category is the most contentious and subjective stage of life cycle studies, with the necessity for trade-offs between the different environmental problems (Burgess and Brennan, 2001). There are also difficulties with the normalization component of the impact assessment, with no internationally accepted norms to carry out this step. The main difficulties are connected with the choice of reference area and finding appropriate data (Udo de Haes, 1996).

Section 4 Windrow composting system model

A comprehensive model has been developed for windrow composting systems (Figure 4.1). The inputs, outputs and emissions of windrow composting (including land application of compost product) are defined in this model. For the purposes of the current investigation, the system boundary of the model is from the point at which feedstock is received at a composting facility gate, to the post-application effects of adding compost to soil.

4.1 Overview

A conventional windrow composting system results in impacts and avoided impacts from a number of individual components (sub-systems). These sub-systems include:

- Collection of compostable organics material
- Construction of capital equipment and infrastructure
- Feedstock receival and initial processing
- Processing
- Transportation and application of end-product
- Post-application effects of adding compost to soil

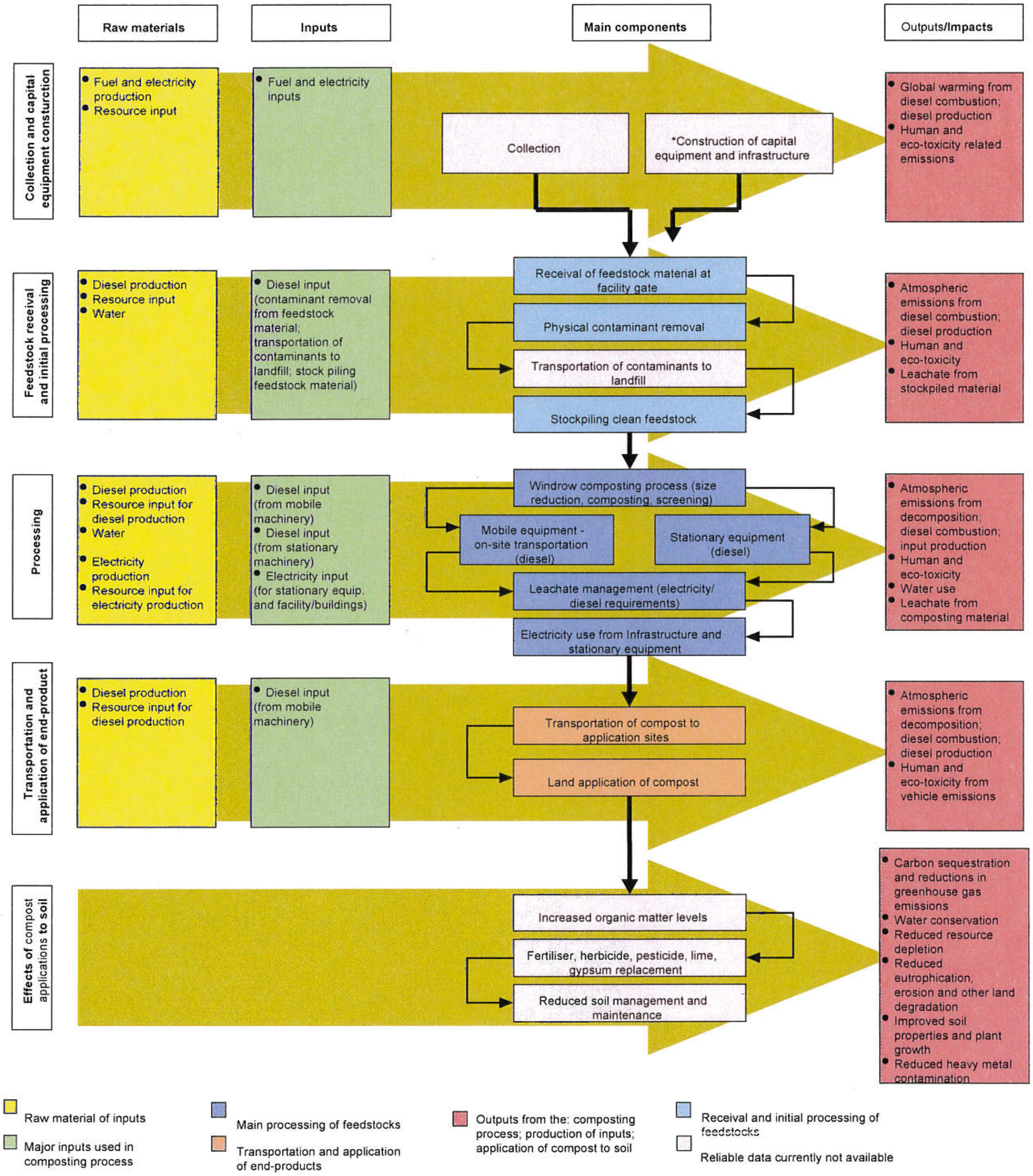
These different sub-systems are described in the following sections. The scope of the study excludes the first two sub-systems (collection of compostable organics material; and construction of capital equipment and infrastructure), and focuses on composting facility processes, the transport and application of composted product to agricultural land, and the post-application effects of compost on soil. These system boundaries are detailed below.

4.2 System boundary

The system boundary for this study has been defined within the context of the windrow composting system model, and is shown graphically as dotted line in Figure 4.1. The system boundary encompass impacts associated with:

- The receival of feedstock material at a facility gate, short-term storage or stockpiling of feedstock material and the removal of physical contaminants from feedstock,
- The fuel usage at a facility – from both mobile and stationary sources,
- The electricity used by a composting facility,
- The dry weight loss of the feedstock material,
- The transportation of the end-product to application sites,
- The spreading of the composted end-product onto land, and
- The post-application effects resulting from the application of compost to agricultural soil.

Figure 4.1 System model for windrow composting. * Note: environmental impacts from embodied energy in capital equipment and infrastructure are relevant to every stage of this model.



4.3 Exclusions

In defining the system boundaries, the following elements were excluded from the current study:

- Collection of compostable organic materials,
- Emissions from the construction of capital equipment or infrastructure,
- Transportation of physical contaminants to landfill, and
- Product blending, testing, and bagging, etc.

4.3.1 Collection of compostable organic materials

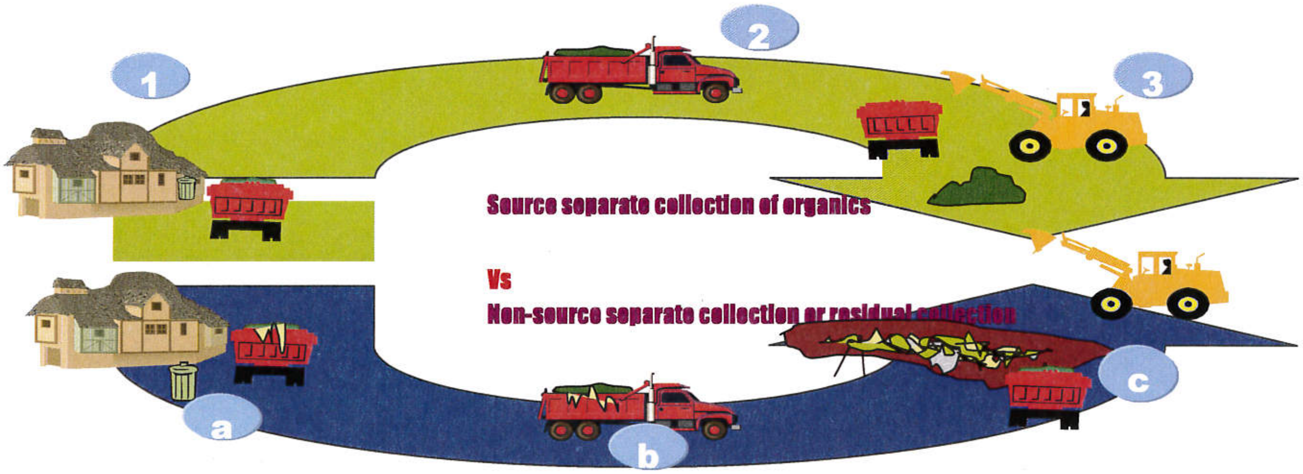
Although compostable organic materials (raw materials) collection is recognised as an important aspect of the entire composting system, it is beyond the scope of the current investigation. Collection systems have been studied elsewhere, and is therefore not a key gap requiring additional attention within the scope of this study.

Whilst environmental impacts associated with collection have not been included in this study, this does not suggest that such impacts are unimportant, but rather that they are beyond the scope of this study and are better examined in the context of complementary and comprehensive studies of collection systems and their associated emissions.

There are a great variety of collection systems in operation It should not, however, be assumed that the separate collection of compostable organic materials will necessarily have any significant impact in terms of increased emissions resulting from source-separated collection systems when compared with emissions from single bin collection of municipal solid waste. In both instances, if the total quantity of material collected is the same, if collection frequency is unchanged (which is supported by new technology such as the *Cleanaway Bioinsert*), and if trucks operate to collect 'waste' material until they are full (Figure 4.2), then there should be negligible increase in fuel use or resulting emissions from collection activities. Under such conditions, there is no difference in time requirements or distance travelled if the composting facility is located in a similar location to the landfill site.

The resulting modelling will allow for variation in transport distance to landfill (or alternative technology treatment site) versus transport distance to compost facility. Therefore if the same quantity of materials is collected and transported to a treatment or disposal facility, there is no expectation that environmental impacts will necessarily be different regardless of the nature of the processing / disposal facility.

Figure 4.2 Comparison between a source separate collection system (1-3) and the regular collection of urban municipal solid waste (a-c).



Note: In both instances, a truck collects material from houses until the truck is full (1 and a). Once the truck is fully loaded it transports the material (2 and b) to the end point (either composting facility, landfill, or other alternative processing technology facility) (3 and c).

4.3.2 Capital equipment and infrastructure

The environmental impacts associated with the construction of windrow composting facility equipment and infrastructure is not considered in this study. This excluded component includes emissions from the construction of:

- Buildings, concrete or asphalt pads
- Tractors, bulldozers, front-end loaders, windrow turners, shredders, grinders, macerators, trommel/power screens, manure spreaders (often used for compost application to land).

If this component were included, total emissions from the construction of capital equipment and infrastructure would be divided by the expected life span (in years) of the equipment/infrastructure to obtain yearly amounts.

Impacts associated with the construction of capital equipment and infrastructure is commonly excluded from LCA studies as such impacts are commonly relatively insignificant compared the emissions associated with operations across the life of the facility, particularly in less capital-intensive operations such as composting facilities (pers comm. Sven Lundie). However, it is important to consider that such embodied energy and impacts in relation to infrastructure will necessarily be greater for processes involving high capital-intensive infrastructure such as *waste to energy* facilities and enclosed and in-vessel biological treatment systems.

4.3.3 Transportation of physical contaminants

The transportation of contaminants to landfill or some alternative location is not considered within the system boundary of this study, as physical contamination of compostable organic materials in a source separate collection system is rightly considered an externality of the collection system, which is excluded from this study. Solid waste residuals and their transport to landfill for disposal are not by-products of composting operations, but rather by-products of the entire collection system. Composting of organic materials does not produce inorganic solid waste. Any such solid waste is an externality of the collection system and its component parts including education and enforcement, and residuals collection.

Section 5 LCI: The composting process

5.1 Introduction

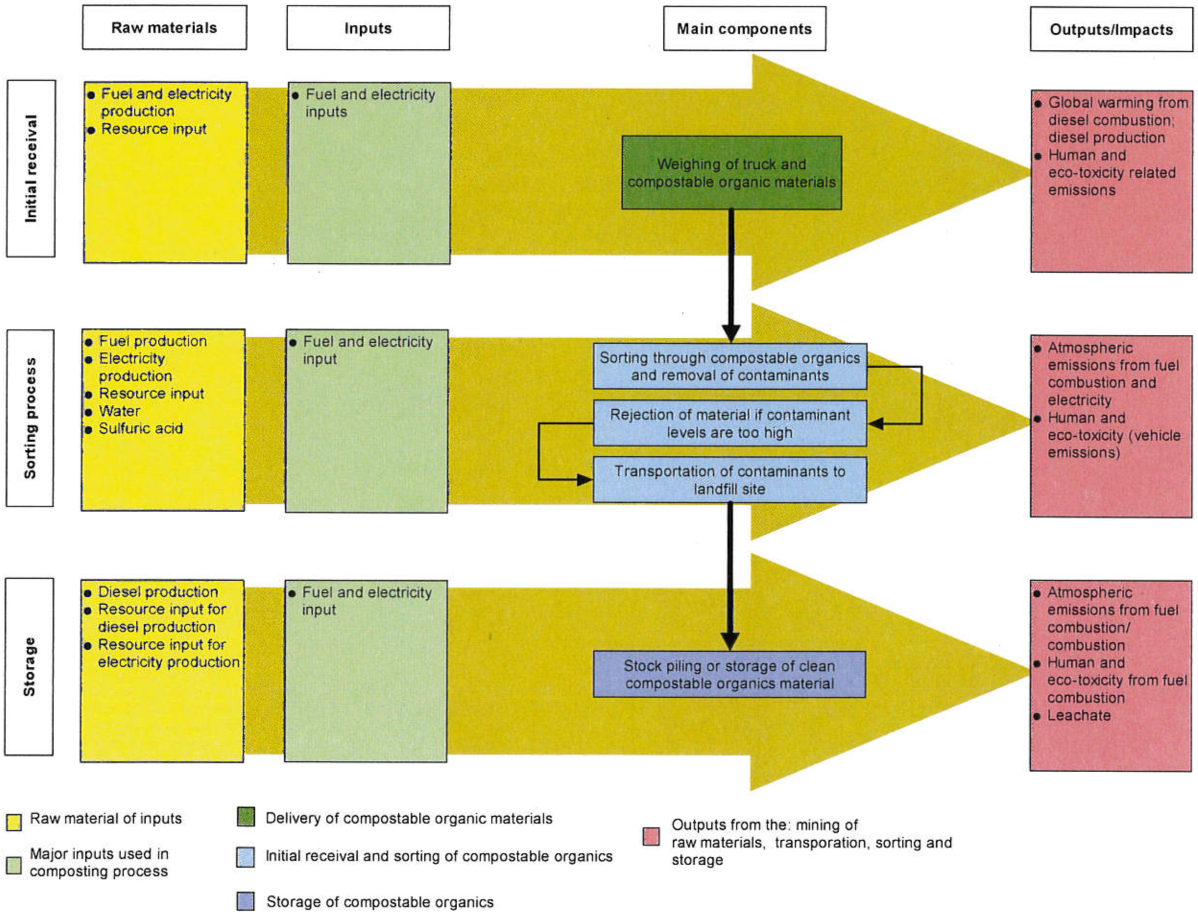
A conventional windrow composting process in a composting facility involves following two major operations.

- Receival of compostable organic materials (raw materials) and initial processing (involving removal of physical contamination and stockpiling of clean raw materials), and
- Composting (including size reduction, formation of windrows, composting operations, product testing and preparation).

5.2 Receival of compostable organic materials and initial processing

Receival of compostable organic materials and initial processing includes: receiving organic materials at the facility gate; weighing and visual assessment of organic materials; and removal of physical contaminants such as plastic, metal and glass; stockpiling of clean materials for future size reduction and composting.

Figure 5.1 Sub-system model for compostable organic materials receival process in windrow composting system model.



The physical contaminant removal procedure commonly involves the spreading of raw materials by a wheel loader, and the use of manual labour for the removal of physical contaminants. If physical contaminant levels are sufficiently low enough (e.g. <5%), then the contaminants are removed and the relatively 'clean' feedstock is stockpiled. If contaminant levels are too high, the raw material is rejected, and loaded back onto a truck for landfill disposal. The rejection and disposal of excessively contaminated loads is not considered within this study, as it is an externality of a poorly managed collection system (refer to Section 4.3.1 and 4.3.3). Compost facilities only process clean, compostable organic materials; compost facilities are not contracted to process contaminated materials. Hence the delivery of contaminated materials is equivalent to any other manufacturing enterprise being delivered with materials that do not comply with supply specifications, and are therefore rejected.

Table 5.1 LCI data for key receival processes.

Function	Equipment	Engine capacity / diesel consumption			Quantity of materials	Duration of use
Spreading of compostable materials & physical contamination removal *	Volvo L120 wheel loader, with 5 cubic metre (m^3) bucket	165 kW	224hp	15 L/hr**	220 Tonnes (@ bulk density of ~ 285 – 333 kg m^{-3})	7 hrs

Source: Australian Native Landscapes Pty. Ltd. (NSW), and Natural Recovery Systems (VIC).
*Note: this includes stockpiling of clean materials for future size reduction and composting.
**Volvo has provided fuel consumption data for handling light materials.

5.3 The composting processes

The composting process involves the size reduction of clean compostable organic materials (as necessary); the mixing of different materials to form an appropriate feedstock; the addition of water to the windrow to achieve suitable moisture content; the turning of the windrow for aeration and mixing; and the screening of the mature composted windrow for particle size grading into soil conditioner, and mulch. The sub-system model showing the processing stages is depicted in Figure 5.2.

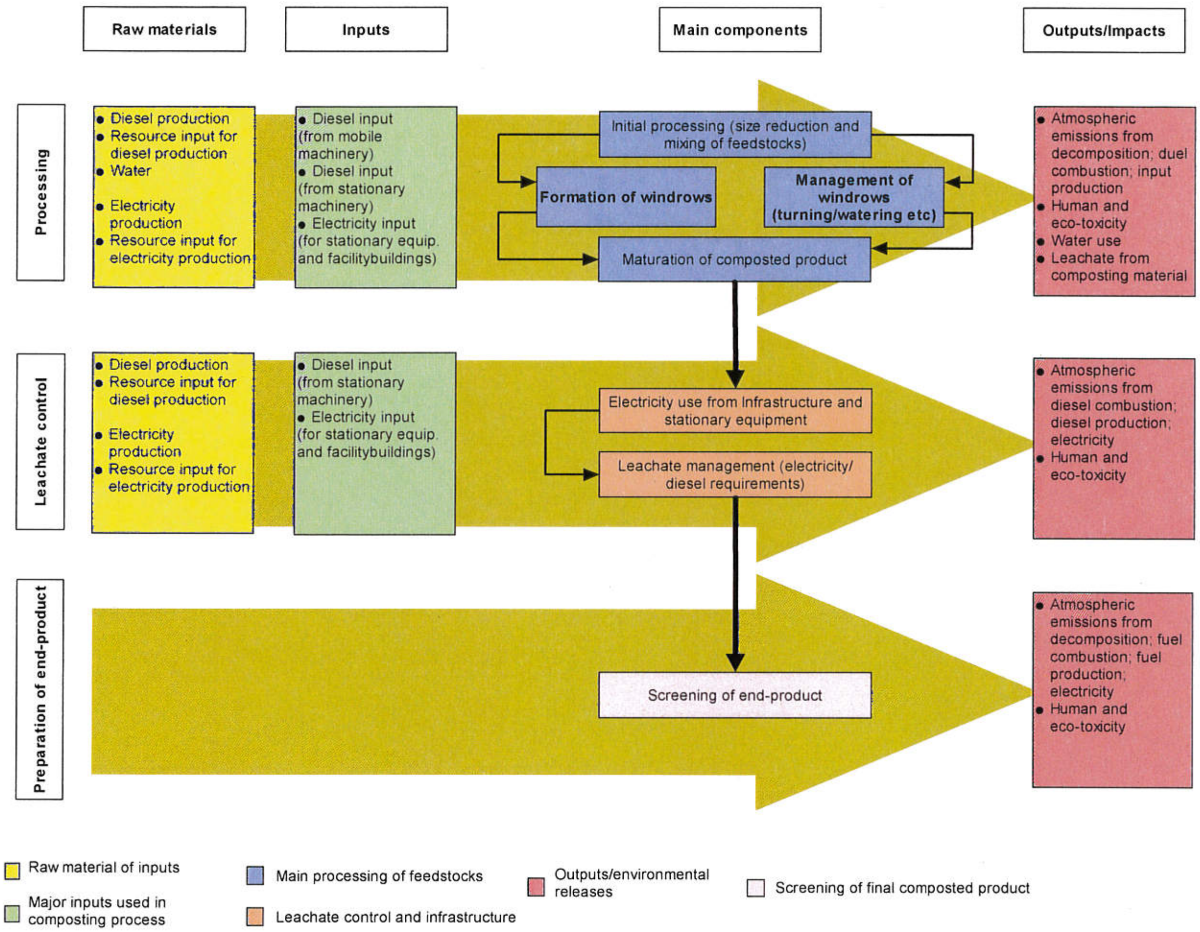
There are additional ancillary management procedures including the monitoring of windrow conditions, sampling of product for quality testing etc. These activities are very minor in terms of resource and energy uses compared to the physical composting operations, and are not included in this analysis.

For source-separated collected municipal garden organics, the C:N ratio of the materials is reasonable for composting without any need to blend with other materials. Clean garden organics are simply shredded to reduce particle size, and formed into a windrow using wheel loaders. Water is added early in the process as required to achieve suitable moisture content throughout the windrow.

Generally for garden organics composting, all the material handling, windrow formation and turning is conducted using a wheel loader. Under good management practices a windrow usually takes 12 – 16 weeks for processing, and windrows are turned every 3 – 4 weeks.

After composting is completed, windrows are passed through a screen to grade the gross compost into a range of graded products according to particle size. Screening can also remove remaining physical contaminants for disposal and oversized wood particles for reprocessing.

Figure 5.2 Sub-system model for processing stages in windrow composting system model.



Note that runoff/leachate management is a component of licensing conditions for commercial composting facilities, and such effluents are collected for use as a process input for adding water to windrows as required. As a result, commercial facilities do not normally consume external water resources, and do not produce leachate as a pollutant. Therefore, in relation to environmental impacts resulting from water budgeting and windrow irrigation, only energy consumption resulting from the pumping of collected water for windrow irrigation is included in this study.

As mentioned above the composting facility only uses water collected on-site i.e. from rainwater and leachate generated from composting piles. This water is collected typically in large "catchment dam". High volume, low pressure electric pumps are used to pump water from the main catchment dam to a series of "feeder dams" (@ 1200 L/min), from which water is pumped through an irrigation line to a series of 8-10 sprinklers using smaller pressure pump (@100 L/min) which water a single windrow. Sprinklers operate 10 hrs/day for between 7-10 days in total across the first 4 weeks of the composting cycle after the windrow is first formed. Once this is completed, the windrow contains sufficient moisture for the remainder of the composting cycle, which is redistributed throughout the composting mass each time the windrow is turned.

The changes (chemical, physical and biological) involved in the composting process are summarised in Figure 5.3. Note that carbon dioxide is identified as the main gas released during composting, and that this is considered as a biogenic emission.

Figure 5.3 The composting process and related emissions (from Gray and Biddlestone, 1971).

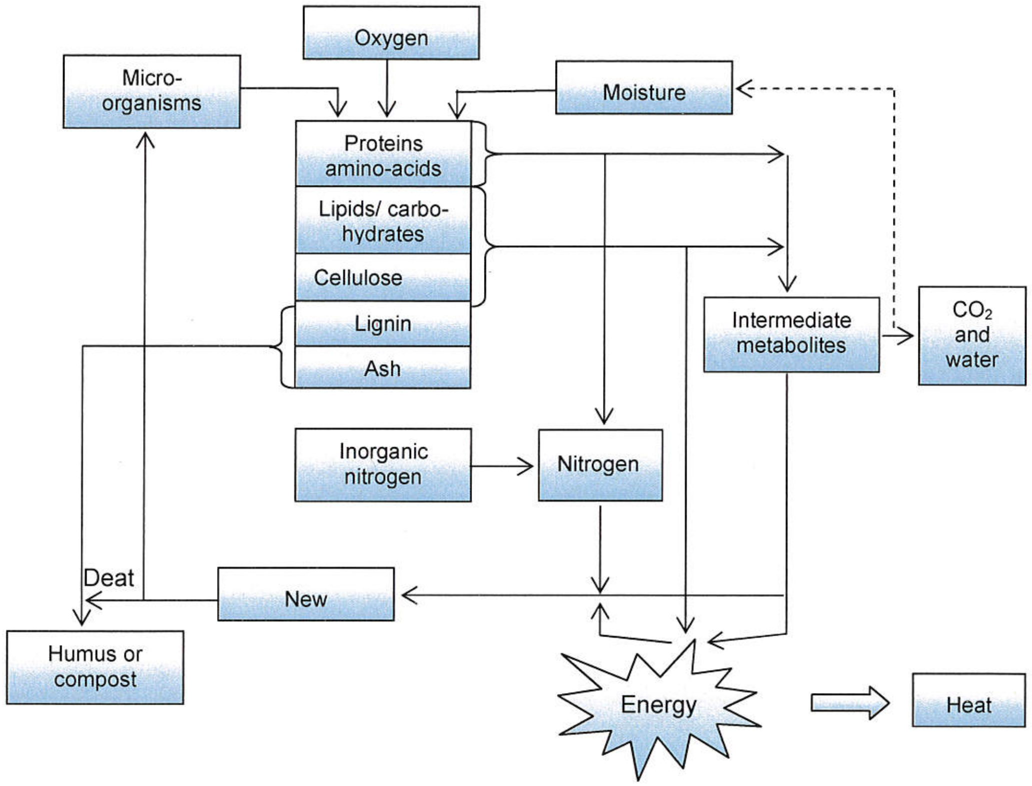


Table 5.2 LCI data for key composting processes.

Function	Equipment	Engine capacity / fuel consumption*			Quantity of materials	Duration of use
Size reduction operations	Matilda Mk 1 tub grinder		1200 hp	200 L/hr	60 Tonnes (@ bulk density of ~309 kg m^{-3})	1 hour
	CAT 322L excavator	123 kW	165hp	17 L/hr	Dictated by shredder rate	Operates continuously whist shredder is operating
Windrow formation and turning **(5 times in total per 16 week cycle)**	Volvo L120 wheel loader, with 5 cubic metre (m^3) bucket	165 kW	224hp	15 L/hr	200 m^3 (@ bulk density of ~ 650 kg m^{-3})	1 hour
Windrow watering is carried out using water collected from rainwater and lechate at the site in a catchment dam.	High volume low pressure, electric pumps	4 kW				Transfers water from catchment dams to a series of feeder tanks @ 1200 L/min
	High pressure electric pumps	2.2 kW				Transfers water from feeder tanks to windrow via 8-10 sprinklers @ 100 L/min.
Screening	Chieftain 1200 Power screen			16 L/hr diesel	60 m^3 (@ bulk density of ~ 650 kg m^{-3})	1 hour
	Volvo L120 wheel loader, with 5 cubic metre (m^3) bucket	165 kW	224hp	15 L/hr	Dictated by power screen operations	Operates 70% of power screen operation time
Dispatch	Volvo L120 wheel loader, with 5 cubic metre (m^3) bucket (high lift arms)	165 kW	224hp	15 L/hr	480 m^3 (refer to Section 5.4 below for bulk density of each compost product)	1 hour

Source: Australian Native Landscapes Pty. Ltd. (NSW), and Natural Recovery Systems (VIC).

*Please note fuel consumption data has been obtained from these companies.

5.4 Changes in bulk density and moisture during composting

Upon arrival at a composting facility, garden organics can range in bulk density from ~ 165 kg m^{-3} (6 m^3 per tonne) for loose drop off materials, and between ~ 285 – 333 kg m^{-3} (3 – 3.5 m^3 per tonne) for source separated kerbside collected materials that are commonly compacted either in collection vehicle and/or transfer stations prior to receival at the compost facility (pers. comm. Australian Native Landscapes; Natural Recovery Systems).

The size reduction (via tub grinder or shredder) of these materials results in shredded garden organics with a reasonably consistent density (freshly shredded) of ~ 333 kg m^{-3} (3 m^3 per tonne). This freshly shredded material is formed into a windrow, with the addition of water in the initial stages to achieve a moisture content of ~50 - 55%. During the composting cycle, in particular prior to turning, the bulk density of the mass increases to ~ 650 kg m^{-3} (pers. comm. Australian Native Landscapes; Natural Recovery Systems).

After screening and curing, for composted soil conditioner the final bulk density is ~ 950 kg m^{-3} (~ 1.05 m^3 per tonne) with moisture content of ~ 40%. After screening and curing, for composted mulch the final bulk density is ~ 480 - 540 kg m^{-3} (~ 2 m^3 per tonne) with moisture content of ~ 20 -25% (pers. comm. Australian Native Landscapes; Natural Recovery Systems).

5.5 Energy consumption

Inputs and energy consumption (i.e. electricity and diesel fuel) during the receival, sorting, storage and composting of one tonne of compostable organic materials is assumed to be the same for all 3 types of raw materials used to produce composts defined within this study. Whilst there many differences in these materials and the manner in which they are processed (e.g. biosolids do not require size reduction, and biosolids windrows are commonly turned more frequently using a purpose build windrow turner; food organics have a significantly higher moisture content and must be mixed with complementary materials to form a suitable feedstock for composting), the scope of this study only considers windrow composting systems in terms of the composting process. The range of products included here, and the post agricultural application impacts are the key focus of this study, hence these products are included regardless of variations in the composting process via which they are manufactured.

5.6 Emissions

The stockpiling of the clean garden organics is not expected to release significant emissions (biogenic or anthropogenic) or leachate due to the initial low density and low moisture content of the material. By contrast, the stockpiling of unstable food organics may contribute to a number of problems (such as leachate, odour and release of methane) if appropriate measures are not taken (Recycled Organics Unit, 2001). Food organics may pose considerable odour, leachate and greenhouse gas (methane) problems if not properly received and stored by a facility. Compost facilities commonly incorporate raw materials such as biosolids or food organics directly into woodchip 'baths' in order to suppress potential odours.

None of the literature reviewed in a recent greenhouse emissions study (Recycled Organics Unit, 2001) identified stockpiling of feedstock material (prior to size reduction and composting) to be a major contributor to greenhouse gases. This finding is particularly relevant to garden organics, as these feedstocks usually comprise low moisture contents (~50 %) when received by a facility and are of a low bulk density. This open structure and low moisture content helps maintain aerobic conditions during the stockpiling period. In addition, little decomposition occurs until the material is sized reduced, moistened and piled.

The processing of organic materials may contribute to both anthropogenic and biogenic environmental releases. Even though Australian Greenhouse Office (1999) identified the waste management sector as a significant source of GHG emissions in Australia, contributing to 3.5% of net emissions, Australian Greenhouse Office (AGO) considers emissions from the composting process (that is from the composting mass as opposed to equipment operations) as biogenic. Biogenic emissions are excluded from LCA studies (US EPA, 2002; Recycled Organics Unit, 2001b).

The anthropogenic emissions originate from the use of stationary and mobile machinery and electricity. The process includes size reduction, material handling, composting and screening of feedstocks. Machinery such as tractors, bulldozers, windrow turners, shredders, grinders, trommel screens and other screening plant technologies are used. Electricity is used to run facility infrastructure such as offices and stationary equipment (e.g. pumps for leachate collection and aeration). For the purposes of this study, emissions are estimated from the production (e.g. drilling, extraction, transportation and refining) and combustion of fuel used in machinery. Emissions associated with electricity production are estimated from the total amount of electricity used by a facility over a given period (Recycled Organics Unit, 2001b).

Poorly run compost piles may not be turned or agitated enough, leading to the development of anaerobic conditions in the pile. Under such conditions, the rate of composting slows, as organic material break down at a slower rate. Under anaerobic conditions (oxygen <5% v/v), methanogenic bacteria can potentially liberate methane during the decomposition of the organic fraction (Derikx *et al.,* 1986). Although this is possible in poorly managed windrow systems, little evidence exists in the literature to suggest that methane can be produced under such conditions. Reports have suggested that the absence of methane in poorly managed windrow systems are because methanogenic microorganisms are strongly inhibited by ammonia released during the thermophilic phase of the composting process (Miller, 1993; Jackson and Line, 1998). In this study, therefore the methane emissions from commercial composting facilities were not included, as they were considered to be negligible (Recycled Organics Unit, 2001b).

Well-managed compost operations usually do not generate methane because they typically maintain an aerobic environment with appropriate moisture content to encourage aerobic decomposition of the materials. Even if methane was generated in anaerobic pockets in the centre of an infrequently turned compost pile, methane is most likely to be oxidised by microorganisms by the time it reaches the oxygen rich surface of the pile (US EPA, 2002). Furthermore, anaerobic pockets are most likely to develop when too much water is added to the compost pile; however, this problem rarely occurs because compost piles are more likely to be watered too little, rather than too much (Recycled Organics Unit, 2001b).

5.7 LCI data for fuel consumption

Although it is recognized that there are a number of fuel types that could be considered in the present study (e.g. LPG, natural gas, diesel, petrol), only diesel is considered. Diesel is the most commonly used at composting facilities.

Amount of fuel and energy used in each operation in a composting process are given in Table 5.3. The data in the Table 5.3 is computed from the information given in Tables 5.1 and 5.2. For simplicity, this study only considers the amount of fuel used (and related emissions) by machines used in a composting operation. Fuel consumption of machinery (litres) was converted to an equivalent weight (kg) value by multiplying the volume by its density (0.845 kg L^{-1}) (ASTM D-86). The weight of fuel was then used to estimate associated production and consumption emissions using data given in Tables 5.4 and 5.5.

The total amount of diesel used in a commercial facility in USA is given by US EPA (2002) as an average of 221,000 Btu of diesel to process one ton of organic materials. This equates to 5.9026 kg of diesel per one tonne of organic material processed (assuming the density of diesel = 0.845 kg L^{-1}, 43.5 MJ per kg of diesel, ton = 0.9072 tonne, and 1 Btu = 0.001055 MJ). Whilst this figure may provide a useful ball park figure for cross reference, the study does not identify the actual operations that are included in the calculation of this figure, the products produced, the equipment used, the types of compostable organic materials received and processed by the facility.

Table 5.3 Fuel, electricity and water consumption during a composting operation

Operation	Consumption (per tonne of feedstock)		
	Diesel (litres)	Electricity (kW)	Water (litres)
Receival	0.48	0	0
Shredding	3.65	0	0
Composting	0.58	0.13	
Screening	0.79	0	0
Dispatch	0.03	0	0
Total	5.53	0.13	

Life cycle inventory of diesel required to process a tonne of "feedstock" material can be computed from the Tables 5.4 and 5.5. The data in Table 5.4 was obtained from the report on "Life cycle inventory of biodiesel and petroleum diesel for use in an urban bus" published by Sheehan *et al.* (1998). LCI data for diesel transportation is based on national average distances of crude oil and diesel transportation in USA. This includes domestic and foreign transportation via tankers, pipelines, rails and truckers. Life cycle inventory of use of electricity for composting operations could be computed from Table 5.6.

Table 5.4. LCI for diesel fuel production and transportation (for 1 kg of diesel fuel)

Material	Units	Production	Transportation
Raw Materials:			
Coal (in ground)	kg	0.1475	0.00235
Oil (in ground)	kg	0.00426	0.00347
Natural gas (in ground)	kg	0.03938	0.00054
Uranium (U, ore)	kg	3.50E-07	5.63E-08
Perlite (SiO_2, ore)	kg	0.000246	7.71E-07
Limestone ($CaCO_3$, in ground)	kg	0.002782	0.000448
Water use (total)	L	0.000967	0.000541
Air Emissions:			
Carbon dioxide (CO_2, fossil)	g	360.415	18.259
Methane (CH_4)	g	0.30982	0.20819
Nitrous oxide (N_2O)	g	0.00728	0.001252
Carbon monoxide (CO)	g	0.25025	0.039881
Hydrocarbon (except methane)	g	0.00272	0.008311
Hydrocarbon (unspecified)	g	1.05653	0.006885
Benzene	g	8.43E-07	7.61E-07
Formaldehyde	g	1.13E-05	1.02E-05
Particulates (PM 10)	g	0.008464	0.012817
Particulate (unspecified)	g	0.458895	0.034011
Sulphur oxides (SO_x as SO_2)	g	2.554930	0.056308
Nitrogen oxides (NO_x as NO_2)	g	0.753419	0.127928
Hydrogen chloride (HCl)	g	0.007873	0.001268
Hydrogen fluoride (HF)	g	0.000984	0.000159
Ammonia (NH_3)	g	6.27E-08	2.27E-08
Water Emissions:			
BOD5 (bio chemical oxygen demand)	g	0.725045	0.002305
COD (Chemical oxygen demand)	g	6.135	0.019501
Metals (unspecified)	g	0.009006	9.59E-05
Ammonia (NH_4^+, NH_3 as N)	g	0.105987	0.000340
Nitrates (NO_3^-)	g	1.25E-05	2.01E-06
Solid waste (hazardous)	kg	0.002360	7.50E-06
Solid waste (non-hazardous)	kg	0.009212	0.00087
Total primary energy	MJ	2.828260	0.27219
Fossil energy	MJ	2.805900	0.26859
Fuel energy per kg of diesel	MJ	43.5	

Source: Sheehan *et al.* (1998)
Note: Total primary energy is energy contained in all raw materials extracted from the environment and fuel product energy is the energy contained in the final product available to do work in an engine.

Table 5.5. LCI for diesel fuel use (for 1 kg of diesel fuel)

Material	Units	Emission
Carbon dioxide (CO_2)	kg	3.28425
Carbon monoxide (CO)	kg	0.01392
Nitrogen oxides (NO_x)	kg	0.038715
Nitrous oxide (N_2O)	kg	8.265E-05
Particulates (PM10)	kg	1.7835E-03
Methane (CH_4)	kg	1.827E-04
Sulphur dioxide (SO_2)	kg	5.22E-03
Hydrocarbons	kg	3.6975E-03

Source: Sheehan *et al.* (1998) (converted from MJ to kg basis assuming 43.5 MJ/kg of diesel).

Table 5.6 Emissions relating to the generation of 1 MJ of electricity.

Material	Units	Emission
Inputs:		
Hard coal Australia 19,5 MJ [Hard coal (resource)]	kg	2.55E-05
Hard coal Australia 22,4 MJ [Hard coal (resource)]	kg	9.78E-05
Hard coal Australia 24,1 MJ [Hard coal (resource)]	kg	0.147
Lignite Australia 14,1 MJ [Lignite (resource)]	kg	1.35E-05
Lignite Australia 7,9 MJ [Lignite (resource)]	kg	0.00027
Bauxite [Non renewable resources]	kg	7.7E-08
Sodium chloride (rock salt) [Non renewable resources]	kg	3.84E-08
Water [Water]	kg	1.3
Natural gas [Natural gas (resource)]	kg	2.92E-05
Primary energy from hydro power [Renewable energy resources]	MJ	0.0765
Crude oil [Crude oil (resource)]		0.000223
Primary energy from wind power [Renewable energy resources]	MJ	3.45E-05
Air Emissions:		
Carbon dioxide (CO_2)	kg	0.26
Methane (CH_4)	kg	0.000734
Nitrous oxide (N_2O)	kg	2.31E-06
Carbon monoxide (CO)	kg	5.59E-05
Heavy metals (As, Cd, Cr, Cu, Hg, Li, Mn, Mo, Ni, Pb, Se, V, Zn)	kg	7.14E-07
Benzene	kg	1.56E-09
Formaldehyde	kg	1.35E-09
Fly ash	kg	0.021
Particulate (unspecified)	kg	7.1238E-05
Sulphur oxides (SO_x as SO_2)	kg	0.00114
Nitrogen oxides (NO_x as NO_2)	kg	0.00068
Hydrogen chloride (HCl)	kg	1.8396E-07
Fluorine (F)	kg	1.75E-06
Water Emissions:		
Acids (calculated as H^+)	kg	1.22E-09
BOD (bio chemical oxygen demand)	kg	1.1E-06
Chloride (Cl)	kg	5.22E-05
Metals (unspecified)	kg	1.18E-10
Sulphate (SO_4^{-2})	kg	01.89E-04
Nitrates (NO_3^-)	kg	4.52E-07
Heavy metals (Zn)	kg	5.51E-09
Inert chemicals (hazardous waste)	kg	1.14E-10
Mineral waste (consumer waste)	kg	5.13E-08

Source: Grant *et al.*, 1999 and Gabi, 2001

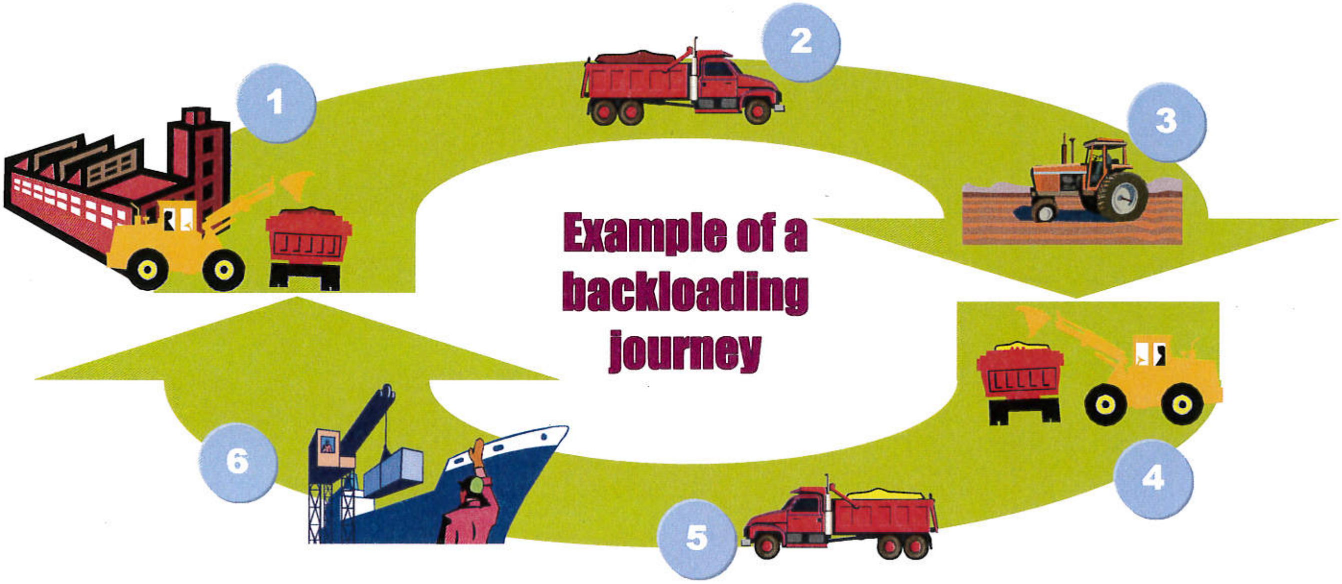
Section 6 LCI – transportation and application of composted products

6.1 Introduction

This component of the model includes the delivery of the end-product from the composting facility to an application site. It also includes the addition of the composted material to soil using mobile machinery such as manure spreaders and tractors. Anthropogenic sources of GHG emissions include those from the production of the fuel used and the combustion of the fuel. Human and eco-toxicity issues may arise from exhaust and production emissions. Distinctions between backloading journeys and discrete journeys to application sites are considered in this study. Where backloading is used, the impact per nominated kilometre should reduce, as there is more efficient haulage. A typical backloading journey is described in Figure 6.1.

Emissions arising from end-product application are dependent upon the type and size of machinery used and also the application rates to the land.

Figure 6.1 Example of a backloading journey. 1. Mulch from a composting facility (located on the fringe of a metropolitan area) is loaded onto an empty truck. 2. The loaded truck transports the mulch to an agricultural enterprise (e.g. vineyard). 3. The mulch is then used by the agricultural enterprise. 4. Agricultural produce is loaded onto the (cleaned) empty truck. 5. The loaded truck transports the agricultural produce to shipping ports in the city (6). The truck returns to the composting facility (1). In a non-backloading journey, the truck would return empty (from the agricultural enterprise) to the composting facility.



6.2 LCI

LCI data from the transport of end-products was based on vehicle type and distance travelled. For the purposes of this study, a Kenworth T601/T604 truck (32 tonnes ≅ 40 m^3 per load) is used to estimate emissions from the transportation of composted end-products to application sites. Consideration was made for transport distance on urban roads (i.e. from composting facility in Sydney to periphery of Sydney metropolitan area) and rural roads. About 20 km transport distance has been considered on urban roads to transport composted products out of Sydney metropolitan area from composting facility. The fuel consumption of this truck under different road conditions is given in Table 6.1.

Table 6.1 Diesel consumption of Kenworth T601/T604 truck under different road conditions.

Road type	Loaded (kg km^{-1})	Unloaded (kg km^{-1})
Urban	0.19 (4.5 km L^{-1})	0.15 (5.5 km L^{-1})
Highway/Rural	0.13 (6.5 km L^{-1})	0.11 (7.5 km L^{-1})

For the purpose of this investigation two different sets of machineries are considered for the application of composted soil conditioner and composted mulch (Table 6.2). Consideration was also made for backloading (truck empty 20% of time) and non-backloading journeys (truck empty 40% of time). The LCI data for total diesel requirement for transport and application of one tonne material is then estimated from the LCI data given for the production and use of diesel in Tables 5.3 and 5.4. Please note diesel consumed for backloading journeys has not been included in environmental impact calculations because that diesel consumption is not related to composting systems. However diesel consumed for non-backloading journeys has been included in GHG emission calculations.

Table 6.2 Types of machinery used for spreading and respective handling rates.

	Loader		Spreader		Handling rate
	Type	Engine capacity/fuel consumption hp/(L/hr)	Type	Engine capacity/fuel consumption* hp/(L/hr)	t/day (8-10 hours)
[a]Soil conditioner	Case 580K	120/(13.5)	Mercedes 1700 Unimog	168/(10)	200 (with 100% loader and spreader use)
[b]Mulch	Tractor with a loading bucket	50/(8.3)	Seymour vineyarder with a tractor	50/(8.3)	225 (with 100% loader and spreader use)

Source: personal communications, [a]Specialised Spreading Services, Rouse Hill, Sydney; [b]Seymour Rural Equipment, Seymour, Victoria. *Please note fuel consumption data has been obtained from respective companies.

Section 7 LCI – post-application effects of adding compost to soil

7.1 Impacts and avoided impacts of using composted products in an agricultural context

The application to soil of mature composted products that comply with Australian Standard AS 4454 (2003) may result in:

- Reduced water use,
- Reduced fertiliser, herbicide, pesticide, lime and gypsum requirements (avoided greenhouse gas emissions from production and transportation),
- Improved soil properties and associated plant growth,
- Sequestration of carbon in soil,
- Reduced soil management and maintenance, and
- Reduced environmental impacts due to reduced soil erosion, waterlogging, nutrient loss, surface crusting, eutrophication of waterways, siltation of waterways, etc.

Although there are many advantages associated with the use of composted products that comply with Australian Standard AS 4454 (2003), there are potential risks related to the use of non-compliant, poor quality and/or immature pasteurised garden organics-derived products. Some problems can also occur with inappropriate product application. Associated risks include:

- Phytotoxicity,
- Nitrogen drawdown,
- Chemical or physical contamination,
- Odours,
- Freeze injury or frost damage,
- Weed propagation, and
- Transfer of pathogens to plants and animals.

Each risk is defined in the Glossary. These risks are negated substantially if products are compliant with Australian Standard AS 4454 (2003) and if the correct products are selected for application. For greater details regarding the benefits and risks of composted products, the reader is directed to "*Buyers Guide for Recycled Organics Products*" (Recycled Organics Unit, 2001a). It should be noted that the current study addresses mature composted products that comply with Australian Standard AS 4454 (2003) and consequently the aforementioned risks are not relevant. The Environmental Protection Authority of NSW (EPA, 1997a) also regulates the maximum permissible concentration of heavy metals and other compounds such as pesticide residuals to ensure products that are applied pose no risk to human and ecosystems health. Therefore, this study does not consider such issues in the context of LCA for compost application.

7.2 Reduced water use

7.2.1 Introduction

Water availability is generally the most important natural limiting factor for the agricultural sector in Australia. Ineffective and inappropriate use of water in this sector can result in environmental problems of national significance. The agricultural sector is the largest net user of water. In 1996-97 the agriculture sector accounted for the majority of net water consumption with 8,795 gigalitres (GL) (GL= 1000 megalitres) used by livestock, pasture, grains and 6,726 GL by other agriculture, totalling 15,503 GL which comprised 70% of total net water use in Australia (ABS, 2000).

Significant net water consumption occurs within NSW-ACT out of which ACT accounts for approximately 1% of the total NSW-ACT (ABS, 2000). Net water consumption in different agricultural sectors in NSW-ACT is presented in Table 7.1. Most of the water supply comes from the natural river system. However, this supply is extremely variable between seasons and between years as river flows are highly erratic depending on the rainfall. Ground water also plays an important role by supplying an average of one million megalitres (ML) (ML= 1,000,000 L) each year on a reliable basis (DLWC, 2000d). NSW also accounts for almost one third of Australia's agricultural water use. Irrigation is clearly the largest class of water use in NSW in terms of volume. Nearly 8% of the land use for crop and pasture production in NSW is under irrigation (Table 7.2). In recent years, the annual value of irrigated production in Australia was around $8 billion. Of this NSW contribution was around $2 billion (DLWC, 2000d).

Table 7.1 Net water consumption in different agricultural sectors in NSW-ACT in 1996-97.

Sector	**Water use (ML)**
Livestock, pasture, grains and other agriculture	3,404,610
Vegetables	194,044
Fruit	279,063
Grapevines	242,288
Cotton	1,417,452
Rice	1,643,306
Total for NSW (including water use in other sectors and excluding in-stream use)	8,715,663
Total for Australia (including water use in other sectors and excluding in-stream use)	22,185,731

Source: ABS (2000)

NSW agricultural industry receives an average return of only $290 per ML compared with the $680 per ML realised in the remaining Australian States, because a greater proportion of water in NSW is used less efficiently and on lower value crops. Therefore, increased efficiency of irrigation water use is seen as a way to balance competing demand for the NSW water resources, which are fully, or in many instances, over committed. In addition, irrigation, as the major user of freshwater resources, has had a detrimental impact on inland waters (reduced environmental flows, eutrophication, turbidity and salinity). Increased irrigation efficiency is expected to reduce the extent of those off-site impacts.

Rainfall distribution in NSW also emphasises the need to conserve water. About 75% of the State's total rainfall occurs in costal catchments compared to 25% in the inland catchments, while 80% of the extraction of water for irrigation occurs in inland rivers of NSW (DLWC, 1998).

The majority of agriculture (about 90%) in NSW is dryland agriculture, which is mainly dependent on the availability of rainfall. This dryland sector includes crops, pasture and grasses as indicated in Table 7.2. However, rainfall distribution in NSW is highly variable and unpredictable. The unpredictability of rainfall has already costed billions of dollars in 2002 due to prolong drought conditions in NSW. Therefore, soil management techniques, which improve the water conservation and water use efficiency, are vital in sustaining dryland agriculture in NSW.

There are significant opportunities for water conservation in all agricultural sectors in NSW. Measures that can reduce demand of water in agricultural sector include efficient application and control of irrigation water use, reducing losses in distribution and reticulating systems, conserving soil moisture and maximising effluent reuse. Although the potential for water use efficiency in NSW Agriculture sector is very high, it varies considerably depending on the crop and the location.

Table 7.2 Area of crops and pasture irrigated in NSW in year 2000.

Sector	Area ('000 ha)
Pasture (annual or perennial)	265
Vegetables	14
Fruit	23
Grapevines	25 (15.194)*
Cotton	268 (249.1)*
Rice	127
Other cereals	165
All other crops	36
Total area irrigated	944
Total land use for crops, pastures and grasses (irrigated + dryland)	12,511

Source: ABS (2001) * Figures in the brackets are for 1996-97 (ABS, 1996)

Studies have indicated that application of composted products can enhance the water use efficiency by improving infiltration and storage in the root zone and reducing deep drainage, run-off, and evaporation, and water use by weeds. The beneficial effects of compost application arise from improvements in soil physical and chemical properties (Bresson *et al.*, 2001; Albaladejo *et al.*, 2000; Movahedi and Cook, 2000; Wahid *et al.*, 1998; Stocking and Albaladejo, 1994; Shiralipour *et al.* 1992; Agassi *et al.*, 1990).

When used in sufficient quantities, the addition of composted soil conditioner has both an immediate and long-term impact on soil structure. Australian soils are generally low in organic matter and poorly structured. As organic matter decays to humus, the humus molecules binds mineral components of the soil (such as particles of sand, silt, and clay) and organic matter into water stable aggregates and improves soil porosity and soil structure (NSW Agriculture, 2002). Due to the aggregate stability and improvements in soil structure, the application of composted soil conditioner reduces surface sealing, improves infiltration and the water holding capacity thus reducing runoff generation. These aggregates are also effective in holding moisture for use by plants. In addition, humus molecules can absorb and hold large quantities of water. Therefore, the addition of composted soil conditioner may provide greater drought resistance and more efficient water use. Thereby, the frequency and intensity of irrigation may be significantly reduced in irrigated agriculture and prolonged cropping is possible in dryland crop production. Increased porosity and decreased soil compaction may also result in increased root penetration, resulting in deeper and more elaborate root systems to explore a larger soil mass for moisture and nutrients.

Increased root exploration and water holding capacity can also reduce deep drainage below the root zone, resulting in reduced nitrate leaching and lowering groundwater tables. Low nitrate leaching can reduce eutrophication of water resources, while deeper ground water tables can lead to reduced salinity problems as discussed in other sections of this report (see Sections 7.3 and 7.7).

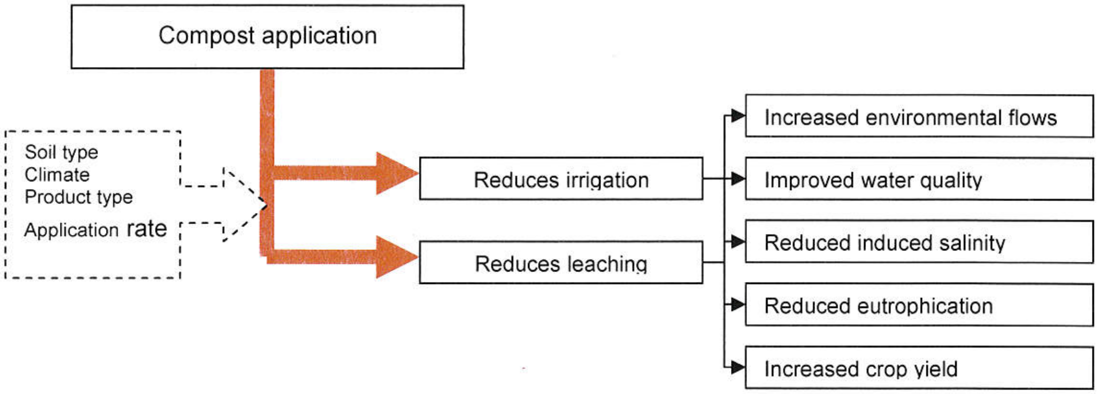
Use of composted mulch in cropping lands can also significantly increase the water use efficiency by lowering the evaporation losses from soil surface. Mulching reduces radiation and wind speed at the surface and hence, reduces the water evaporation from soil surface. Many researchers have reported that surface application of mulch has resulted in reduction of between 30–70% of irrigation water required by crops due to the reduction of water evaporation from soil surface (Abu-Awwad, 1999; Abu-Awwad, 1998; Mahrer *et al.,* 1984). Valzano (2000) showed that water availability was on average 10% higher in agricultural soils with a surface mulch than in equivalent exposed soils. Such effects on soils are also applicable in urban situations (e.g. parks and gardens). Buckerfield and Webster (1995) showed that the surface application of organic mulches to vineyard soils resulted in a 34% increase in soil moisture content and an increase in grape yield. This reduced evaporation offers obvious benefits for both dryland and irrigated agriculture.

7.2.2 Model for water use efficiency

The potential benefits of compost application on water use efficiency vary depending on soil type, climate, product type, application rate and crop parameters. Figure 7.1 depicts the sub-model showing the impacts resulting from compost application.

The compost application increases water holding capacity, resulting in a reduction in leaching and irrigation water requirements. As a consequence, water extractions from natural waterways will be reduced leading to increased water flows and improved water quality in natural river systems. This has a significant impact on ecosystems associated with the natural river systems in NSW. Moreover, reduced leaching will result in reduction in eutrophication and induced salinity. In addition, increased water holding capacity of soil can reduce stresses due to soil moisture deficits leading to higher crop yields.

Figure 7.1 Sub-system model for impacts on water use efficiency.



7.2.3 Soil variation

Soil physical, chemical and biological properties vary greatly among soil types. Soil types are grouped and categorised according to their physical, chemical, morphological and biological properties. The Great soil group concept (Stace et al, 1968) and the Australian soil classification system (Isbell, 1996) are the two common soil classification systems used in Australia. In these classifications, a number of properties of a soil are taken into consideration when it is placed in a relevant soil group. However, it

is necessary to distinguish important characters, which are useful in evaluating the impact of compost application to avoid needless complications. In this context, soil texture and profile drainage of a soil type could be used to evaluate the impact of compost application on agricultural lands as they reflect the soil properties accountable for water retention in soils related to agricultural production.
Among the soil categories, the impact of compost application on water conservation may be more pronounced in sandy soils with well drained profile characteristics as compost can reduce deep percolation by increasing the water holding capacity of the soil. Table 7.3 shows common soil types found in NSW with their respective textural classes of the top layer and the profile drainage.

Table 7.3 Common soil types found in NSW and their respective drainage categories.

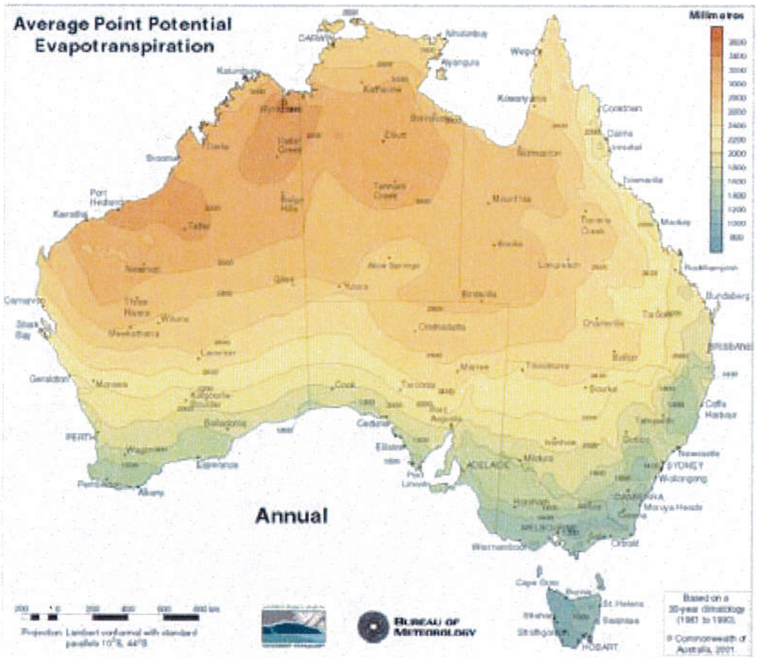
Great soil groups terminology[1]	Australian soil classification[2]	Profile drainage	Soil texture (0-30 cm layer)
Black earths, grey, brown and red clays	Vertosols	Poor to imperfect	Clayey
Red Brown Earth	Chromosols	Moderate	Sandy loam to clay loam
Red Earth	Kandosols	Well	Sandy loam to clay loam
Yellow earth	Kandosols	Moderate	Sandy to sandy clay
Alluvial	Rudosols/Tenosols	Free to well	Loamy sand to sandy loam
Red Podzolics	Chromosols/Kurosols	Well	Fine sand to sandy loam
Yellow Podzolics	Chromosols	Moderate to poor	Loamy sand to clayey sand
Solonised Solonetzic & solodic soils	Sodosols	Moderate to poor	Loamy sand to silty clay
Euchrozems and Krasnozems	Ferrosols	Free to well	Clay loam to clay

Source: 1- Stace et al., 1968, 2 – Isbell, 1996.

7.2.4 Climatic variation

Water evaporation from soil surfaces accounts for major losses of water from soils, especially in the drier climatic regions. Soil evaporation accounts for almost half of the water lost from a soil-plant system and contributes to a large part of evapotranspiration. The potential evapotranspiration rate of a given area indicates the soil evaporation potential of the area and also reflects the climatic parameters, such as temperature, relative humidity, rainfall, wind velocity and solar radiation, which regulate the soil moisture budget. The reduction of soil evaporation resulting from the application of compost mainly as a mulch should be more beneficial in climatic zones with higher potential soil evaporation. Annual potential evapotranspiration rates in NSW vary from 800 mm to 2400 mm from region to region. A map showing different climatic zones based on average annual potential evapotranspiration rates is given in Figure 7.2.

Figure 7.2 Average potential ET across Australia.



7.2.5 Other variations

In addition, the magnitude of the impacts of compost application could be varied depending on the type of crop and the source of water (irrigation or rainfall). These impacts are dependent on their effects on the components of the soil moisture budget. The relevant components of the soil moisture budgets are:

- Run-off,
- Evaporation from the soil surface,
- Drainage from the root zone,
- Storage in the soil,
- Weed water use, and
- Crop water use.

Application of composted mulch can reduce the runoff, evaporation from the soil surface, suppress the weed growth and reduce the associated weed water use. Application of composted soil conditioner can improve the infiltration rate, soil water storage and reduce both runoff and drainage. These effects are also dependent on the type of crop. The effect of a surface mulch may be less expressed under a well established perennial crop with a good ground cover, whereas the effect of incorporation of composted soil conditioner may be less pronounced in crops with extensive and fibrous root systems. The impacts of applying composted products may also differ depending on the cropping system. A cropping system which practices zero tillage and stubble savings may be less benefited by compost application compared to a system managed under more intensive cultivation. Under irrigated conditions, adjusting the intensity and frequency of irrigation according to soil conditions and crop requirements can effectively reduce the unproductive components of soil moisture budget. However, in dryland agriculture, these controls are limited due to the unpredictable nature of rainfall. In this context, application of compost under dryland conditions can contribute to a better soil moisture budget by minimising soil water losses via surface evaporation.

7.2.6 Literature review

Literature relating to the application of a range of composted mulches and soil conditioners has been reviewed to inform the development of LCI data. Tables 7.4 and 7.5 provide performance data that can form the basis for LCI for water use under different contexts relevant to the application of composted mulch and soil conditioner. However, this literature review excluded the application of composted soil conditioners beyond 100 t ha^{-1} range as high application rates may cause harmful effects to the environment by increased nitrate leaching and heavy metal contamination.

Table 7.4 Summary of impacts of application of composted products as a soil conditioner on water use.

Study	Description	Impact
Hortenstine and Rothwell (1973)	Investigated the effect of application of compost at 0, 16, 32, 64, 128 tonne ha^{-1} (t ha^{-1}) on water retention under irrigated sorghum in a sandy soil.	Percentage of available water (gravimetric water measured at 0.1 and 15 bar) was increased from 1.63% of control to 1.64, 1.71, 2.05 and 2.45% by the addition of 16, 32, 64 and 128 t ha^{-1} respectively.
Epstein (1975)	Evaluated the effect of application of digested sludge at 50 t ha^{-1} on moisture retention in a silty loam soil.	The gain in soil moisture budget was 1.8%.
Wang (1977)	Studied the effect of application of composted urban organic materials with sewage sludge at 20, 40, 125 & 495 t ha^{-1} on soil moisture budget in a clayey soil.	Soil moisture content was increased from 18.3% to 18.8%, 19.8%, 20.3% and 25.6% by addition of 20, 40, 125 and 495 t ha^{-1} respectively.
Sabrah *et al.* (1995)	Investigated the effect of applying urban compost at 16.5, 33, 49.5 and 66 t ha^{-1} on soil moisture under irrigated wheat grown in a sandy soil.	Plant available water content was increased by 1.7%, 3.1%, 3.6%, and 4.4% by 16.5, 33, 49.5 & 66 t ha^{-1} treatments respectively.
California Integrated Waste Management Board (1997)	Investigated the effect of adding composted grass clippings at 3.5 to 3.9 t ha^{-1} (dry wt) on soil moisture under irrigated corn/wheat grown in a sandy loam soil.	No significant effect on soil moisture savings.
Illera *et al.* (1999)	Studied the effect of compost derived from organic fraction of MSW added at 80 t ha^{-1} (dry wt) on soil moisture retention.	Soil moisture content (gravimetric) was increased by 3.9%.
Mamo *et al.* (2000)	Evaluated the effect of urban compost application at 90 t ha^{-1} (dry wt) on soil moisture under irrigated corn grown in a loamy sand.	Water holding capacity (volumetric) was increased by 4.4%.
Movahedi Naeini and Cook (2000)	Investigated the effect of urban compost addition at 50 t ha^{-1} on soil moisture under rainfed maize.	Increased available soil moisture content (measured at 0 to 1500 KPa) by 2.0%.
Blair (2001)	Investigated the effect of applying manure at 45 & 60 dry matter t ha^{-1} for two yrs on soil moisture under rainfed sorghum/triticale grown in a clayey soil.	Available soil water content was increased by 1.9 & 3.8% at application rates of 45 and 60 t ha^{-1} respectively.

Table 7.5 Summary of impacts of application of composted products as surface mulches on water use.

Study	Description	Impact
Shearman *et al.* (1979)	Investigated the effect of mulching at 7.5 and 15 cm with composted alfalfa hay on soil moisture under bush beans grown in a clay loam soil.	Soil moisture content (gravimetric water content) of 20 cm depth was increased by 9.9 and 11.8% with the application of 7.5 and 15 cm mulch cover respectively.
Buckerfield (1998)	Studied the effect of mulching at 1, 5, and 15 cm with composted garden organics on soil moisture under irrigated grapevine.	Soil moisture content at 10cm depth was increased by 3.4, 6.3, & 8.1% with the application of 1, 5 and 15 cm mulch respectively over the control treatment.
Buckerfield (1998)	Investigated the effect of mulching at 1, 5, and 15 cm with composted garden organics on soil moisture under irrigated pears.	Soil moisture content at 10 cm depth was increased by 3.2, 5.6, and 6.4% with the application of 1, 5 and 15 cm treatments respectively.
Buckerfield (1998)	Investigated the effect mulching at 2, 5, 5.5 & 7cm with composted garden organics on soil moisture under irrigated cherries.	Soil moisture content at 10 cm depth was increased by 0.9, 2.9, 4.2 & 4.5 % by 2, 5, 5.5 & 7cm treatments respectively.
Buckerfield (1998)	Evaluated the effect of mulching at 1, 5, and 15 cm with composted garden organics on soil moisture under irrigated orange.	Soil moisture content at 10 cm depth was increased by 0.5, 1.5, and 3.2% with the application of 1, 5 and 15 cm treatments respectively.
Pinamonti (1998)	Studied the effect of mulching at 0.5 cm with compost from organic fraction of MSW on soil moisture under irrigated grapevine in a sandy soil.	Increased soil water retention (measured at -0.33 bar and 15 bar) by 0.6% after 4 months due to mulching.
Pickering *et al.* (1998)	Investigated the effect of mulching at 5 and 10 cm with green waste compost and graded bark chips on soil moisture in a sandy clay loam.	Increased soil moisture content by 8.7, 10.4, 7.5 & 8.7% by 5 cm green compost, 10 cm green compost, 5 cm bark chips and 10 cm bark chips treatments respectively.

7.2.7 LCI data

There is a general lack of data for each application scenario relating to the application of composts at varying rates and varying soil and climatic contexts. The majority of studies that have measured changes in soil moisture content (gravimetric or volumetric water content present in the soil) or soil water retention (relationship between the soil water content and soil-matric potential) after addition of composted products have shown increased soil moisture content or soil water retention (Tables 7.4 & 7.5). Some investigators suggested that the increase in soil moisture retention or soil moisture content in soils due to the application of composted soil conditioner does not necessarily result in an increase in plant available water (Mamo *et al.*, 2000; Turner *et al.*, 1994; Khaleel *et al.*, 1981) i.e. water available for extraction by plants. It is the increase in soil moisture that can be extracted by plants or so called plant available water (PAW), which provides the opportunity for quantification of the reduction in irrigation water requirements.

Plant available water is the soil moisture content retained at soil-matric potential between field capacity and permanent wilting point. The studies reviewed for this report have not evaluated the relationship between increased soil moisture content and plant available water, which is important in terms of the potential for reduced irrigation water usage. A few studies have shown composted soil conditioners increased plant available soil water content (Blair, 2001; Movahedi and Cook, 2000; Sabrah, 1995).

The available data in the literature have been grouped to obtain relationships between the application rates (or the depth and width of mulch cover) and soil moisture changes (Figures 7.3 & 7.4) as the compost application rate is the most influential variable on changes on soil moisture content. Please note in the case of composted soil conditioners, studies that have shown increases in plant available water have been used to graph a relationship between application rates and soil moisture availability (Figure 7.3) as it provides an indication in the reduction in irrigation water requirements. However due to lack such data for composted mulches, most relevant data i.e. increase in soil moisture content was used to graph this relationship.

In Figures 7.3 & 7.4, changes in the moisture content in the top soil layer (0-15 cm) is considered since the data obtained to draw up the relationships were mostly from previous studies focused on the changes in that soil layer. In addition, it is the most important soil layer for agricultural crop production.

Figure 7.3 Effect of compost application as a soil conditioner on soil moisture content of 0 – 15 cm layer.

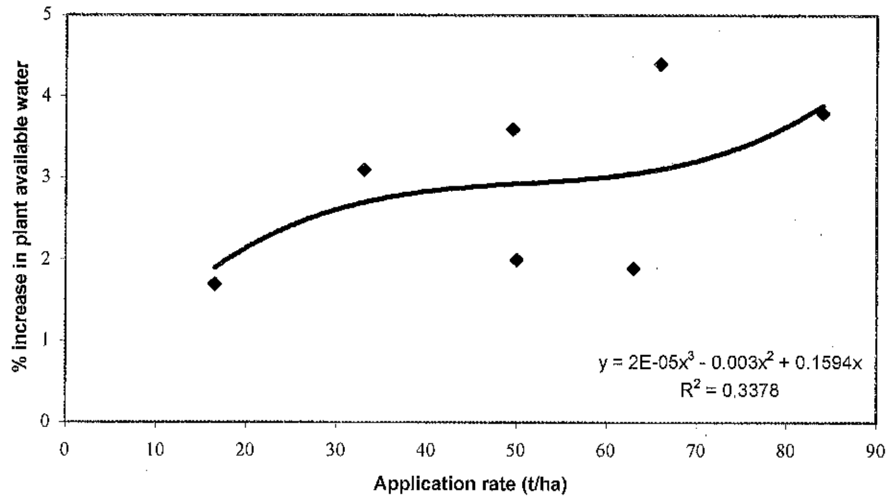
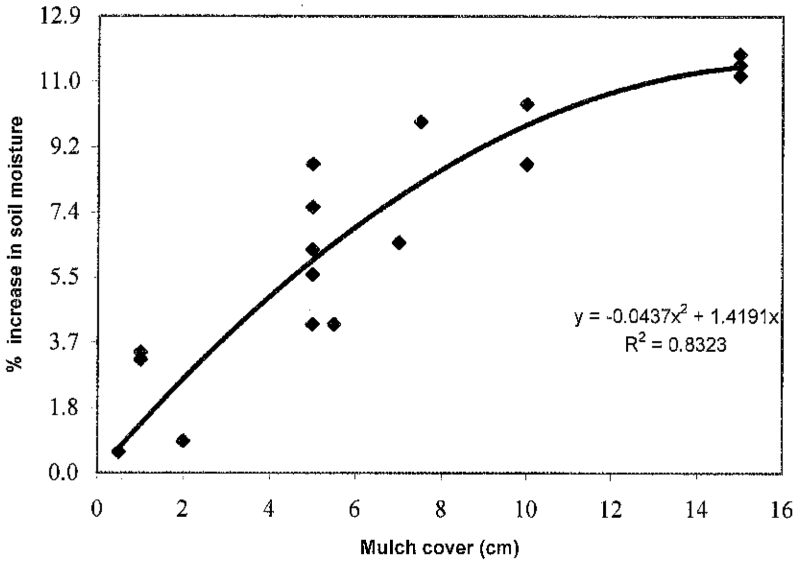


Figure 7.4 Effect of composted mulch on soil moisture content of 0–15 cm layer.



7.2.8 Application scenario

Please note data on net water consumption and total area occupied by cotton and grapevines for the period of 1996-97 have been used to calculate water use for irrigation on one hectare of cotton and grapevine, as net water consumption data for 2000 was not available. However, total water savings for cotton and viticulture industry have been based on total area occupied by these industries for the period of 2000.

Net water consumption for cotton production in NSW in 1996-97 was 1,417,452 ML and total area under irrigated cotton during this period was 249,100 ha (ABS, 1996) suggesting about 5.7 ML (1,417,452 ML /249,100 ha) of water was used for irrigation on one hectare of cotton. According to the relationship shown in Figure 7.3, an application of a composted soil conditioner at 12 t ha^{-1} would expect to increase the plant available water of 0-15 cm layer by approximately 1.5%. This may result in savings of approximately 85,500 L (5.7 ML*1.5%) of water per hectare in a cotton crop per season in irrigated cotton. This water saving for the total cotton industry occupying a total area of 268,000 ha in NSW (ABS, 2001) may result in savings about 22,914 ML (268,000 ha*85500 L) per year.

Net water consumption in grapevine production in NSW in 1996-97 was 242,288 ML and the total area under irrigated grapevines during this period was 15,194 ha (ABS, 1996) suggesting about 9.69 ML (242,288 ML /15,194 ha) of water was used for irrigation of one hectare of grapevines. According to the relationship shown in Figure 7.4, an application of a 10 cm composted mulch cover would result in an increase in moisture retention of 0–15 cm topsoil layer by approximately 9.82%. This could result in savings of 951,558 L or 0.95 ML (9.69 ML*9.82%) of water per hectare under irrigated viticulture, if this water is available for plant use. This water saving for the viticulture industry occupying a total area of 25,000 ha in NSW (ABS, 2001) may result in savings of about 23,750 ML (25,000 ha*0.95 ML).

7.3 Reduced fertiliser requirements

7.3.1 Introduction

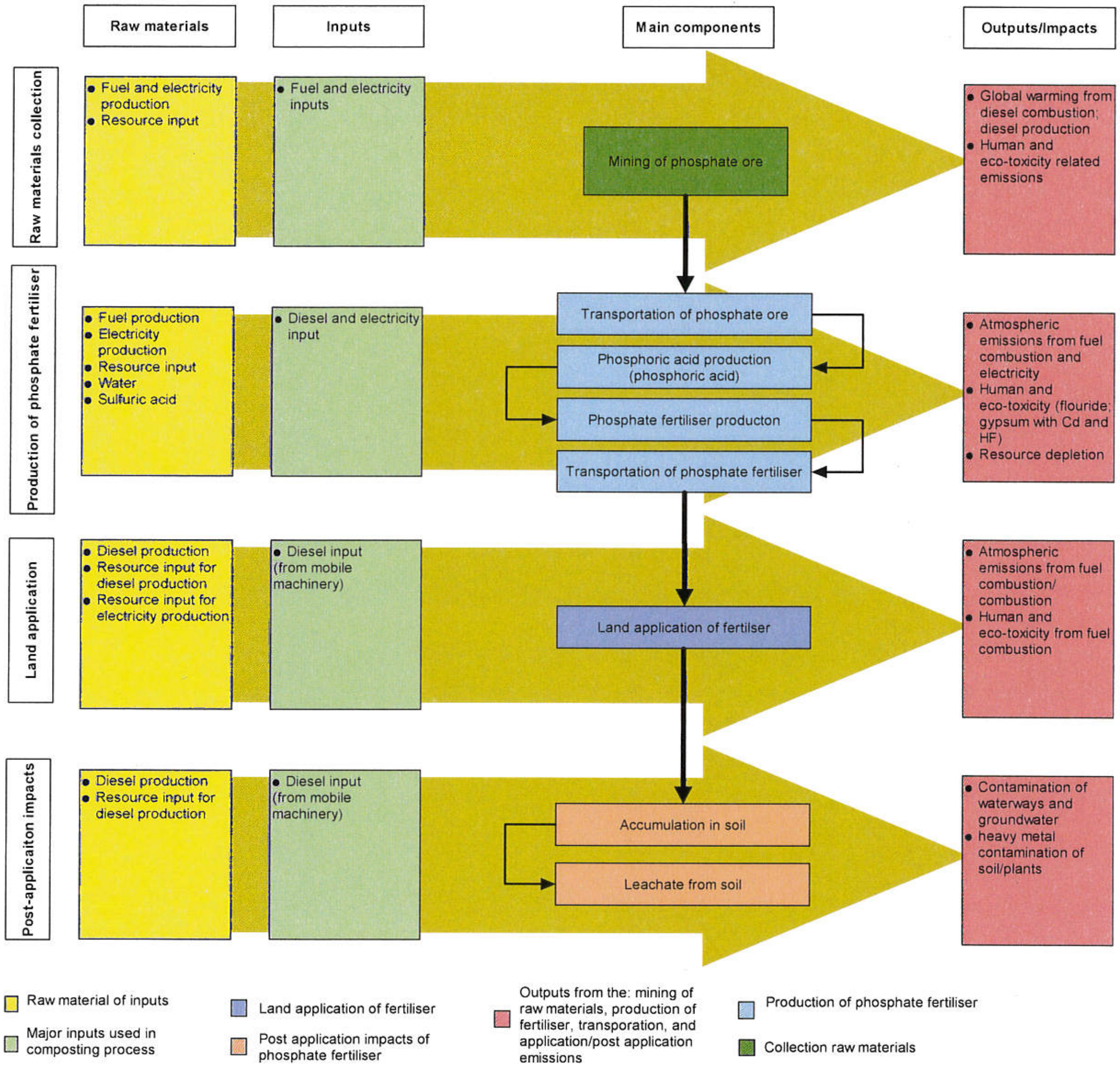
Production and use of inorganic fertiliser may result in significant environmental impacts such as resource depletion, greenhouse gas emissions, human and eco-toxicity and eutrophication. The model in Figure 7.5 on fertiliser production depicts some of the environmental impacts resulting from the production process of phosphate fertiliser. Therefore, returning crop residues or adding compost to soils can significantly reduce the adverse environmental impacts resulting from the use of inorganic fertilisers, as composted products contain many nutrients necessary for plant growth and development. Use of composted products in agricultural production therefore can reduce the fertiliser cost and the environmental impacts associated with fertiliser production and its use.

Hill (1984) found that annual amendment of leaf compost reduced the fertiliser needs of vegetable crops by 33–66% of the normal rates. However, the total replacement of inorganic fertiliser by composted products may not be practical due to low nutrient content and slow mineralization potentials of macro nutrients in composts (Sikora and Enkiri, 2001). In addition, high compost application rates have the potential to cause secondary effects such as excess salts and accumulation of heavy metals and other toxic compounds that may develop with 100% replacement of inorganic fertiliser with compost (Leita and DeNobile, 1991; Sims *et al.*, 1994). Combining low amendment rates of composts with sufficient fertiliser to meet crop requirements also reduces the amount of inorganic fertiliser that potentially could leach past the root zone into ground water.

Although high doses of urban compost have sometimes been claimed to be more efficient than normal mineral fertiliser (Gallardo-Lara and Nogales, 1987), the literature also shows that the combined use of composts and mineral fertilisers may be more efficient than when they are used separately (Gallardo-Lara and Nogales, 1987; Buchanan and Gliessman, 1992). The authors consider this strategy is very promising for most urban composts, because of problems that can arise from heavy compost applications. However, combining low amendment rates of compost with fertiliser may not improve the fertiliser use efficiency of inorganic fertilisers (Sullivan *et al.*, 2002; Sikora and Enkiri, 2001; Sullivan *et al.*, 1997).

Nutrient availability of composted products is extremely variable depending on the type and composition of the compost used, and the soil and climatic conditions. The release of mineral N from composts, or any organic material in soil, is the balance between the amounts of N released through microbial decomposition of the compost driven by the microorganisms need for carbon, and the amount of N assimilated by the microbes that feed on compost (Hadas and Portnoy, 1997). Therefore, the carbon content and the C:N ratio of composts are indeed important properties for assessing their ability to decompose and release available N (Hadas and Portnoy, 1997).

Figure 7.5 Model for the manufacture of a phosphorus-based fertiliser.



About 5 – 25% of the total compost N becomes plant-available during the first year after application of compost having C:N ratios <20:1 (Dick and McCoy, 1993; Tester, 1989). Compost with C:N ratio greater than 30:1 may reduce crop productivity via microbial immobilization of available N during the first year after application (Shiralipour *et al.*, 1992). However, composts may provide long-term benefits because of slow releasing N potential. Long term monitoring of compost-amended soils showed that the half-life of soil organic N supplied by sewage sludge or farmyard manure was approximately 12 years (Sullivan *et al.*, 1998).

Increased application of compost has a tendency to increase the amount of NO_3^- leaching and thus poses an environmental threat to ground water quality. Ground water is a major source of drinking water. According to the drinking water quality standard, NO_3-N level higher than 10 mg L^{-1} is considered not suitable for drinking water (NHMRC & ARMCANZ, 1996). Nitrate leaching has been reported after excessive applications of inorganic fertilizers (Lea-Cox and Syvertsen, 1996; Goodrich *et al.*, 1991). Land application of sewage sludge and manure has also been reported to cause leaching of NO_3-N into the ground water (Daliparthy *et al.*, 1995).

7.3.2 Literature review

Table 7.6 provides performance data collected from the literature relating to the effect of application of a range of composted products on nutrient availability, fertiliser use efficiency and nitrate leaching under various soil and climatic conditions.

Table 7.6 Summary of impacts of application of composted products on nutrient availability.

Study	Description	Impact
Terman *et al.* (1972)	Compared commercial fertiliser with urban compost on bioavailability of N.	16% of the N in urban compost is bioavailable.
Tester *et al.* (1977)	Examined the mineralization rates of sewage sludge compost with a C:N ratio of 17 in a green house study.	10% of the organic N in the compost is available to plants during the first year.
Parker and Sommers (1983)	Studied the mineralization of N in sewage sludge compost.	<20% of the total nitrogen in compost is available.
Avnimelech and Kochva (1992)	Carried out series of experiments to compare the effect of superficial application and incorporation of urban compost in a clay loam soil.	Superficial application of compost gave better yields, higher soil N and carbon contents compared to the incorporation. 20-40 m^3ha^{-1} was found to be the optimal rate of urban compost.
Maynard (1993)	Studied the nitrate leaching from application of spent mushroom compost (C:N = 17.2) in fine sandy loam soils.	Annual compost application containing 365 and 731 kg N ha^{-1} made for 3 consecutive years, had little effect on the NO_3^- concentration in the ground water.
Hue *et al.* 1994, Pierzinski *et al.* 1994, Dick and McCoy (1993).		45-75% of total P in composted waste materials is inorganic P and plant available.
Murillo *et al.* (1995)	The response of ryegrass to low-quality urban compost (applied at 12 and 48 t ha^{-1}) was evaluated in a sandy clay loam soil.	22% of the total N is mineralizable.
Baldoni *et al.* (1996)	Investigated the influence of compost (sludge and straw) and sludge on agricultural crops.	17 – 26% of total N in the compost and 22 – 30% of total N in the sludge were plant available.
Cortellini *et al.* (1996)	Studied the P availability in soil after the application of 7.5 – 15 t ha^{-1} (dry wt) of compost for 6 consecutive years.	50% increase in plant available P content of the soil over the mineral fertiliser treatments.
Hadas and Portnoy (1997)	Municipal waste compost with C:N ratio of 15.5 was evaluated for its N mineralization during 33 weeks.	The N recovery after 33 weeks was 22% of total N and 15% of the organic N. The rate of recovery was independent of the soil and compost application rate and only dependent on C:N ratio.
Li *et al.* (1997)	Evaluated the leaching of nutrients from various urban composts applied at 100 t ha^{-1} to a sandy soil.	The amount of N and P leached accounted for 3.3 – 15.8% of total N and 0.2– 2.8% of total P in the compost.

Table 7.6 Continued Summary of impacts of application of composted products on nutrient availability.

Study	Description	Impact
Sullivan *et al.* (1997)	Fertilizer N uptake efficiency was evaluated with and without compost over five fertilizer N application rates ranging from 0 – 67 kg N. ha^{-1}.	Fertilizer N uptake efficiency was not increased due the application of compost.
Biala and Wynen (1998)	Reviewed the nutrient availability of composts.	Nutrient availability of garden organics compost: N: 15% (year 1), 25-35% (year 2) P: 50% (year 1), 50% (year 2) K: 80% (year 1), 20% (year 2) Mg: 30% (year 1), 70% (year 2)
Sikora (1998)	Studied the N availability of different rates of composted municipal refuse combined with inorganic fertiliser to supplement the total N requirement of tall fescue.	N availability of compost was approximately 10% of its original N content.
Sullivan *et al.* (1998)	Conducted a field experiment with a perennial grass to evaluate N replacement value of food waste compost addition at 148 – 172 t ha^{-1}.	Fertiliser N equivalent of compost was considered to be 7.6 – 8.1% of total compost nitrogen and no significant effect of adding compost on NO_3-N concentration of lower soil layers.
Grey and Henry (1999)	Conducted a laboratory study to evaluate leaching of nutrients in urban compost treated soils at different rate of inorganic fertiliser additions.	Observed that leaching of mineral P was insignificant suggesting strong retention of P by composts. However, K lost from compost was substantial.
Mamo et al. (1999)	Investigated the N mineralization potential of urban compost applied on to field grown corn in loamy sand soil.	0 – 12% of the compost N was mineralised during the first year.
Sikora and Enkiri (2001)	Evaluated the effects of biosolids compost at different rates blended with different rates of urea and ammonium nitrate on tall fescue growth and N uptake.	25% of the total N present in the compost was mineralised after 60 days and no significant effect on fertiliser use efficiency by blending of compost at low rates (2 – 6 t ha^{-1})
Frossard *et al.* (2002)	Assessed the P status of compost using complementary approaches, isotope exchange kinetics, sequential extraction and solid state P^{31} NMR.	Composted organic solid waste contained between 2 and 16% of the total P as rapidly exchangeable inorganic P, between 40-77% of the total P as slowly exchangeable or not exchangeable inorganic P and the remainder was organic P.
Houot *et al.* (2002)	Evaluated the performance of field application of urban compost at 3-4 t ha^{-1} (dry wt.)	3% of the total N present in the compost was utilized by a maize crop.
Sinaj *et al.* (2002)	Assessed the importance of soil and compost properties on the utilization of compost-P by white clover.	6.5% to 18.5% of the compost P was plant available.
Sullivan *et al.* (2002)	Determined the effect of FW compost (food waste + wood waste + sawdust) and FYT compost (food waste + yard trimmings + paper) on N fertilizer use efficiency applied at 78 t ha^{-1}.	Compost did not affect the fertiliser N uptake efficiency.

7.3.3 LCI data

Wide variation in results exists in the literature on the effectiveness of compost to supply nutrients to crops and on the environmental impact of fertiliser production and use. Therefore, the available data in the literature has been grouped, averaged and given in Tables 7.7 and 7.8. In this study, it is assumed that the nutrient availability of a composted mulch and a soil conditioner are parallel as some investigators (Graham, 1998) observed similar plant responses under mulching and soil incorporation using the same composted product.

Table 7.7 LCI data for nutrient availability of composted products.

	Impact	Impact
N availability	1st year - 15%	Over 3-5 years approx. 40%
P availability	1st year - 38%	Over 3-5 years approx. 100%
K availability	1st year - 80%	Over 3-5 years approx 100%
Fertiliser use efficiency	<50 t ha^{-1} – not significant	
Nitrate leaching	<50 t ha^{-1} – not significant >50 t ha^{-1} – 10% of total N	
Phosphate leaching	<50 t ha^{-1} – not significant >50 t ha^{-1} – 1.5% of total P	

Table 7.8 Average energy consumption, emissions and leaching from production and use of inorganic fertilisers.

	Production			Use		
	N	P	K	N	P	K
Energy requirement (MJ kg^{-1})	72.27^{e}	12.92^{e}	9.95^{e}			
Phosphate extraction (kg kg^{-1}) (rock phosphate)		9.65^{a}				
Potash extraction (K_2O)(kg kg^{-1})			1.402^{a}			
Limestone ($CaCO_3$), (kg kg^{-1})	0.0264^{a}	0.0659^{a}	0.0174^{a}			
Water requirement (L kg^{-1})	1.5208^{a}	0.9898^{a}	2.77E-03^{a}			
CO_2 emission, air (kg kg^{-1})	3.96^{e}	1.76^{e}	1.36^{e}			
Methane emission, air (g kg^{-1})	7.1278^{a}	5.052^{a}	0.8344^{a}			
Hydrocarbon emission, air (g kg^{-1}) (except methane)	0.0181^{a}	4.1662^{a}	0.0109^{a}			
N_2O emission, air (g kg^{-1})	0.0177^{a}	0.0659^{a}	5.47E-03^{a}	12.5^{d}		
NO_x emission (as NO_2), air (g kg^{-1})	15.184^{a}	6.2775^{a}	1.001^{a}			
SO_x emission (as SO_2), air (g kg^{-1})	30.8692^{a}	7.592^{a}	2.3435^{a}			
NH_3 emission, air (g kg^{-1})	103.3^{a}	9.31E-05^{a}	2.98E-07^{a}	*140^{c}		
CO emission, air (g kg^{-1})	12.696^{a}	0.9496^{a}	0.0950^{a}			
NO_3^- emission, water (g kg^{-1})	1.21E-04^{a}	3.87E-04^{a}	7.71E-05^{a}			
Fluorides, air (g kg^{-1})		0.46^{b}				
Fluoride, water (g kg^{-1})		167^{b}				
P_2O_5 emission, air (g kg^{-1})		0.45^{b}				
P_2O_5 emission, water (g kg^{-1})		103^{b}				
Particulate emission, air (g kg^{-1})	4.4832^{a}	3.2765^{a}	1.2712^{a}			
HCl emission (g kg^{-1})		0.1885^{a}				
NH_3emission, water (g kg^{-1})	0.0617^{a}	7.57E-03^{a}	2.91E-04^{a}			
BOD, water (g kg^{-1})	1.878^{a}	0.0483^{a}	1.19E-03^{a}			
COD, water (g kg^{-1})	5.66^{a}	0.4027^{a}	9.95E-03^{a}			
As emission, water (g kg^{-1})		0.01^{b}				<1.2E-04^{f}

	Production			Use		
	N	P	K	N	P	K
Cd emission, water/soil (g kg^{-1})		0.01[b]			0.075[f]	<1.2E-04[f]
Cu emission, water/soil (g kg^{-1})		0.05[b]			0.017[f]	3.6E-04[f]
Cr emission, water/soil (g kg^{-1})		0.05[b]			0.666[f]	9.6E-05[f]
Hg emission, water/soil (g kg^{-1})		0.0095[b]				1.2E-04[f]
Ni emission, water/soil (g kg^{-1})		0.04[b]			0.085[f]	6.1E-04[f]
Zn emission, water/soil (g kg^{-1})		0.06[b]			0.796[f]	1.08E-03[f]
Pb emission, water/soil (g kg^{-1})		0.043[b]			0.055[f]	3.6E-04[f]
Gypsum emission, water (g kg^{-1})		7500[b]				

Source: [a]Extracted from data given in Sheehan *et al.* (1998), [b]Audsley *et al.* (1997), [c]FAO (2001), [d]UNEP (2000), [e]Wind and Wallender (1997), [f]US EPA (1999) (TSP is taken for P fertilizer reference), *weighted average for all N fertilisers.

7.3.4 Application scenario

Application of a composted soil conditioner at 12 t ha^{-1} would provide approximately 11 to 22 kg of N, 3 to 25 kg of P, and 6 to 17 kg of K per hectare in a cotton crop during the first year of application depending upon nutrient and moisture contents of soil conditioners. Total amount of N, P and K supplied through this application rate over the period of 3-5 years would roughly be in the range of 29-58 kg of N, 7-65 kg of P and 7-22 kg of K. Similarly, an application of a 10 cm deep and 50 to 75 cm wide composted mulch cover (average bulk density = 510 kg m^{-3}, application rate = 80 to 120 t ha^{-1} depending upon the width of mulch application) to a vineyard would provide approximately 27-40 kg of N, 45-68 kg of P, and 71-108 kg of K per hectare during the first year of application. Total amount of N, P and K supplied through this application rate over the period of 3-5 years would roughly be in the range of 72-108 kg of N, 120-180 kg of P and 90 to 135 kg of K. These quantities were calculated on the basis of the average composition of composted products given in Table 2.2 and availability of nutrients from composted products provided in Table 7.7. Therefore, the application of a composted soil conditioner or a composted mulch could replace the above mentioned N, P and K quantities which otherwise would have to be supplied by inorganic fertilisers resulting in reduction of adverse environmental impacts of using inorganic fertilisers. The reduction of adverse environmental impact resulted from reduced usage of inorganic fertiliser could be quantified by using the Table 7.8.

It has been reported that application of composted soil conditioners cause leaching of NO_3-N and PO_4-P (Insam et al., 1997; Jakobsen, 1996; Li et al., 1997). This has mainly resulted in situations firstly where mineral fertilisers were continued to be applied along with composted products and secondly for high application rates and high nutrient composted soil conditioners such as composted biosolids, composted manures, etc. Nutrient leaching from compost amended soils should be controlled if compost application rates and nutrient loading are based on crop nutrient requirements. Similarly application of composted mulch based on grapevine's nutrient requirements should not lead to leaching of NO_3-N and PO_4-P from the soil profile. In fact, nutrient leaching from compost amended soils should be reduced due to slow and long term nutrient releasing ability of composted products when compared with mineral fertilisers.

7.4 Reduced herbicide requirements

7.4.1 Introduction

Weeds reduce crop quality and yield by competing for light, water and nutrients and increase the harvesting cost. Therefore, weed control is an important task in agricultural production. There are many ways of controlling weed populations in agricultural lands. i.e. mechanical, chemical and biological control. Growers have become dependent largely on chemical weed control, as the cost of the herbicides and the labour to apply them are lower than the cost of mechanical control. Most agricultural crop production systems in Australia use herbicides for controlling weeds (except very few organic farms). Their popularity reflects cost savings in farm labour, species selectivity, their ability to increase yield and reduction in production costs (Altieri and Liebman, 1988).

However, long-term herbicide usage can have a potential negative impact on the environment because of ground water contamination and adverse effects on humans and wildlife (Crnko *et al.*, 1992). Major pathways for herbicide removal from croplands are through leaching, surface runoff water and sediments carried in the water (Schneider *et al.*, 1988).

Mulches can be effectively used to suppress weed growth in agricultural lands (FAO 1987; Grantzau, 1987). Application of composted and non-composted mulches was an important method of weed control in early agricultural systems prior to the development of herbicides (Altieri and Liebman, 1988). The suppression effects of mulches are due to the physical presence of material on the soil surface, and/or the chemical effects of phytotoxic compounds (volatile fatty acids and/or ammonia), higher CO_2 level and temperatures (Ozores-Hampton *et al.*, 1999; Shiralipour *et al.*, 199; Niggli *et al.*, 1990) and organic acids (Worsham, 1984) resulting from biological activity during degradation of organic matter.

The degree of weed control by organic mulch depends on compost mulch type, weed species, environmental conditions and thickness of the mulch layer (Ozores-Hampton, 1998). Weed control usually improves as the thickness of the organic mulch layer increases due to the lack of weed growth promoting factors such as light, temperature, moisture or O_2 level (Reisman-Berman and Kigel, 1991; Baskin and Baskin, 1989). Weed seed germination is frequently inhibited when seeds are buried in soil (Baskin and Baskin, 1989), and inhibition of germination increases with burial depth (Reisman-Berman and Kigel, 1991). Generally, a 10–15 cm thick mulch layer is necessary to control weeds most effectively (Marshal and Ellis, 1992; FAO, 1987).

Organic acids such as acetic, propionic and butyric acids are common phytotoxins found in mature and immature urban compost (De Vleeschauwer *et al.*, 1981). These phytotoxins in composts sometime can injure crops as well. The type and degree of injury are directly related to compost maturity or stability (Ozores-Hampton, 1998). Crop injury has been usually linked to use of immature composts (Zucconi *et al.*, 1981b). Phytotoxins present in immature urban compost may be primarily responsible for poor vegetable seed germination and growth inhibition rather than C:N ratio (Keeling *et al.,* 1994).

Organic mulches can reduce cultivation cost by 20–40% (Kolb, 1983) and can control weed growth as effectively as strong herbicides (Grantzau, 1987). Organic mulches applied at 5 t ha^{-1} in herb production controlled weeds effectively and at lower cost than did simazine (6-chloro-*N*-*N'*-diethyl 1,3,5-triazine-2,4-diamine), diuron [*N'*-(3,4-dichlorophenyl)-*N,N*-dimethylurea], and oxyfluorfen [2-chloro-1-(3-ethoxy-4-nitrophenoxy)-4-(tryfluoromethyl) benzene] herbicides (Singh *et al.*, 1985).

7.4.2 Literature review

Table 7.9 provides performance data collected from the literature relating to the effect of application of composted products on weed suppression under various soil and climatic conditions.

Table 7.9 Summary of weed suppression effects.

Study	Description	Impact
Shearman *et al.* (1979)	Investigated the effect of mulching at 7.5 cm and 15 cm with turf grass clipping, oat straw, and alfalfa hay on weed growth under bush beans in clay loam soils.	Weed cover after 45 days: Control - 57.5% Turf clippings (7.5 cm) - 2.5% Turf clippings (15 cm) - 0% Alfalfa hay (7.5 cm) - 6.5% Alfalfa hay (15 cm) - 1.0% Oat straw (7.5 cm) - 2.0% Oat straw (15 cm) - 1.3%
Nigli *et al.* (1990)	Conducted field trials for 8 years to evaluate the effect of various organic mulch materials on weed control in orchards.	Effect of 10 cm mulch cover on weed control was as good as herbicide treatments (spring: Simazine + Glufosinate, autumn: Glufosinate).
Roe and Stofella (1993)	Evaluated the effect of 5 cm urban compost mulch on weed growth in a sandy soil with bell pepper.	Weed cover after 73 days: Control - 72% Glyphosate - 49% Compost - 21%
Pinamonti (1998)	Studied the effect of mulching at 5 mm with urban compost and sludge and bark compost on weed growth in vineyards.	Weed growth was reduced by the compost treatments to the same extent as in mechanical tillage and herbicide treatments.
Wilkinson (2001)	Studied the effect of 10 cm deep straw, fine and coarse composted mulch from garden organics on weed growth in vineyards.	10 cm deep composted mulch reduced percentage of weed cover by 90% compared to control (no mulch).
Biala (2001)	Studied effect of application of 10 cm thick pasteurised and composted mulches on weed growth in vineyards.	After 4 months of application, pasteurised and composted mulches reduced weed growth from 70-80% compared to control.
Percy (2001)	Studied the effect of vermicompost+straw, composted mulch and wheaten straw on weed growth in grapevine at different sites.	10 cm deep composted mulch highly significantly (>90%) reduced weed growth under vines compared to control.
Ozores-Hampton *et al.* (2001)	Evaluated application of 8 weeks old urban compost at several mulching depths on weed control under bell pepper and squash in a sandy soil.	Average weed cover for two seasons after 240 days of application. 3.8 cm – 60% 7.5 cm – 48% 11.3 cm – 31% Paraquat – 26% Control – 100%

7.4.3 LCI data

A set of LCI data for production and use of pesticides (herbicides and insecticides) is given in Table 7.10. The data set is more generalized and therefore LCI data for each individual pesticide production and use may vary greatly from this set. Transformation and transport of pesticides are strongly influenced by site-specific conditions and management practices. In addition, pesticides include a broad class of agrochemicals with widely ranging properties that defy generalization. Therefore, there are no appropriate estimates available to calculate some of the LCI data for pesticide. The LCI data given in Table 7.10 could be used commonly for herbicide as well as for insecticide as the available literature on their environmental impacts does not distinguish them and commonly designated as agrochemicals.

The amount of leaching of herbicide could be considered to be very low as the rate of herbicide use is much lower compared to the fertiliser rates. Moreover, some herbicides are quickly degraded in soils or plants after the application (Sheehan et al., 1998). However, there are some herbicides, which have long half-life (time taken to degrade 50% of the chemical) with low soil sorption coefficients and higher water solubility. These herbicides are likely to leach and contaminate ground water.

Drift losses of herbicides depend on factors such as wind, nozzle size, and height at which the material is released. Majewski and Cabal (1995) showed that drift for ground- rig application ranged from less than 0.5% to 8% of the nominal application and was dependent on the nozzle type, hydraulic pressure, and wind speed. The drift from aircraft application ranged from 1 to 31% of the applied amounts.

In addition to the application drift, pesticide residues can be transported into atmosphere attached to dust particles disturbed during tilling operations. Another route of entry of pesticides into the atmosphere is through volatilisation. This is a major cause of pesticide disappearance from target area, particularly where they are surfaced applied, and the rate of this loss can exceed the rate of chemical degradation (Taylor and Glotfelty, 1988). Volatilisation is continuous and depends on soil moisture content (wet is worse), temperature, relative humidity, wind velocity, and soil type (Lewis and Lee, 1976). When spray application is used, loss from volatilisation ranges from 3% to 25% for most pesticides (Sheehan *et al.*, 1998). In contrast, pesticide losses from soil-incorporated application methods are usually much lower.

Table 7.10 LCI data for production and use of pesticides.

	Production	Use
Energy requirement, (MJ kg^{-1} active ingredient)		
- herbicide	288^c	
- insecticide	237^c	
- fungicide	196^c	
Water requirement (L kg^{-1})	2.797^a	
CO_2 emission, air (kg kg^{-1})	29.7^b	
Run off coefficient (kg kg^{-1})		0.03^a
VOC emission, air (kg kg^{-1}) (volatilisation and drifts)		0.31^a
Methane, air (kg kg^{-1})	0.0223^a	
Nitrous oxide, air (g kg^{-1})	0.7592^a	
CO, air (g kg^{-1})	2.5885^a	
Hydrocarbons, air (kg kg^{-1}) (except methane)	0.0613^a	
Particulate, air (g kg^{-1})	1.0755^a	
Sulphur oxides, air (kg kg^{-1}) (SO_x as SO_2)	0.0211^a	

	Production	Use
Nitrogen oxides, air (kg kg^{-1}) (NO_x as NO_2)	0.0147[a]	
Hydrogen fluoride, air (g kg^{-1})	5.48E-05[a]	
Ammonia, water (g kg^{-1})	0.0529[a]	
BOD, water (g kg^{-1})	0.0158[a]	
COD, water (g kg^{-1})	0.0886[a]	
Ammonium, water (g kg^{-1})	0.011[a]	
Nitrate, water (g kg^{-1})	5.584E-03[a]	

Source: [a]Extracted from data given in Sheehan *et al.* (1998), [b]Wind and Wallender (1997), [c]Biermann et al. (1999). Please note these are generic averages for herbicides, insecticides and fungicides.

7.4.4 Application scenario

In viticulture, increasing attention is being paid to floor management in order to minimise weed competition. However, there is no common guideline for weed control in grape cultivation in NSW. Weed management depends on the age of vines, weed types present, critical period of weed control (e.g. autumn, late winter and spring) and availability of registered herbicides (NSW Agriculture, 2002). In NSW grape growers use various herbicides to control weeds in their farms. Most of the herbicides are applied as directed sprays to avoid the crop rows in established vineyards. For example 360 g/L glyphosate is applied at the rate of 2-3 L per sprayed hectare to control annual weeds. However higher rate of glyphosate at 3-6 L are required to control perennial weeds such as couch, paspalum, etc. Some common herbicides used in Australian viticulture are Trifluralin, glufosinate-ammonium, amitrole, ammonium thiocyanate, fluazifop, paraquat, glyphosate, oxyfluorfen, diuron, napropamide, oryzalin, simazine, dichlobenil, and norflurazon (Parson, 1995).

Based on above literature, an application of 10 cm of composted mulch could effectively control weeds in vineyards (Wilkinson, 2001; Biala, 2001; Percy, 2001). Therefore, it could totally avoid herbicide applications in vineyards resulting in environmental and economical benefits. The reduction of adverse environmental impacts due to avoided herbicide applications could be quantified by using the Table 7.10.

Application of mature, stable composted soil conditioners does not suppress growth of weeds. Therefore there is no likelihood of weed suppression from application of soil conditioners in cotton fields.

7.5 Reduced biocide requirements

7.5.1 Introduction

Composts have the potential to suppress and control plant diseases. Variety of bacterial species such as *Bacillus* spp., *Enterobacter* spp., *Flavobacterium balustinum*, *Pseudomonas* spp. and fungal species such as *Streptomyces* spp., *Tricoderma* spp., and *Gliocladium virens* have been identified as biocontrol agents in composts (Hardy and Sivasithamparam, 1991; Phae *et al.*, 1990; Hoitink and Fahy, 1986). The disease suppression potential in composts is largely induced during the curing phase as biocontrol agents recolonise composts after the peak heating phase (Hoitink and Grebus, 1994).

Disease suppression with compost may be achieved by either limiting growth or saprophytic survival of pathogen, expression of the disease, or both. The disease suppression is often attributed to four mechanisms:

- competition for nutrients among pathogens and beneficial microorganisms (on roots as well as leaf surfaces),
- antibiotic production (by microorganisms in the compost),
- predation and parasitism, and
- systemic disease resistance induced in plants in response to compost treatments (Goldstein, 1998).

Compost prepared from heterogeneous organic materials may have highly suppressive effects against disease caused by *Pythium* spp. (Pascual *et al.*, 2000; Mandelbaum and Harder, 1990), *Phytophthora* spp. (Hoitink and Boehm, 1999; Widmer *et al.*, 1999), *Rhizoctonia* spp. (Tuitert *et al.*, 1998; Kuter *et al.*, 1983), and *Fusarium* spp. (Trillas-Gay *et al.*, 1986; Chef *et al.*, 1983). These diseases cause severe losses in many crops and are difficult to control.

The physical and chemical properties and the presence of beneficial microorganisms account for the suppressive effects of such composts (Hoitink *et al.*, 1993). The suppression action varies with the type of composting process, maturity and type of feedstock. For example, compost produced in the open near a forest, an environment that is high in microbial species diversity, are more consistently suppressive than those produced from the same materials in facilities that are partially enclosed, where few of these microbes survive (Kuter *et al.*, 1983). However, most of the beneficial microbes are killed due to heat build up during the commercial composting processes. These composts, although naturally suppressive for some pathogens, have to be incubated for a month or more before they are recolonised by the right-specific microbiota (Kuter *et al.*, 1988). In the field, several months may pass before suppression is induced (Lumsden *et al.*, 1983). To solve the problem of variability in suppressive action of composts to disease pathogens, specific bacterial and fungal inoculants, which can suppress disease pathogens, are introduced into composts after the peak heating phase (Hoitink, 1990; Phae *et al.*, 1990).

The decomposition level of organic matter in compost has a major impact on disease suppression (Hoitink and Grebus, 1994). For example, *Rhizoctonia solani* cannot colonise in mature compost with low cellulose. On the other hand, Trichoderma (biocontrol agent of *Rhizoctonia solani*) are capable of colonising in mature compost (Hoitink and Grebus, 1994). Because organic matter decomposition level is so important, compost must be stabilised adequately to enhance its biological control potential (Hoitink and Grebus, 1994). In practice this occurs in composts (tree barks, saw dusts, etc.) that have been stabilised far enough not to induce phytotoxicity or nitrogen immobilization (Hoitink and Grebus, 1994). The slow release nature of organic nutrients tied up in mature compost support activity of microbiota for a longer duration thus sustaining its biocontrol potential. However, excessively stabilised organic matter does not support adequate activity of biocontrol agents (Hoitink and Grebus, 1994). For example, disease suppression is lacking in highly mineralised soils with high content of humic substances (Hoitink and Grebus, 1994). Abiotic factors such as soil pH and nature of clays influence the disease suppressive effects of composts and soils (Alabouvette, 1999).

The compost can also suppress vascular and foliar diseases by inducing plant resistance due to root colonisation of some of the microbes present in the compost (Maurhofer *et al.*, 1994; Wei *et al.*, 1991). Some investigations revealed that compost water extracts could be sprayed to control above ground plant diseases (Weltzhien, 1992).

The disease suppression induced by composts can be temporarily destroyed by unfavourable chemical conditions such as high salinity and soluble nitrogen content. Hoitink and Grebus (1994) reported that composted municipal sludge with a high salinity value (>10 mS/cm in the saturated paste) applied to soybean just before planting increases Phytopthera root rot over that in the control.

These disease suppressive effects can be used to control several important soil borne diseases, effectively leading to a reduction in the amount of pesticides used in agriculture to a great extent. Therefore, a major research and development effort must be directed into this aspect of disease control.

7.5.2 Literature review

Table 7.11 provides performance data collected from the literature relating to the disease suppressive effect of compost application on a variety of crops grown under a range of soil and climatic conditions.

Table 7.11 Summary of disease suppressive effects.

Study	Description	Impact
Lewis *et al.* (1992)	Assessed the effect of field amendment with composted sewage sludge (7 - 10 t ha^{-1}) on damping-off of peas and cotton caused by *Pythium ultimum* and *Rhizoctonia solani.*	Compost reduced the incidence of damping-off in pea and cotton probably due to induction suppression. The disease suppressive effect was sustained over 4 years period even though compost was added only during the first 2 years.
Serra-Wittling *et al.* (1996)	Investigated the suppressive properties of urban composts to Fusarium wilt of flax caused by *Fusarium oxysporum* f. sp. *lini* in soil/compost mixtures.	Compost addition increased disease suppressiveness of the soil proportionately to the application rate. Both microflora of soil and compost were involved in the suppressiveness and mainly acted through nutrient and space competition towards the population of the pathogen.
Marull *et al.* (1997)	Examined the value of urban compost residues for control of root-knot nematodes in a typical Mediterranean-intensive horticultural production system.	Plants in soils amended with compost had lower number of root-knot nematodes (*Meloidogyne javanica*).
Pitt *et al.* (1998)	Evaluated the phytosanitary potential of recycled organic material compost for the control of soilborne pathogens in a pot trial.	Addition of compost reduced wheat take-all (*Gaeumannomyces graminis*) up to 80%, root-rot of peas (*Phoma medicaginis*) to 66%, red-core of strawberries (*Phytophthora fragariae*) to 70% and club root of brassicas (*Plasmodiophora brassicae*) to a negligible level. The disease suppressive properties of compost were dose responsive and due to a combination of direct parasitism, competition for resources and the production of anti-fungal compounds.
Widmer and Graham (1998)	Studied the effect of application of urban compost on Phytophthora root rot in citrus seedlings.	Incidence of root infection of *Phytophthora nicotianae* in citrus seedlings was reduced in soils amended with composts as long as they were relatively fresh (< 6 months from time of composting).
Boulter *et al.* (2002)	Evaluated the capacity of selected compost formulations (at 5 –10 t ha^{-1}) to suppress snow molds of turf grass caused by *Microdochium nivale* and *Typhula ishikariensis.*	Compost application reduced the snow molds severity to the levels not significantly different from fungicide control.
Cotxarrera *et al.* (2002)	Investigated the effects of applying compost (prepared from vegetable and market wastes, sewage sludge and yard waste) on control of Fusarium wilt in tomato grown in pots inoculated with the pathogen.	Compost showed a high ability to suppress Fusarium wilt caused by *Fusarium oxysporum* f. sp. *lycopersici* race 1. *Trichoderma asperellum* was found to be the biocontrol agent.
Khalilian *et al.* (2002)	Evaluated the effect of surface application of urban compost (at 11.2, 22.4 and 33.6 t ha^{-1}) on nematode management in cotton cultivation.	Columbia lance nematode (*Hoplolaimus Columbus*) densities were reduced due to the surface application of compost and the nematode control by compost application was comparable to plots treated with Temik 15G nematicide.
Reuveni *et al.* (2002)	Examined the potential of compost made by mixing cattle manure, chicken manure and wheat straw to suppress Fusarium in sweet basil (*Ocimum basilicum* L.).	Compost induced protection against *Fusarium oxysporum* f. sp. *basilici* and reduced the severity of the visual symptoms of Fusarium wilt.

7.5.3 Application scenario

Even though compost is known to have disease suppressive effects, it is rather difficult to attach a value to it because at present it is not known which diseases can be suppressed by compost application in cotton and grape cultivation in NSW and replace biocides used to control those diseases. However, according to information from NSW Agriculture (Media release 10 April 2002, http://www.agric.nsw.gov.au/reader/12905), cotton growers in NSW face a possible Fusarium wilt epidemic. Currently, there are about 57 farms in NSW where the disease has been identified. Since compost is well known to control Fusarium wilt in a variety of crops, there is strong potential for cotton growing to benefit from compost additions in controlling the Fusarium wilt. Vineyards also have the potential to reduce or replace the application of biocide used against some diseases. Compost application can potentially substitute for any biocide, which would be required to control some diseases in future. The avoided impact of not spraying biocides could be quantified from Table 7.11. Use of compost against diseases reduces the risk of crop damage and application of biocide including application rate and frequency of biocide application.

7.6 Sequestration of carbon in soils

7.6.1 Introduction

Carbon sequestration refers to natural or man-made processes that remove carbon from the atmosphere and store it for long periods or permanently. A store of sequestered carbon (i.e. soil) is known as a carbon sink (US EPA, 1998). Carbon sequestration in soils will be an important agenda in controlling global warming attributed to greenhouse gas emissions. Principle processes of carbon sequestration in soil include humification of organic materials, aggregation by formation of organomineral complexes, deep placement of organic matter beneath the plough zone, deep rooting, and calcification (Bruce *et al.,* 1999).

The Kyoto protocol (1997), an international treaty to limit greenhouse gas emissions, when fully implemented requires most nations of the world to reduce their net emissions of greenhouse gases by agreed, specified amounts by year 2012. In recent agreements under this protocol (Article 3.4), carbon sequestration in soils qualifies for inclusion in international carbon accounting and trading process. Therefore, the potential capacity of agricultural soils in NSW to sequester carbon might represent a tradeable carbon credit to offset greenhouse gas emissions. Estimates of the carbon sequestration potential of NSW soils from experimental data by paired sites comparisons and modelling ranged from 6–40 t ha^{-1}, depending on soil texture and climate (NSW Agriculture, 2002).

Changes in soil carbon content reflect the net result of carbon input (e.g. compost application, crop residues) and carbon loss (via decomposition). To gain carbon storage in soil, it is therefore necessary to increase the amount of carbon entering the soil and/or to suppress the rate of soil carbon decomposition (Bruce *et al.,* 1999). The amount of carbon entering the soil is a function of the amount of organic material applied to soil in addition to related management regimes affecting plant yield and plant management (e.g. stubble retention). Long term field experiments reviewed by the US EPA (2002) and others have shown a linear relationship between carbon inputs and soil carbon levels. This relationship varies depending upon climate and other soil and management factors, which influence decomposition rates (Post and Kwon, 2000).

Post and Kwon (2000) examined the effects of returning agricultural land to perennial vegetation or forests. When agricultural land is no longer used for cultivation and allowed to revert to natural vegetation or replanted to perennial vegetation, soil organic carbon can accumulate. Post and Kwon (2000) determined that the average rates of carbon (C) accumulation are similar for forest or grassland establishment: 33.8 g C m^{-2} and 33.2 g C m^{-2} respectively. These results are relevant to the rehabilitation of degraded soils.

Adoption of sustainable practices such as minimum tillage, crop rotations, stubble retentions, green manuring and soil and water conservation practices represent opportunities to accumulate soil organic carbon through carbon sequestration (Pretty and Ball, 2001; Rosenzweig and Hillel, 2000; Smith *et al.*, 2000). In this context, application of composted product derived from recycled organics could enhance these opportunities in building up of soil carbon sinks in many agricultural systems in NSW.

The cereal cropping soils alone in the Western district of NSW offer a potential sink for over 5 million tonnes of carbon (NSW Agriculture, 2002).

Horst *et al.* (1998) simulated the effect of compost applications on cropping soils in Europe. It was found that soil organic matter content on a loam soil continues to increase even after 50 years of compost application. For sandy soils organic matter levels reached the equilibrium after about 25 years. This increase in soil organic carbon is sequestered carbon, not contributing to greenhouse gases in the atmosphere.

The rate of decomposition is controlled by soil conditions (e.g. moisture, temperature, and oxygen, composition of the organic material, placement of the material within the soil profile, and the degree of physical protection i.e. within soil aggregates) (Bruce *et al.,* 1999). For example, the mineralisation resulting from soil cultivation is estimated to contribute the loss of 4 million tons of carbon per annum in Australia (Swift, 2001). This may be due to aggregate breakdown by cultivation leading to rapid mineralisation of carbon previously encapsulated within the aggregates.

Soil carbon storage can also change through erosion, which redistributes carbon across the landscape. Thus, some parts of the landscape may lose carbon while others may gain carbon. Because some of the eroded material is deposited elsewhere on the landscape or in water systems, not all the carbon lost by erosion can be considered a net contribution to atmospheric carbon dioxide. For the same reason, soil carbon gains resulting from a practice that reduces erosion cannot be entirely equated to removal of atmospheric carbon dioxide (Bruce *et al.,* 1999).

The process of turning agricultural soils into sink for organic carbon sequestration would be complete only if the microbially stable humic matter is increased since most of the soil carbon is mainly allocated in rapidly cycling unstable labile carbon pools (Schlesinger and Licther, 2001). Humified organic carbon, humic acids and humin in particular, represents the most persistent pool of soil organic carbon with mean residence times of several hundreds of years (Piccolo, 1996).

The percentage of stable carbon remaining in the soil is a factor of climate, soil type, type of organic matter and land use and management practices. Soil organic carbon in a stable pool is chemically and/or physically protected and less readily impacted by agronomic practices (Eshwarn *et al.*, 1995). Organic matter decomposition in clayey soils is slower than in sandy soils and is linked to greater physical protection against microbial attack (Hassink, 1992). The soil organic carbon, adsorbed on to mineral surfaces or occluded within soil structural units, is resistant to decomposition and could be considered more stable (Jenkinson, 1988). Skjemstad *et al.* (2001) observed that soil organic carbon composed of aromatic carbon structures are more resistant to decomposition than that of aliphatic carbons.

Globally, 20 to 50% of the stable carbon pool present in natural grassland and forests is lost by 40-50 years of land use change (Swift, 2001; Bruce *et al.*, 1999). In Australia, the soil carbon lost due to the cultivation over 10-80 years was estimated to vary from 10–60% (Dalal and Chan, 2001). Lal (1997) suggested that 15% of crop residue would be converted to stable form of soil organic carbon. Heenan et al., (1996) conducted long-term trials at Wagga Wagga, NSW and suggested that only 4.6% of the crop residue would be converted to stable soil organic carbon.

The application of composted products from recycled organics has the potential to contribute to the carbon sequestration process in agricultural lands in NSW and thus represents a modest contribution in the overall carbon accounting process in addition to other benefits. Adding compost has the potential to shift the soil carbon balance by retaining some of the compost carbon in the soil system. In addition, compost application has the potential to stimulate crop productivity resulting in further increase in the soil carbon pool due to a larger volume of crop residues. Further, composting leads to increased formation of stable carbon compounds (i.e. humic substances, aggregates) that can be stored in the soil for long (>50 years) period of time (US EPA, 2002). Therefore, the application of compost produces a multiplier effect by quantitatively changing the dynamics of the carbon cycling systems and increasing the retention of carbon from non-compost sources (crop residues). This multiplier effect could enable compost to increase carbon storage by more than its own direct contribution to carbon mass accumulation.

In addition, composting produces a greater proportion of humus than that typically formed when organics are left directly on the ground due to the involvement of thermophilic bacteria in the composting process (US EPA, 2002). Thermophilic bacteria tend to produce a greater proportion of stable, long-chain carbon compounds than do bacteria and fungi that predominate at ambient temperature. However, according to some authors, addition of organic amendments like compost and livestock manures is not likely to yield a net sink for carbon in soils, as would be required by the Kyoto protocol because carbon accumulated in these products are merely recycling crop carbon (Schlesinger, 2000; Bruce *et al.*, 1999). Nevertheless, the framework used here describes the soil carbon benefits of compost application relative to other processes of municipal waste disposal.

7.6.2 LCI data and application scenario

Due to the complexity of the processes involved in soil carbon storage, it is difficult to identify a precise sequestration LCI data set. The carbon storage process due to compost application probably is controlled by a number of biological, physiochemical, and compost management factors such as application (i.e. silviculture, horticulture, agriculture, and landscaping), application rate, climatic factors, soil type, and, to a lesser extent, compost feedstock (US EPA, 2002). Therefore, this study directly applies the data on carbon storage estimates for composted organics developed by US EPA (2002).

In developing estimates, US EPA (2002) extensively researched the processes that affect soil carbon storage, reviewed the results of experiments on the soil carbon impacts of applying organic amendments, and interviewed experts on the potential carbon storage benefits of composting organics as compared to other methods of disposal. As it was difficult to find any primary data that could be used to develop quantitative estimates of the carbon storage benefits of compost, US EPA (2002) opted for using a simulation model, CENTURY (Metherell *et al.*, 1993), to quantify the soil carbon sequestration. However, US EPA (2002) was unable to estimate the quantities of carbon stored due to multiplier effects (i.e. increased crop residues due to increased crop production). Therefore, estimates given by US EPA (2002) are likely to be conservative (i.e. understate carbon storage rates), at least for soils with high silt and/or clay content where the multiplier effect is most likely to apply.

Carbon addition to a soil can flow through various pools and can also exit the system (i.e. as CO_2, dissolved carbon, or through crop removal). The carbon in a soil system could be in either an active pool, slow pool or in the passive pool (US EPA, 2002). The active pool of soil organic matter includes living biomass and debris from dead animals and plants. The active pool is estimated to have a mean resident time (MRT) of a few months to 10 years (Metherell *et al.*, 1993; Brady and Weil, 1999). The slow pool includes resistant plant material (i.e. high lignin content) and has and MRT of 15–100 years (Metherell *et al.*, 1993). The passive pool of organic matter includes very stable materials remaining in the soil for hundreds to thousands of years (Metherell *et al.*, 1993). Therefore, this study considers only the fraction of organic carbon, which will remain in the passive pool. However, for any carbon accounting process, both passive and slow pools could be considered as tradeable carbon credits to offset greenhouse gas emissions. According to US EPA (2002), the passive pool consists of 45-60% of the carbon in composts while 4–20% and 30-45% of the carbon in composts can be considered to be in the active and in the slow pool respectively.

Since the model CENTURY does not simulate for organic matter additions with high humus content to directly increase the magnitude of the passive pool, US EPA (2002) used bounding analysis to estimate the upper and lower limits of the magnitude of this effect. In this analysis, they evaluated the amount of long-term soil carbon storage when organics are composted and applied to soils. It has been estimated that approximately 0.03 metric ton of carbon equivalent per wet short tonne (1 metric tonne = 1.12 short tonne) of organics is sequestered after about 100 years. This is equivalent to 0.07056 (0.03*1.12*2.1) metric tonne of carbon equivalent per metric tonne of compost based on assumptions (2.1 tonnes of yard trimmings [garden organics] are required to generate 1 tonne of compost with 30% moisture) made by US EPA (2002). NSW Agriculture (2002) literature review on soil carbon sequestration using a variety of organic materials has provided a value of 0.008 to 0.08 tonnes of carbon for each tonne of added RO products in the top 10 cm soil layer after about 1-5 years of application.

US EPA (2002) also estimated that approximately 0.05 metric tonne of carbon equivalent per wet short ton of organics is sequestered after 10 years. This is equivalent to 0.1176 (0.05*1.12*2.1) metric tonne of carbon equivalent per metric tonne of compost based on assumptions made by US EPA (2002). This figure on carbon sequestration after a 10 year period could be used in any carbon accounting process, as the Kyoto protocol when fully implemented requires most nations of the world to reduce their net emissions of greenhouse gases by agreed, specified amounts by year 2012.

In addition, US EPA (2002) has found that the carbon storage rates decline with time after initial application and is similar across application rates and frequencies, and across site conditions. Soil carbon storage rate varies from about 0.08 metric tonne carbon equivalent per wet short tonne organics immediately after compost application to about 0.02 metric tonne carbon equivalent per wet short tonne organics 24 years after the last application.

In the context of this study, the value of 0.07056 metric tonne or 70 kg of carbon equivalent per metric tonne of compost is taken as the carbon sequestration potential of composted product in agricultural use.

7.7 Remediation of saline-sodic soils

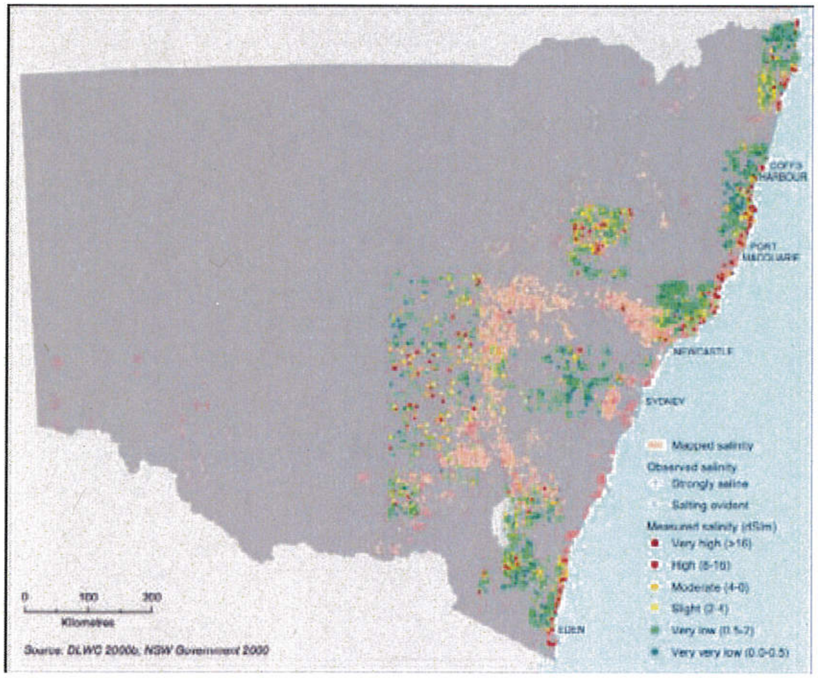
7.7.1 Introduction

Salinity and sodicity are serious problem in NSW with many environmental, economic and social impacts. The costs associated with salinity are potentially enormous and are borne not only by the rural community, but ultimately by the whole state and its environment. Salinity contributes to significant losses of productivity in agricultural lands in NSW and may take some lands entirely out of production. In addition to reduced agricultural production, there are also costs associated with protecting land from further degradation, remediation of salt scalds, groundwater monitoring, fencing, revegetation and repairing damage to farm infrastructure. Estimation of the area of land affected by dryland salinity in NSW by 1999 varies between 120,000 and 174,000 hectares (DLWC, 2000a). A ragged band stretching from Albury, past Yass, Wagga Wagga, Forbes, Dubbo, north to Tamworth and Gunnedah in the area most extensively affected by salinity as can be seen in Figure 7.6. Salinity has also been observed in low-lying coastal lands (EPA, 2000). Estimated costs of dryland salinity in NSW included $700 million in lost land and $130 million annually in lost production (EPA, 2000).

Although soil sodicity is less commonly known than soil salinity, it affects considerably greater land areas and costs at least $1.3-$2 billion each year in lost agricultural production (EPA, 2000). Land affected by soil sodicity has many adverse soil properties. The poor structure of sodic soil causes drainage and infiltration problems, erosion, compaction, surface sealing, and excessive surface runoff (EPA, 2000). Sodic soils are generally found in low rainfall areas. In NSW sodic soils are found across eastern, central and on the western plains of NSW. Sodic soils are also found in coastal areas of Northern NSW (see Figure 7.8). Regular gypsum application can ameliorate sodic soils to a certain extent.

Land clearing has led to increased recharge of groundwater resulting in rising groundwater levels, possibly reaching the surface. Since many areas of Australia have naturally high levels of salt in the soil and groundwater, the rising groundwater carries salt dissolved from the underlying soils and bedrock material to the surface. Large areas of the Western Slopes, the Hunter Valley, and the Sydney Basin already have saline groundwater within two meters of the surface (Figure 7.7). Modelling of groundwater systems in NSW by DLWC for the Murray-Darling Basin salinity audit (MDBMC, 1999) estimates that there are now 5.4 million ha in NSW with high groundwater at or near the land surface. The findings of the Murray-Darling Basin Salinity Audit also indicate that, if we continue to use our landscape the way we use it now, the area of land affected by serious salinisation could increase to 2-4 million hectares by 2050 in the New South Wales section of the Murray Darling Basin alone.

Figure 7.6. Mapped, observed and measured salinity (NSW EPA, 2000).

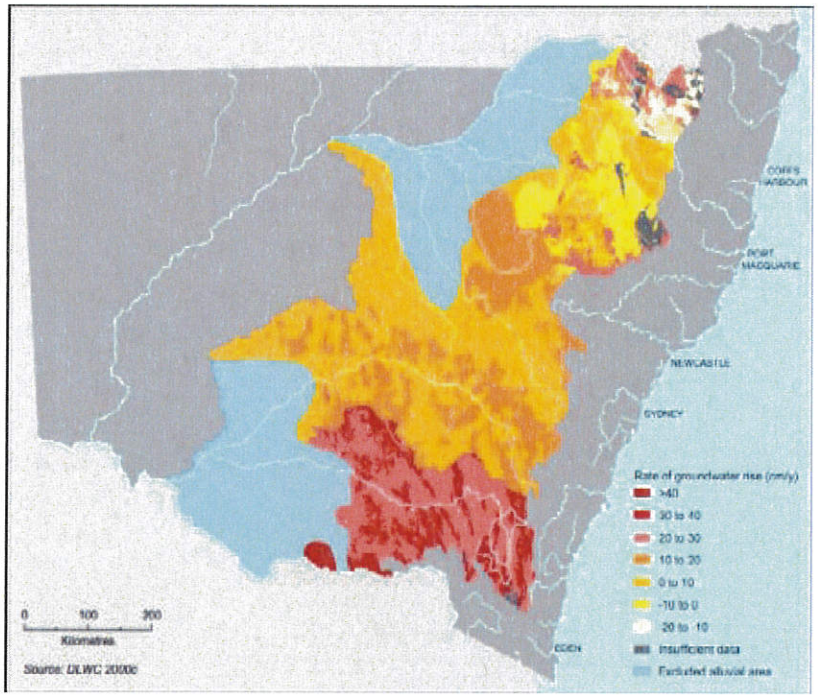


Source: DLWC 2000b; NSW Government 2000

The quality of water in NSW river systems also declines due to the increase in saline discharges from saline landscapes. Already more than 230,000 tonnes of salt pass Narromine in the Macquarie River annually (DLWC, 2000a). River salinity makes water less suitable for domestic consumption and irrigation and results in increase in cost for water treatment. It also affects the natural ecosystems associated with rivers. As river salinity rises, vegetation along the bank may be threatened, resulting loss in important wildlife corridors and stream stability. The health of the aquatic environment may decline or be entirely lost. For example, it has been predicted by Murray-Darling Basin Salinity Audit (1999) that the Macquarie Marshes and Great Cumbung Swamp will experience a severe loss of biodiversity over the coming years, if no actions are taken to address the threat.

The potential effects of salinity and associated water logging on infrastructure, building and domestic houses are believed to be enormous. Salinity causes breaking down of bricks, mortar and concrete in buildings, corrosion of metal, shifting or shrinking of foundations, and pot holing, cracking and crumbling of road bases. Salinity currently affects at least 40 towns in the Murray-Darling Basin as well as part of Western Sydney and lower Hunter Valley (DLWC, 2000a).

Figure 7.7. Groundwater rise in selected regions of the Murray-Darling Basin (NSW EPA, 2000).



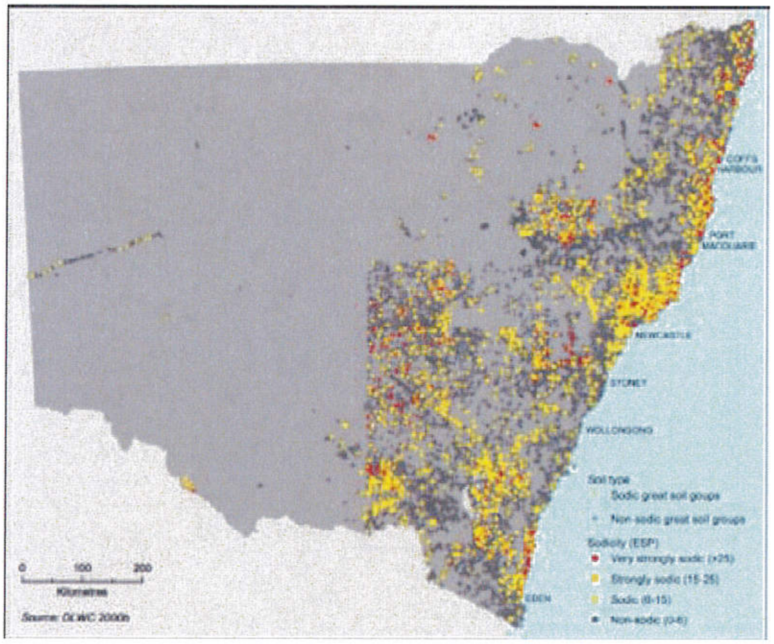
Source: DLWC 2000c

Saline soils, being dominated by NaCl, lead to the formation of sodic soils when salts are leached. Sodic soils derived from saline conditions tend to lose their structure, become compacted and impermeable and difficult to reclaim. As mentioned earlier many NSW soils naturally contain salt in the top 1 to 2 metres and when the soil dries out, the concentration of salt increases and moves up the profile affecting plant growth. This is called transient salinity (Rengasamy 2000), which starkly differs from water table-induced salinity. Transient salinity is extensive in sodic soils in Australia where water tables are deep. This type of salinity may be prevalent in viticultural areas and potentially causing production loss in millions of dollars. Unfortunately such processes, though slow are irreversible in nature. There is evidence that stubble retention (like mulch) stores water deep in the profile, e.g. at 2m according to O'Leary & Connor (1997) though much of this water may deep drain and add to the water table (pers comm. Paul Hutchinson-CLW Griffith).

The different means to reclaim saline-sodic soils are based upon the leaching of the excessive salinity out of the root zone. Reasonable infiltration rate and hydraulic conductivity are prerequisites for any such process (Avnimelech et al., 1994). In this regard, addition of composts is useful in reclamation of saline-sodic soils since it improves and stabilizes the soil structure ensuring a relatively high infiltration rate and hydraulic conductivity (Avnimelech et al., 1994). It also increases solubility of $CaCO_3$ due to the organic acids and chelating agents generated along with degradation of organic matter (Sekhon and Bajwa, 1993). This in turn can lead to the replacement of exchangeable sodium by calcium. In addition, soil channels made by decaying plant roots due to improvement in plant growth from compost addition may further enhance the leaching process (Avnimelech et al., 1994). Further, compost can significantly reduce the recharge of the groundwater table as discussed in Section 7.2. This can substantially reduce the risk of dryland salinity caused by rising groundwater levels in agricultural lands.

Major requirements for increasing productivity in dryland sodic soil are improved water storage and transport in the root zone and improved crop water-use efficiency. This will also lead to improved drainage and leaching of accumulated salts below the root zone. Development of soil structure and its maintenance over time is essential for sustained plant productivity, which can be achieved through compost application as an alternate to chemical ameliorants such as lime and gypsum. A significant potential benefit from compost use in saline areas could also come from increased crop growth increasing demand for water, which in turn has the potential to reduce groundwater recharge.

Figure 7.8. Soils with sodic horizons (NSW EPA, 2000).



7.7.2 Literature review

Table 7.12 shows performance data collected from literature on the effect of compost addition on soil sodicity and provides a base for the life cycle inventory of the effect.

Table 7.12 Effect of compost addition on sodicity

Study	Description	Impact
Sekhon and Bajwa (1993)	Studied the effect of green manuring, farm yard manure, rice straw (0.5–1.0% w/w) and different levels of gypsum in controlling build up of Na in a calcareous sandy loam receiving sustained sodic irrigation in a rice-wheat-maize system.	Incorporation of organic product has similar results compared to gypsum treatments in reducing exchangeable Na percentage and increasing crop yields.
Avnimelech *et al.* (1994)	Compared the effect of urban compost application at 200 and 400 m^3ha^{-1} and addition of gypsum (5 t ha^{-1}) on the reclamation of saline and alkaline cotton growing soils in Israel in a two year field experiment grown with oats and wheat.	Addition of urban compost was equivalent, or even superior to the addition of gypsum in reclaiming saline and alkaline soils. The effect of compost was long lasting.
Minhas *et al.* (1995)	Conducted a field experiments involving rice-wheat rotation to evaluate the effect of farmyard manure (15 t ha^{-1}) and gypsum in ameliorating the soil irrigated with saline water.	The farmyard manure reduced soil sodicity significantly and was more effective in ameliorating sodicity compared to gypsum treatments.
Wahid *et al.* (1998)	Examined the amelioration of saline-sodic soils with application of farmyard manure, wheat straw and clover hay at 1 and 3% of soil weight.	Farmyard manure at 3% rate ameliorated salt affected soils and promoted wheat growth better than that of clover hay and wheat straw.
Slattery et al. (2002)	Investigated the effect of composted feedlot manure (109 t ha^{-1}) on chemical properties of duplex soils in north eastern Victoria.	Decreased exchangeable sodium percentage in 40-60cm layer by 10%.

7.7.3 LCI data and application scenario

The above literature review makes us to assume that continuous application of composted soil conditioners has the potential to ameliorate sodic lands and thus could substitute gypsum requirement (2-5 t ha^{-1}) necessary to reclaim those lands, however there are no data available how much gypsum can be replaced through the addition of composted products. Gypsum requirement varies between 2.5 to 5 t ha^{-1} depending on the sodicity level and the soil type. The substitution of gypsum with composted soil conditioner will avoid environmental impacts of producing gypsum. There is no information available on the effect of compost as a mulch on soil salinity and/or sodicity.

Even though the addition of compost has the potential to influence groundwater levels and subsequent salinity development, it is very difficult to obtain a definite relationship between the soil salinity and the rise in the groundwater table since it varies from place to place depending on the hydraulic properties of the catchment and the amount of salt present in groundwater and soil.

7.8 Reduced erosion

7.8.1 Introduction

It has been estimated that 14 billion metric tons of topsoil are lost throughout Australia every year through erosion and more than 35% of NSW is affected by some form of water erosion (DLWC, 2000e). Erosion occurs when wind and rain dislodge topsoil from fields and hillsides. Stripped of its valuable top layer, which contains many nutrients, the soil left behind is often too poor to sustain good plant growth. Eroded topsoil can be carried into rivers, streams, and lakes. This excess sediment, sometimes containing fertilizers or toxic materials, threatens the health of aquatic organisms. It can also compromise the commercial, recreational, and aesthetic value of water resources. As a result, preventing erosion is essential for protecting waterways and maintaining the quality and productivity of soil.

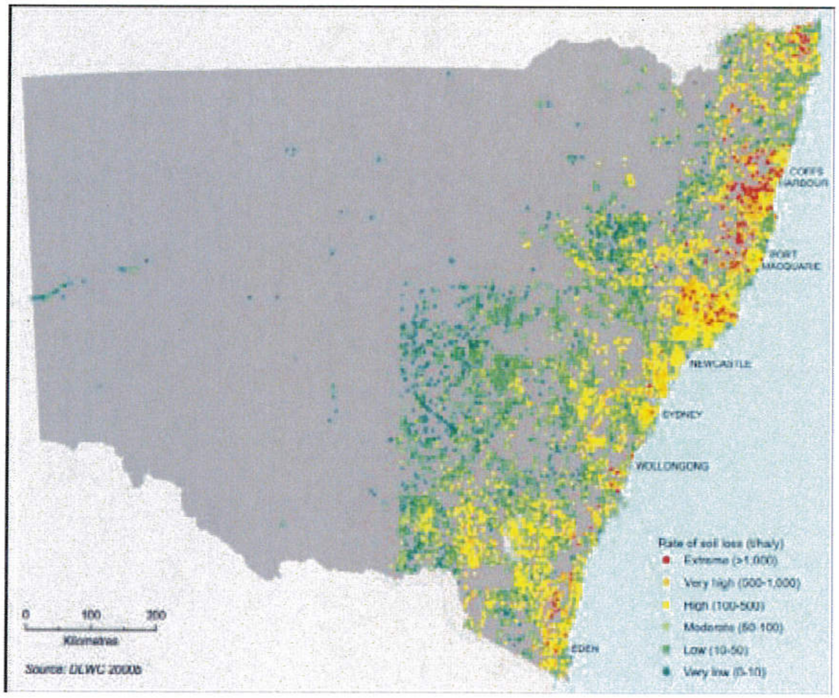
The most severe water erosion is found on the inland slopes of the Great Dividing Range in NSW. These are areas subjected to hot dry seasons alternating with a wet season with a total rainfall greater than 500 mm (EPA, 2000). The water erosion is largely due to past land use practices such as excessive clearing of native vegetation followed by over grazing by introduced animals and pests. Fire also removes both the native plants and vegetation litter covering the ground, exposing areas subjected to fire to erosion (EPA, 2000). Disturbance and earthworks associated with urban development and redevelopment in NSW also result in substantial soil losses through erosion. Significant areas of NSW are also susceptible to wind erosion. Wind erosion is a particular feature of the drier part of NSW, especially those areas of fine sandy soils such as the mallee lands of central and southwestern plains of the Murray-Darling Basin and is largely due to cultivation of these lands (EPA, 2000). Figure 7.9 shows the inherent sheet erosion risk by water for NSW soils. Inherent sheet erosion risk is the long-term susceptibility of a parcel of land to erosion if the soil is exposed for 12 months and no erosion control measures are employed (EPA, 2000).

Deposition of eroded sediment can lead to nutrient enrichment of streams, resulting in eutrophication of waterways. The level of nutrients generated from various forms of land use can be substantial. A recent study in the Hawkesbury-Nepean catchment (where market gardens had a history of fertilizer overuse) indicates that 11 kg of P and 127 kg of N per hectare could be lost with storm water runoff due to various land use activities in each year (Hollinger *et al.*, 2001).

Future climate change will bring about changes in precipitation. In case of increased precipitation, the runoff and erosion rates are likely to increase at a even greater rate than the rainfall, if we don't change our farming practices (Pruski, 2002). In the case of decreased rainfall scenario, it is predicted that erosion rates are just as likely to increase as to decrease. Given these results, along with the likelihood of overall increases in heavy rain during the next century, Karl *et al.* (1996) concluded that the overall story is one of increased erosion. This is of particular concern for vineyards, which are often established on hill slopes.

Application of composted products as surface mulch can reduce water erosion significantly as the mulch cover will protect the surface layer from the impact of rain. Surface mulching with composted product can also reduce the wind erosion substantially. Incorporation of composted products can reduce soil erosion by reducing effects of runoff generation due to improved soil structure, increased infiltration and water holding capacity, and reduced surface sealing.

Figure 7.9. Inherent risk of sheet erosion by water in NSW soil profiles (Source: NSW EPA, 2000).



7.8.2 Literature review

Table 7.13 shows performance data collected from literature on the effect of compost addition on soil erosion and provides a base for the life cycle inventory of the effect.

Table 7.13 Effect of compost addition on soil erosion

Study	Description	Impact
Ettlin and Stewart (1993)	A study was conducted to assess the effectiveness of mixed yard debris compost and leaf compost at 3" (7.62 cm) mulch cover on a landfill with 34 – 42% slope.	Untreated bare soil had 31,000 ml L^{-1} of suspended solids whereas mixed yard debris compost and leaf compost treatment had 740 and 280 ml L^{-1} of suspended solids.
de Vos (1996)	Investigated the potential of garden and household waste compost mulch (5.6 t ha^{-1}) in wind erosion control in a sandy topsoil.	Surface application of compost prevented soil loss from wind erosion at a wind speed of 11-15 ms^{-1}.
Agassi *et al.* (1998)	Evaluated the effect of the amount and mode of application of urban compost on runoff. A control, three mulching treatments (100, 200, and 300 m^3ha^{-1}) and one soil incorporation (200 m^3ha^{-1}) were exposed to a 40 mm h^{-1} rainfall at 5% slope.	It was possible to control the runoff effectively by mulching at 100 m^3ha^{-1} (1 cm mulching depth). 100, 200 and 300 m^3ha^{-1} mulching treatment reduced runoff by 74.5, 74.2, and 70.9% compared to the control respectively while the incorporation treatment of 200 m^3ha^{-1} (120 t ha^{-1}) decreased the runoff by 17.3%.
Bazzoffi *et al.* (1998)	Studied the effect of compost derived from urban refuse on runoff and soil erosion in a clay loamy soil (15% slope) in a 3 year field experiment with maize. The compost was incorporated into the soil at a rate of 64 t ha^{-1}.	Runoff reduction due to compost ranged between 7 and 399 m^3ha^{-1}. Compost application reduced soil loss by 31% compared to the control.
Albaladejo *et al.* (2000)	Determined the effect of applying several rates (65, 130, 195, and 260 t ha^{-1} mixed in top 20 cm layer) of organic urban solid refuse on total runoff and soil loss in a silty clay soil with 10% slope.	Organic urban solid refuse reduced the soil loss by 81, 97, 98, and 99% respectively by 65, 130, 195, and 260 t ha^{-1} application rates compared to the control. This was due to a reduction of runoff by 67, 94, 95, and 98% by respective treatments compared to the control.
Loughran *et al.* 2000	Conducted an investigation to measure the soil erosion in 19 vineyard blocks in Pokolbin, NSW and compared with estimated soil loss values generated from two water erosion models (SOILOSS & MUSLE).	Average measured soil loss was 17.5 t ha^{-1}. However, SOILOSS model predicted an average soil loss of 14.6 t ha^{-1} yr^{-1} and MUSLE model predicted soil loss of 38.3 t ha^{-1} yr^{-1} in vineyards.
Edwards *et al.* (2000)	Evaluated the effect of compost (produced from manure, potato and sawdust) on potato growing soils by incorporating at 15 t ha^{-1} (dry matter).	Soil loss from compost amended plots was similar to that from control plots.
Bresson *et al.* (2001)	Investigated the impact of urban compost application (15 g kg^{-1}) on soil erosion in a silty loam exposed to a 19 mm h^{-1} rainfall for 60 minutes. Application rate is approximately 39 t ha^{-1} with incorporation up to 20 cm.	Compost application reduced the sediments in the runoff from 36.4 to 11 g L^{-1} (by 69.7%).
Ros et al. (2001)	Conducted a two year field experiment to evaluate the effect on soil erosion of adding different urban organic waste, urban compost, unstabilized municipal waste, and aerobic sewage sludge on soils with 15% slope. The organic wastes were added to the soil in sufficient amount to increase the soil carbon content by 2%, and were incorporated into the top 20 cm layer (approximately 215 t ha^{-1} of compost).	Organic waste addition reduced the soil loss by 78, 80, and 94% respectively by unstabilized municipal waste, aerobic sewage sludge and municipal waste compost compared to the control. Recommended to use a single application at 250-300 t ha^{-1} to control erosion in susceptible lands.

7.8.3 LCI data and application scenario

On the basis of the above literature review, it could be assumed that an application of a 10 cm surface mulch would cut off the soil loss through erosion totally in vineyards. As in the section on water use (Section 7.2), the available data on reduced soil loss due to the incorporation of compost has been grouped to obtain a relationship between the application rates and percentage reduction in soil loss and shown in Figure 7.10. According to the Figure 7.10, an application of compost at 12 t ha^{-1} would reduce the soil loss due to erosion by 14.8% in cotton fields.

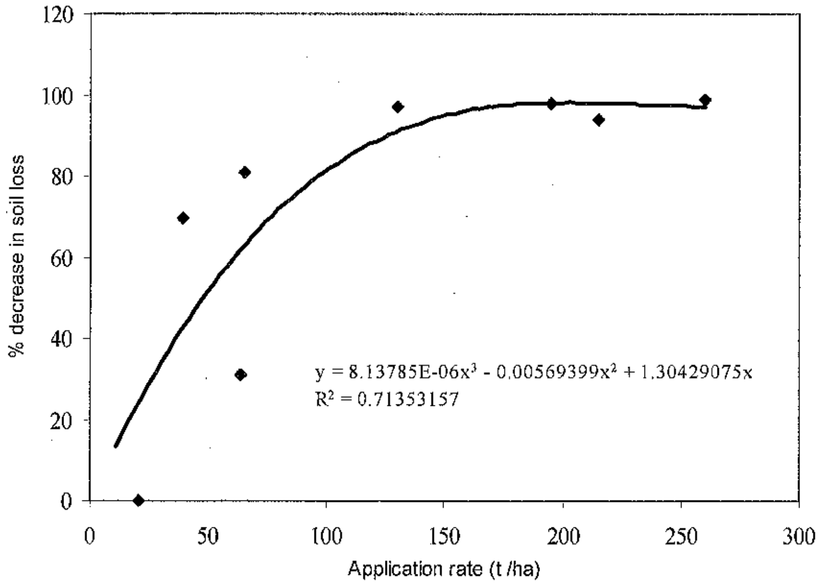
While extensive information available on erosion rates for cotton and grape growing soils in many regions of the world, comparatively limited information is available for Australia. Hamblin (1996), while reviewing Australian land resources indicated that the economic loss resulting from small amounts of erosion from high value horticultural areas greatly exceeds that coming from much larger areas of pastoral and arable lands in New South Wales. In this study, therefore, the average values for soil loss under cotton and grape cultivation estimated from existing data are used.

However, the erosion rates can vary greatly due to the variability in climate, soil type and the management. In cotton growing soils the run off varies typically between 100–200 mm and the soil loss varies between 4–12 t ha^{-1} per year (Silburn *et al.*, 1998). In this study, annual rate of soil erosion in conventionally managed Australian cotton growing soils is estimated as 8 tons per hectare by averaging existing data (Silburn and Glanville, 2002; Connolly *et al.*, 1999; Silburn *et al.*, 1998; Kennedy, 1997; Silburn *et al.*, 1997; Simpson, 1997; Waters, 1997; Holden, 1995; Carroll *et al.*, 1991). Therefore, an application of composted soil conditioner at 12 t ha^{-1} could result in preventing a soil loss of 1.2 tonnes (8 t of soil*14.8%) per hectare annually in cotton fields. Connolly *et al.* (1999) have reported slightly higher soil erosion (1.7 t ha^{-1} yr^{-1}) in cotton fields practicing stubble retention compared to the potential when soil conditioner is applied. According to Brown *et al.* (1985), cotton yields can decline by as much as 4% for each centimetre of topsoil loss.

In this study we assume that the average annual soil loss in vineyards in NSW due to erosion is 17.5 t ha^{-1}. This figure was an average soil loss measured in 19 vineyard blocks in Pokolbin, NSW during an investigation carried out by Loughran *et al.* (2000). Therefore, mulching at 10 cm depth could prevent a soil loss of 17.5 tonnes per hectare annually in vineyards because mulch cover will reduce the direct impact of rainwater dislodging the soil layer. Atech Group (2000) estimated that annual loss of N and P in Australian cotton fields due to runoff is about 0.5 kg and 4.0 kg per hectare respectively. In addition, they estimated that in Australian vineyards (perennial horticulture), there is a loss of 1.4 kg of P and 14 kg of N per hectare annually through runoff. It was also reported that an average of 3% of applied agrochemicals is lost through runoff (Sheehan *et al.*, 1998).

Therefore, an application of compost as a soil conditioner in cotton lands at 12 t ha^{-1} could result in reduction of soil loss by 1.2 tonnes, total phosphorus by 0.07 kg, and total nitrogen by 0.6 kg per hectare annually (based on 14.8% reduction in soil loss). Whereas, mulching of vineyard at 10cm would prevent a soil loss of 17.5. tons, total P loss of 1.4 kg, and total N loss of 14 kg per hectare annually.

Figure 7.10 Effect of compost as a soil conditioner on soil loss.



7.9 Soil structure

7.9.1 Introduction

The degradation of soil physical conditions such as soil structure is strongly related to continuos and intensive cultivations that have caused a gradual decrease of the soil organic matter (Giusquiani et al., 1995). Loss of soil structure is one of the significant land degradation problems facing most Australian agricultural soils leading to soil compaction, surface crusting, reduced water permeability, soil erosion and evidently reduced plant productivity. Numerous researchers have reported improvement in soil structure through the application of composted products (Pagliai et al., 1981; Guidi et al., 1983) and attributed this change to an increase in organic matter from the addition of composts.

A significant beneficial effect of compost applications to improvement of soil structure is due to increased stability of soil aggregates (Stratton et al., 1995). The high organic matter content in compost and the resultant effects of the organic matter on humic fractions and nutrients in soil increases microbial populations, activity and enzyme production, which in turn increases aggregate stability (Stratton et al., 1995). As the stability of soil aggregates increases, bulk density is decreased and porosity is increased (Mays et al., 1973). The modifications of soil structure have often been quantified through the characterisation of porosity and pore size distribution, since pores determine the most important soil physical properties for plant growth (Giusquiani et al., 1995). The variation of soil porosity directly reflects the soil structure and an increase in soil porosity due to application of composted products leads to a decrease of bulk density (Giusquiani et al., 1995). Therefore soil bulk density can also be used as indicator of soil structure.

Stratton et al. (1995) has reported that the most dramatic bulk density response to the application of compost has been on marginal soils with poor soil structure and low levels of soil organic matter. Changes reported in soil bulk density though have occurred for low compost applications but more pronounced effects have been produced for higher application rates (Stratton et al., 1995) and these effects are realised slowly over time.

7.9.2 Literature review

Table 7.14 shows performance data collected from literature on the effect of compost incorporation on soil bulk density and provides a base for the life cycle inventory of the effect.

Table 7.14 Effect of compost addition on soil bulk density

Study	Description	Impact
Turner et al. (1994)	Urban compost was applied to fine sand at the rate of 0, 67 and 134 t/ha for two years.	Application of 134 t/ha of compost after two years decreased bulk density from 1.3 to 1.1 g/cm^3.
Martens and Frankenberger, 1992	Various organic based wastes (poultry manure, sewage sludge, barley straw and green alfafa) were applied 3 times over 2 years at the rate of 25 t/ha/year to coarse loamy soil and incorporated into 15 cm topsoil.	Incorporation of total 75 t/ha of organic products over 2 years decreased bulk density from 1.48 to 1.32 g/cm^3.
Zebrath et al. (1999)	Various organic based wastes (biosolids, biowastes, poultry and food waste compost, composted hog manure solids, mined peat moss) were applied for 4 years at the rate of 45 t/ha/year to coarse textured loamy sand.	Incorporation of total 180 t/ha of organic products over 4 years decreased bulk density from 1.67 to 1.41 g/cm^3.
Stamatiadis et al. (1999)	Compost from green wastes, cow manure, spoiled hay, clay soil and various crop processing residues was applied at the rate of 0, 22 and 44 t/ha to broccoli grown on silt clay loam.	Compost application at the rate of 44 t/ha decreased bulk density after 1 year application from 1.15 to 1.08 g/cm^3.
Mays et al. (1994)	Compost from municipal refuse and sewage sludge was applied to sorghum, bermudagrass and corn at the rate from 0 to 448 t/ha	Incorporation of 327 t/ha of compost over 3-4 years significantly decreased bulk density of silt loam soil from 1.37 to 1.12 g/cm^3.
Illera et al. (1999)	Anaerobically digested biosolids and composted organic fraction of MSW was applied at the rate of 0 to 80 dry t/ha.	After 1 year application of 80 dry t/ha compost bulk density of degraded arid soil decreased from 1.22 to 1.06 g/cm^3.
Aggelides and Londra (2000)	Compost produced from a mixture of 17% sawdust, 21% sewage sludge and 60% town wastes by volume was incorporated to loamy and clayey soils at the rate of 0, 39, 78 and 156 t/ha	After 1 year incorporation of 156 t/ha of composted soil conditioner bulk density of loamy soils decreased from 1.37 to 1.1 and clayey soils from 1.12 to 0.94 g/cm^3.

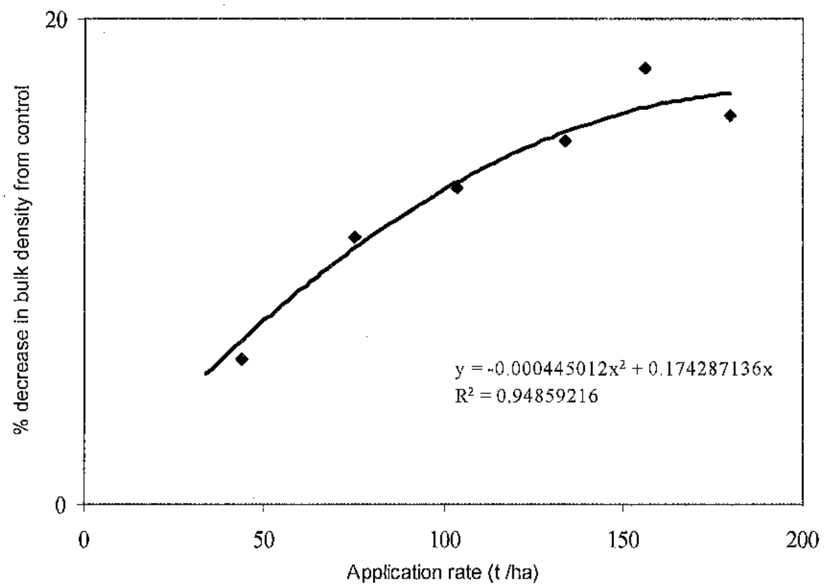
Data on changes in soil bulk density have been reported only for composted soil conditioners i.e. when composted products are incorporated into soil. The reason for this is that changes in soil physical properties including bulk density occur only when organic matter present in composted products interacts with the soil.

Application of composted products as surface mulch will also change soil bulk density only after mulch interacts with the soil, which happens slowly as compared to soil conditioner. Therefore, no bulk density data was available for effects of composted products applied as surface mulch.

7.9.3 LCI data and application scenario

As mentioned above that higher compost application rates are more effective in improving soil physical condition, including soil bulk density. A relationship has been obtained by grouping available information on the effect of compost application on soil bulk density (Figure 7.11). The relationship shown in Figure 7.11 may change (i.e. level out) for higher compost application rates over 200 t ha^{-1}. On the basis of this relationship, it is expected that an incorporation of 12 t ha^{-1} could reduce soil bulk density by about 2%.

Figure 7.11 Effect of compost as a soil conditioner on bulk density of the topsoil.



7.10 Plant response to compost applications

Improvements in the physical, chemical, and microbiological characteristics of soil from addition of composted products are well documented and explained in previous sections. The beneficial effects of compost addition are usually reflected through increased crop production resulting from enhanced seedling emergence, improved and healthier plant growth, and increased yield.

Although compost application can often result in increased crop yields, the response varies with compost feedstock and maturity, rates and methods of application, crop species and variety, fertiliser rates and intervals between application and planting. Yield responses are generally greater in soils with low fertility. Numerous reviews (Roe, 1998; Shiralipour *et al.*, 1992; Gallardo-Lara and Nogales, 1987) and research findings on the beneficial effect of compost application on agronomic and horticultural crop production have been published. Table 7.15 provides summarised information from various investigations on compost use and crop production.

Table 7.15 Summary of compost applications to plants/crops effects.

Study	Description	Impact
Manios and Syminis (1988)	Studied the effect of urban compost amendment (15 and 30 t ha^{-1}) on cucumber production.	Yield was increased by 17.6% and 20.8% with 15 and 30 t ha^{-1} treatments respectively.
Bevacqua and Mellano (1994)	Investigated the effect of composted sludge on onion, turf grass and lettuce production on a sandy loam. Sludge compost was incorporated at 37 and 74 t ha^{-1}.	Composted sludge application at 37 t ha^{-1} increased the yield of onion, lettuce and turf grass by 46, 56, and 235% respectively over the control.
Obreza and Reeder (1994)	Evaluated the growth and yield response of a tomato/watermelon successional cropping system to incorporated urban compost (at 13 – 112 t ha^{-1}) on a sandy soil.	Observed 16% increase in tomato yield and 54% increase in watermelon yield compared to the control.
Sabrah *et al.* (1995)	Investigated the effect of applying urban compost at different rates (16.5, 33, 49.5 and 66 t ha^{-1}) as a soil amendment on field grown wheat.	Application of 33 t ha^{-1} of composted urban was found to be the most economical one giving 26 – 34% yield increase over the control.
Aguilar *et al.* (1997)	Evaluated the effect of urban compost at 11.2–45 t ha^{-1} (incorporated) on olive and orange production in sandy loam to clay loam soils.	Compost application resulted in an average yield increase of olive and orange crops by 50 and 17% respectively compared to the control.
Hallmark and Brown (1999)	Studied the effect of urban compost addition (20–100 tons per acre) on sugarcane production on a silty clay soil.	Incorporating large amounts of compost into soil before planting can result in large increase in sugarcane yields when inorganic fertiliser is not used.
Clark *et al.* (2000)	Evaluated the effects of incorporated urban compost (134 t ha^{-1}) on yield of irrigated tomato and pepper production on sandy soils.	Compost application resulted in increasing tomato yield by 18-27% and pepper yield by 17% over the control.
Ouédraogo *et al.* (2001)	Investigated the influence of compost (household refuse + manure + crop residue) amendment (5 t ha^{-1}) on sorghum grown on a loamy sand.	Compost application increased the sorghum yield by 45% over the control.

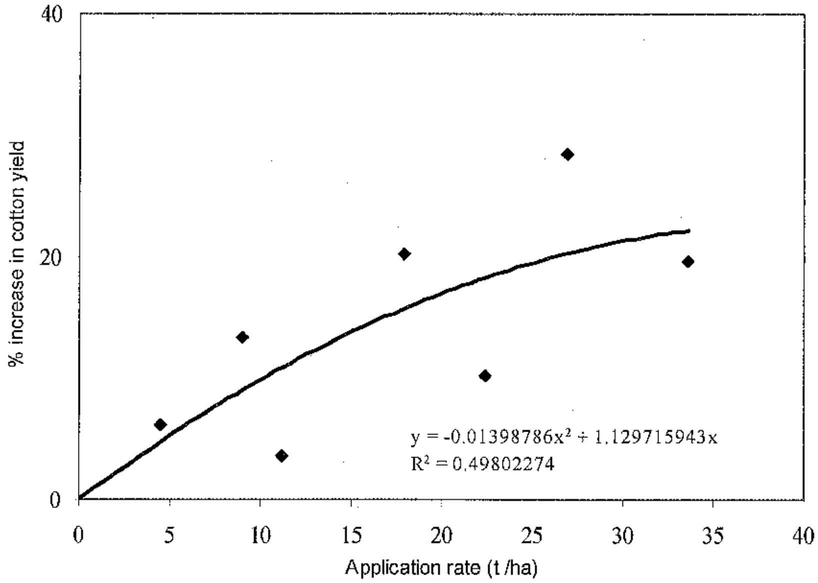
7.10.1 LCI data and application scenario

Information on productivity improvement in cotton and grape by compost addition is very limited. A summary of some findings on the effect of compost application on cotton and grape production is given in Table 7.16. A relationship is obtained by grouping available information on the effect of compost application on cotton yield as depicted in Figure 7.12. The relationship obtained here is assumed to be holding for the purpose of this study. According to the relationship, an application of 12 tons of composted soil conditioner products in a hectare would expect to increase the cotton yield by 11.5%. We also assume that an application of 10 cm composted mulch in a vineyard would increase the grape yield roughly by 27% as Buckerfield (2001) obtained an average figure of 35% and 19% yield increase for 7.5 and 15 cm composted mulch measured over 4 harvests after application of mulch to 2 years old vines in South Australia.

Table 7.16 Summary of effect of compost application on cotton and grape production.

Study	Description	Impact
Aguilar *et al.* (1997)	Evaluated the effect of urban compost at 30 tha^{-1} (incorporated) on grape production in unirrigated sandy loam to clay loam soils.	Produced an average increase in yield of 30% over the control.
Buckerfield (1998)	Investigated the effect of 10 cm composted mulch (composted green organics) on grape yield under 800 vines in South Australia.	Observed 20.5% increase in grape yield (kg/vine) due to compost mulching. Berry weight and sugar content were also increased.
Pinamonti (1998)	Studied the effect of addition of two composts (urban compost and sludge and bark compost) as a 5 mm mulch on grape production.	The compost mulch improved the general performance and the growth of vines. The pruning weight was 120 – 140% higher than that in the control.
Khalilian *et al.* (2002a)	Evaluated the effect of addition of urban compost at 11.2, 22.4 and 33.6 t ha^{-1} as banded and broadcast application on cotton in a 3 year field experiment on loamy sand soil.	Observed average yield increase of 3.6, 10.2, and 19.7% with 11.2, 22.4 and 33.6 t ha^{-1} treatments respectively.
Khalilian *et al.* (2002b)	Conducted a 3 year field experiment to evaluate the effect of addition of urban compost at 8.98, 17.94, and 26.90 t ha^{-1} on cotton production.	The compost application resulted in average yield increase of 13.3, 20.2, and 28.5% with 8.98, 17.94, and 26.90 t ha^{-1} treatments respectively.
CIWMB (2002)	Studied the effect of compost application (2-8 tons per acre) in vineyards.	8 tons per acre treatment increased the grape yield by 0.9 tons per acre.
Rabb *et al.* (2002)	Evaluated the effect of compost application at 4.5 t ha^{-1} with various rates of N fertilizer on cotton production.	Observed 6.2% average yield increase over the control due to compost application.

Figure 7.12 Effect of compost on cotton yield.



7.11 Summary of post-application impacts

The summaries of post-application impacts resulting from applying composted soil conditioner in an irrigated cotton field and composted mulch in a vineyard are given in Tables 7.17 and 7.18 respectively. Section 8 modelling and subsequent discussions suggest that 25-50 t ha^{-1} of soil conditioner is the relevant application rate (depending on soil, climate, etc.), therefore summary results below are calculated from this range of applications (Table 7.17). Post-application impacts of composted mulch have been calculated for 10 cm mulching depth (Table 7.18). The results in both tables refer to single application and per year impact.

Table 7.17 Summary of post-application impacts resulting from applying a composted soil conditioner at 25-50 t ha^{-1} (wet weight basis) in an irrigated cotton field.

Impact	**Tangible (potential) benefits**
Water use (Section 7.2.8)	• Increasing water holding capacity of top 0-15 cm soil layer by 2.4-3.0 %. • Savings of 0.13-0.16 ML of water per hectare per season in irrigated cotton • Savings of 37,006-45,370 ML of water to cotton industry per season of irrigated cotton.
Fertiliser use (Section 7.3.4)	• Savings of 34-68 kg of N, 29-57 kg of P, and 24-48 kg of K per hectare during the first year of application • Savings of 90-180 kg of N, 75-150 kg of P, and 30-60 kg of K per hectare during 3-5 years.
Herbicide use (Section 7.4.4)	• Soil conditioners do not suppress weeds.
Biocide use (Section 7.5.3)	• Potential to control Fusarium wilt in cotton, which is spreading at epidemic levels in NSW. There is no known biocide that can be replaced with compost.
Carbon sequestration (Section 7.6.2)	• Sequestering about 2.9-5.9 tons of carbon per hectare after 10 years. This figure may be taken for carbon accounting process. • Sequestering about 1.8-3.5 tons of carbon per hectare after 100 years. This figure is relevant to LCI studies.
Sodicity (Section 7.7.3)	• Potential savings of 2-5 tonnes of gypsum per hectare in affected lands. No such data available yet.
Erosion (Section 7.8.3)	• Preventing a soil loss of 2.3 to 4.2 tonnes per hectare annually.
Soil structure (Bulk density) Section 7.9.3	• Bulk density decreased by 4.1 to 7.6%
Yield response (Section 7.9.1)	• Increasing the cotton yield by 19.5 to 21.5%

Table 7.18 Summary of post-application impacts resulting from applying a composted mulch at 10 cm depth in a vineyard (roughly 100 t ha^{-1} of material at bulk density of 510 kg m^{-3}).

Impact	**Tangible (potential) benefits**
Water use (Section 7.2.8)	• Increasing moisture retention of top 0 – 15 cm soil layer by 9.82%. • Savings of 0.95 ML of water per hectare per season in irrigated viticulture, if it is available to plants. • Savings of 23,750 ML of water per season of irrigated viticulture.
Fertiliser use (Section 7.3.4)	• Savings of 27-40 kg of N, 46-68 kg of P, and 72-108 kg of K per hectare during the first year of application • Savings of 72-108 kg of N, 120-180 kg of P, and 90-135 kg of K per hectare during 3-5 years after application.
Herbicide use (Section 7.4.4)	• Replacing the total herbicide requirement in a vineyard, i.e. 2-6 L of glyphosate per hectare.
Biocide use (Section 7.5.3)	• No data available
Carbon sequestration (Section 7.6.2)	• Sequestering about 11.56 tons of carbon per hectare after 10 years. This figure may be taken for carbon accounting process. • Sequestering about 7.056 tons of carbon per hectare after 100 years. This figure is relevant to LCI studies.
Salinity and sodicity (Section 7.7.3)	• No data available.
Erosion (Section 7.8.3)	• Preventing a soil loss of 17.5 tonnes per hectare annually.
Soil structure (Bulk density) Section 7.9.3	• No data available.
Yield response (Section 7.9.1)	• Increasing the grape yield by 27%.

7.12 Limitations and gaps of post-application impacts

The main limitation of post-application impacts is that the majority of data for these impacts have been obtained from overseas studies. Also performance does not necessarily reflect the level of performance that could be realized from appropriate application rate of products that are specifically tailored to the specific agricultural application context. Though relationships and trends identified for these impacts in this study will most likely be similar for Australian agriculture, quantitative impacts may not be entirely appropriate for Australian conditions. Australian agriculture (particularly soil and climate conditions) is different from where (Europe and North America) most of the data in this study has been obtained.

Soil and climatic conditions play an important role in determining post compost application impacts. As mentioned in Section 2.1, significant reduction of soil organic matter in Australian soils due to continuous cropping has reduced soil fertility, productivity and stability. Land degradation has been identified as the most severe land use issue affecting the state of the environment in NSW (NSW EPA, 1997). In this context, application of composted products to highly degraded and low productive soils has the potential to produce even greater direct and indirect beneficial environmental impacts. The gaps in LCI data for post-application impacts identified from this review are listed below:

Many studies have found increased soil water content or soil water retention after application of composted soil conditioners. Very few studies have measured changes in plant available water. It is the increase in plant available water that provides an opportunity for and quantification of the reduction in irrigation water requirements. There are no data available for change in plant available water after application of composted mulches. Such data are required to quantify the extent to which the application of composts increase plant available water and reduce irrigation water requirements.

Recycled organics product are source of major plant nutrients particularly nitrogen, phosphorus and potassium which has been demonstrated in several studies. Recycled organics products release nutrients variably and steadily over 3-5 years after the decomposition of organic matter present in the RO products. Mineraliasation of RO products, and release and availability of major nutrients for cotton and grapevine use under Australian environmental conditions are not known. This information is critical for accurate estimation of nutrient contribution to meet requirements of these crops through RO product applications. Long term data on nutrient release and availability for all three major plant nutrients (N, P and K) is necessary in this regard.

Several studies have identified that composted products have disease suppressive effects. Disease suppression must be relevant to the specific crop and region being evaluated. However there is no current evidence that composts will be effective in suppressing diseases and displacing the use of biocide known to be used in cotton or grapevine cultivation in NSW to control diseases.

This literature review has suggested that application of composted soil conditioners has the potential to reduce soil sodicity of the topsoil layer due to improvement in soil structure, hence reduced use of gypsum. However there are no data available as to how much gypsum can be replaced with composted soil conditioners to ameliorate sodic soils. There is also no information available on the effect of compost as mulch on sodicity.

Application of composted soil conditioner has been shown to lead to decrease in soil bulk density, an indicator of improved soil structure, however there are no data available for use of mulch on soil bulk density. Mulch applied as surface cover does not interact with the soil like soil conditioners therefore not directly decrease bulk density as do soil conditioners. However mulch does impact on the levels of biological activity in the topsoil, and eventually would interact directly with the soil over time as the mulch breaks down to improve soil bulk density and soil structure.

In the case of composted mulches, post-application performance beneficial relationships have been derived for mulch application depth (optimum depth assumed in this study is 10 cm) without considering mulch application width because benefits from mulch application in the literature are cited mainly for application depth. Though mulch application depth is responsible for producing post-application benefits, changing mulch application width changes mulch application rate. This certainly alters nutrient provision and is likely to change post-application benefits. In this study, varying mulch application rates due to varying application width take into account post-application benefits such as reduction in fertilizers and increased carbon sequestration. However due to lack of such information, varying mulch application rates due to mulch application width did not take into account impacts on other benefits, e.g. water and herbicide savings, reduced erosion, improved soil structure, and improved plant productivity.

In quantifying the impacts and benefits of compost, there is strictly limited value in assessing the application of generic compost products. Products assessed should be relevant to the needs of the crop, soil and region, and should be applied at an optimum rate to achieve maximum benefit. The establishment of performance data from such directly relevant applications will markedly improve the quality of LCI data, and provide a more accurate indication of the associated post-application environmental impacts.

Section 8 Life Cycle Impact Assessment

Life Cycle Impact Assessment (LCIA) is used to characterise and assess the affects of resource consumption and environmental loadings identified in the inventory stage (see Section 3.2.3). The calculation involves the conversion of LCI results to common units and the aggregation of the converted results within the impact category. This conversion uses characterisation/equivalency factors. Characterisation/equivalency factors are developed from characterisation model relevant to environmental mechanism of each impact category. Characterisation/equivalency factors have been developed within the LCA framework to identify how much a substance contributes to a particular environmental impact category compared with a reference substance. For example, methane contributes 21 times more to global warming than carbon dioxide. Carbon dioxide is used as a reference substance for GWP. Thus the quantity of methane released by a system is multiplied by 21 to derive a carbon dioxide equivalent global warming figure. The relevant environmental issues (impact categories) are modelled in the LCIA and the results of this modelling reported in the context of these impact categories.

A LCIA calculator has been constructed as an Excel spreadsheet (Excel 2000) to model LCI data collected from the literature for the production and application of composted products. This calculator calculates and presents total environmental impacts (positive and negative) from the compost system as defined in system model and application scenarios documented in Section 4 of this report. The LCIA calculator also provides an option for inclusion of economic value modelling in future when such data become available. The LCIA calculator is also adaptable for future use by simple alteration of compost quantities, application rates and transport distance. Detailed instructions for use of the LCIA calculator are provided in an instructions worksheet (worksheet 2) in the attached calculator (ROU, 2003).

8.1 Modelling scenarios

The following modelling scenarios have been considered to assess environmental impacts of production of composted products and post-applications to grapevine (intensive) and cotton (broadacre) agricultural production systems using this calculator. Assumptions, results, sensitivity analysis and interpretation of these modelling scenarios are presented below:

8.1.1 Production and application of composted mulch to intensive (grapevine) agricultural production system

Modelling scenarios 1 to 3-Assumptions

Composted mulch produced in a commercial windrow composting facility system in the Sydney metropolitan area is transported to agricultural (viticulture) application sites for application as surface mulch. Prominent grapevine growing areas located in the Hunter (distance 175 km) and Mudgee (distance 275 km) are modelled to assess impacts arising from varying transport distance. Trucks return with backloading from both grape growing regions.

In scenarios 1 and 2, mulch is applied (in each of the regions) along the rows of grapevine at the same application rate of 10 cm depth and 50 cm width, which is equivalent to 83 t ha^{-1} (Table 8.1).

In modelling scenario 3, a higher mulch application rate for the Hunter region has been considered. In scenario 3, mulch application width is 75 cm (as opposed to 50 cm wide in scenarios 1 and 2), while mulch depth remains the same 10 cm. Total application rate in Scenario 3 is equivalent to 125 t ha^{-1}.

These three scenarios model the impacts of two variable transport distances (175 and 275 km), and varying application rates (83 and 125 t ha^{-1}) whilst keeping backloading as a constant. Backloading in this study means trucks transporting composted products to application sites return (from the region) loaded with other products unrelated to this system (for details please see Section 6).

8.1.2 Production of composted soil conditioner and application to broadacre (cotton) agricultural production system

Modelling scenario 4 to 7-Assumptions

Composted soil conditioner produced in a commercial windrow composting facility in the Sydney metropolitan area is transported to an agricultural (cotton farm) application site for use. The application site is the prominent cotton growing area located at Narrabri (constant transport distance of 600 km). In scenarios 4-7 two logistic scenarios (backloading and without backloading) and two application rates of 25 t ha^{-1} (~2.5mm depth) and 50 t ha^{-1} (~5mm depth) have been modelled. Soil conditioners are applied with compost spreader and incorporated into the soil in the process of normal soil cultivation practices. These scenarios model the impacts of varying backloading and non-backloading, and varying application rates whilst keeping transport distance constant (600 km).

Backloading in this study means trucks transporting composted products to application sites return (from the region) loaded with other products unrelated to this system. Without backloading or non-backloading means trucks transporting composted products to application sites return empty (for details please see Section 6).

Total area of application (hectares) of composted products for all scenarios (1 to 7) is the same and equals the 100 ha. All modelling scenarios are presented in Table 8.1.

Table 8.1. Modelling scenarios for assessing environmental impacts of a windrow composting system.

Modelling scenario	**Composted mulch**	**Crop type**	**Application rate**	**Application area**	**Transport distance (km)**	**Backloading/ without backloading**
Scenario 1	Composted mulch	Grape vine	10 cm depth and 50 cm width (83 t ha^{-1})	100 ha	Hunter (175)	Backloading
Scenario 2	Composted mulch	Grape vine	10 cm depth and 50 cm width (83 t ha^{-1})	100 ha	Mudgee (275)	Backloading
Scenario 3	Composted mulch	Grape vine	10 cm depth and 75 cm width (125 t ha^{-1})	100 ha	Hunter (175)	Backloading
Scenario 4	Composted soil conditioner	Cotton	25 t ha^{-1}	100 ha	Narrabri (600)	Backloading
Scenario 5	Composted soil conditioner	Cotton	25 t ha^{-1}	100 ha	Narrabri (600)	Without backloading
Scenario 6	Composted soil conditioner	Cotton	50 t ha^{-1}	100 ha	Narrabri (600)	Backloading
Scenario 7	Composted soil conditioner	Cotton	50 t ha^{-1}	100 ha	Narrabri (600)	Without backloading

This range of scenarios allows for evaluation and comparison of the key application variables, including application rate, transport distance, backloading/non-backloading. Note that scenarios 5 and 7 represent an extreme case of the greatest potential transport distance, with no backloading. This represents the worst case scenario possible in terms of environmental impacts from compost.

8.2 Results of modeling scenarios

8.2.1 How to read results of modelling

Please note when reading results of impact categories, negative numbers represent a net environmental benefit, or beneficial environmental impact. Therefore more negative results represent a greater environmental benefit.

Positive numbers represent no environmental benefit or detrimental environmental impact. Therefore more positive results represent more detrimental environmental impact.

The results refer to total impacts arising from production and application of total product on 100 ha of agricultural land.

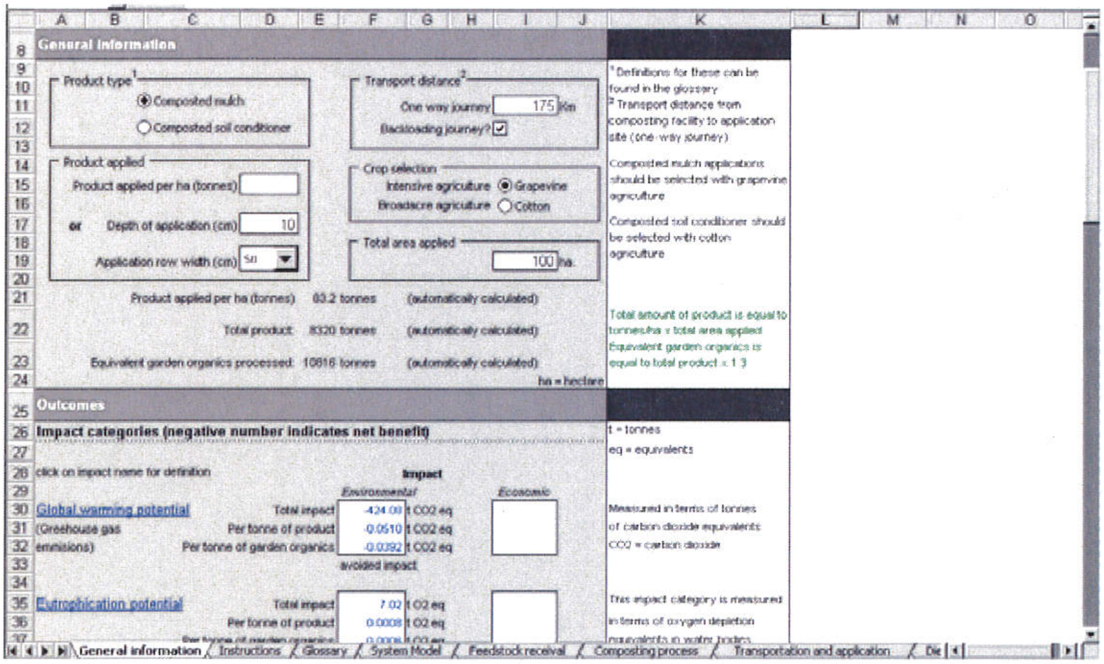
8.2.2 Production of composted mulch and application to grapevine

The LCIA results of modelling of the environmental impacts and avoided loads for production and application of composted mulch to grapevine are presented in Figure 8.1 and Table 8.2.

Please note LCI data of many post-application environmental performance benefits could not be assessed because relevant relationships and estimates of data were not available. Therefore some post-application performance benefits have been presented as un-characterized avoided loads and not category impact potentials.

Figure 8.1. Environmental impacts and avoided loads of production and application of composted mulch to grapevine. Note the relevant mulch application rate; varying transport distances and selection of backloading for each scenario as selected in the LCIA calculator for compost systems.

Figure 8.1a – Scenario 1-Application of 83 t/ha of composted mulch to grapevine at 175 km distance.



General Information

Product type[1]
● Composted mulch
○ Composted soil conditioner

Transport distance[2]
One way journey 175 Km
Backloading journey? ☑

Product applied
Product applied per ha (tonnes)
or Depth of application (cm) 10
Application row width (cm) 50

Crop selection
Intensive agriculture ● Grapevine
Broadacre agriculture ○ Cotton

Total area applied
100 ha

Product applied per ha (tonnes) 83.2 tonnes (automatically calculated)
Total product 8320 tonnes (automatically calculated)
Equivalent garden organics processed 10816 tonnes (automatically calculated)
ha = hectare

[1] Definitions for these can be found in the glossary
[2] Transport distance from composting facility to application site (one-way journey)

Composted mulch applications should be selected with grapevine agriculture

Composted soil conditioner should be selected with cotton agriculture

Total amount of product is equal to tonnes/ha x total area applied
Equivalent garden organics is equal to total product x 1.3

Outcomes

Impact categories (negative number indicates net benefit)

click on impact name for definition

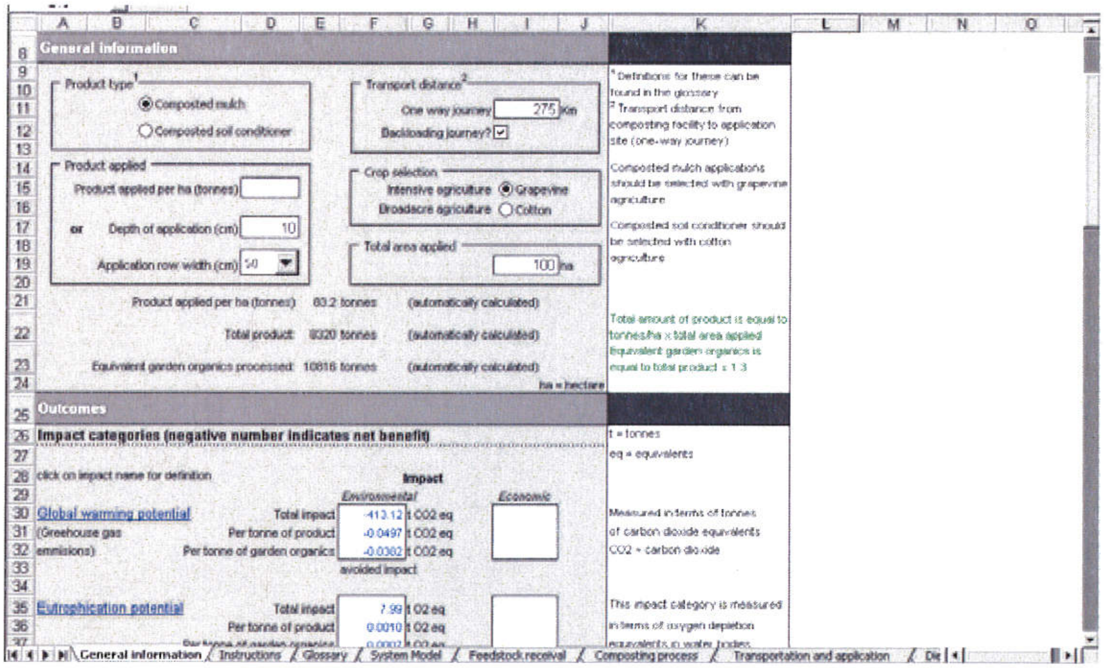
		Impact Environmental		Economic
Global warming potential (Greenhouse gas emmisions)	Total impact	-424.00	t CO2 eq	
	Per tonne of product	-0.0510	t CO2 eq	
	Per tonne of garden organics	-0.0392	t CO2 eq	
		avoided impact		
Eutrophication potential	Total impact	7.02	t O2 eq	
	Per tonne of product	0.0008	t O2 eq	

t = tonnes
eq = equivalents

Measured in terms of tonnes of carbon dioxide equivalents
CO2 = carbon dioxide

This impact category is measured in terms of oxygen depletion equivalents in water bodies

Figure 8.1b – Scenario 2-Application of 83 t/ha of composted mulch to grapevine at 275 km distance.



General information

Product type[1]
● Composted mulch
○ Composted soil conditioner

Transport distance[2]
One way journey 275 Km
Backloading journey? ☑

Product applied
Product applied per ha (tonnes)
or Depth of application (cm) 10
Application row width (cm) 50

Crop selection
Intensive agriculture ● Grapevine
Broadacre agriculture ○ Cotton

Total area applied
100 ha

Product applied per ha (tonnes) 83.2 tonnes (automatically calculated)
Total product 8320 tonnes (automatically calculated)
Equivalent garden organics processed 10816 tonnes (automatically calculated)
ha = hectare

[1] Definitions for these can be found in the glossary
[2] Transport distance from composting facility to application site (one-way journey)

Composted mulch applications should be selected with grapevine agriculture

Composted soil conditioner should be selected with cotton agriculture

Total amount of product is equal to tonnes/ha x total area applied
Equivalent garden organics is equal to total product x 1.3

Outcomes

Impact categories (negative number indicates net benefit)

t = tonnes
eq = equivalents

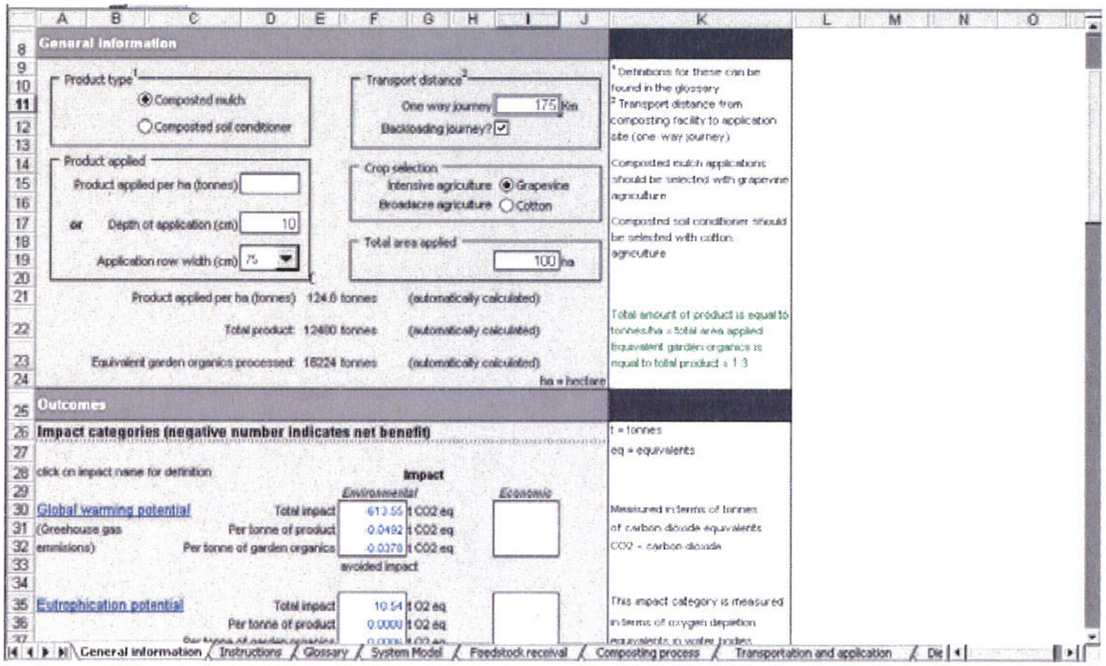
click on impact name for definition

		Impact *Environmental*		*Economic*
Global warming potential	Total impact	-413.12	t CO2 eq	
(Greenhouse gas	Per tonne of product	-0.0497	t CO2 eq	
emmisions)	Per tonne of garden organics	-0.0382	t CO2 eq	
		avoided impact		
Eutrophication potential	Total impact	7.99	t O2 eq	
	Per tonne of product	0.0010	t O2 eq	

Measured in terms of tonnes of carbon dioxide equivalents
CO2 = carbon dioxide

This impact category is measured in terms of oxygen depletion equivalents in water bodies

Figure 8.1c – Scenario 3-Application of 125 t/ha of composted mulch to grapevine at 175 km distance.



General information

Product type[1]
● Composted mulch
○ Composted soil conditioner

Transport distance[2]
One way journey 175 Km
Backloading journey? ☑

Product applied
Product applied per ha (tonnes)
or Depth of application (cm) 10
Application row width (cm) 75

Crop selection
Intensive agriculture ● Grapevine
Broadacre agriculture ○ Cotton

Total area applied
100 ha

Product applied per ha (tonnes) 124.8 tonnes (automatically calculated)
Total product 12480 tonnes (automatically calculated)
Equivalent garden organics processed 16224 tonnes (automatically calculated)
ha = hectare

[1] Definitions for these can be found in the glossary
[2] Transport distance from composting facility to application site (one-way journey)

Composted mulch applications should be selected with grapevine agriculture

Composted soil conditioner should be selected with cotton agriculture

Total amount of product is equal to tonnes/ha x total area applied
Equivalent garden organics is equal to total product x 1.3

Outcomes

Impact categories (negative number indicates net benefit)

t = tonnes
eq = equivalents

click on impact name for definition

		Impact *Environmental*		*Economic*
Global warming potential	Total impact	-613.55	t CO2 eq	
(Greenhouse gas	Per tonne of product	-0.0492	t CO2 eq	
emmisions)	Per tonne of garden organics	-0.0378	t CO2 eq	
		avoided impact		
Eutrophication potential	Total impact	10.54	t O2 eq	
	Per tonne of product	0.0008	t O2 eq	

Measured in terms of tonnes of carbon dioxide equivalents
CO2 = carbon dioxide

This impact category is measured in terms of oxygen depletion equivalents in water bodies

Table 8.2 Modelling outcomes for Scenarios 1, 2 and 3: Environmental impacts and avoided loads of production and application of composted mulch to grapevine at same/varying transport distances and mulch application rates.

Impact Categories		**Environmental impacts of transport distances and mulch application rates.**		
		Scenario 1	**Scenario 2**	**Scenario 3**
		Hunter	**Mudgee**	**Hunter**
		83 t ha^{-1}	**83 t ha^{-1}**	**125 t ha^{-1}**
Global warming potential (t CO_2 eq)	Total impact	-424.08	-413.14	-613.55
	Per tonne of product	-0.0510	-0.0497	-0.0492
	Per tonne of garden organics	-0.0392	-0.0382	-0.0378
Eutrophication potential (t O_2 eq)	Total impact	7.02	7.99	10.54
	Per tonne of product	0.0008	0.0010	0.0008
	Per tonne of garden organics	0.0006	0.0007	0.0006
Human toxicity potential (t DCB eq)	Total impact	-0.62	-0.61	-0.93
	Per tonne of product	-0.0001	-0.0001	-0.0001
	Per tonne of garden organics	-0.0001	-0.0001	-0.0001
Ecotoxity potential-Terrestrial (t DCB eq)	Total impact	-6.06	-5.88	-9.08
	Per tonne of product	-0.0007	-0.0007	-0.0007
	Per tonne of garden organics	-0.0006	-0.0005	-0.0006
Ecotoxity potential-Freshwater (t DCB eq)	Total impact	-0.14545	-0.14545	-0.21818
	Per tonne of product	-0.00001	-0.00001	-0.00001
	Per tonne of garden organics	-0.00001	-0.00001	-0.00001
Ecotoxity potential-Marinewater (t DCB eq)	Total impact	0.00062	0.00062	0.00093
	Per tonne of product	0.00000	0.00000	0.00000
	Per tonne of garden organics	0.00000	0.00000	0.00000
Photochemical oxidation potential (t ethylene eq)	Total impact	0.65083	0.67939	0.97625
	Per tonne of product	0.00008	0.00008	0.00008
	Per tonne of garden organics	0.00006	0.00006	0.00006
Resource depletion potential (t antimony eq)	Total impact	0.16967	0.17768	0.25451
	Per tonne of product	0.00002	0.00002	0.00002
	Per tonne of garden organics	0.00002	0.00002	0.00002
Avoided loads		**Hunter**	**Mudgee**	**Hunter**
Water (ML)	Total impact	94.98	94.98	94.98
	Per tonne of product	0.01142	0.01142	0.00761
	Per tonne of garden organics	0.00878	0.00878	0.00585
Nitrogen (tonnes)	Total impact	2.808	2.808	4.212
	Per tonne of product	0.00034	0.00034	0.00034
	Per tonne of garden organics	0.00026	0.00026	0.00026
Phosphorus (tonnes)	Total impact	4.74	4.74	7.11
	Per tonne of product	0.00057	0.00057	0.00057
	Per tonne of garden organics	0.00044	0.00044	0.00044
Potassium (tonnes)	Total impact	7.49	7.49	11.23
	Per tonne of product	0.0009	0.0009	0.0009
	Per tonne of garden organics	0.0007	0.0007	0.0007
Herbicides/pesticides (litres)	Total impact	225	225	225
	Per tonne of product	0.0270	0.0270	0.0180
	Per tonne of garden organics	0.0208	0.0208	0.0139
Erosion (tonnes)	Total impact	1750	1750	1750
	Per tonne of product	0.1402	0.1402	0.2103
	Per tonne of garden organics	0.0543	0.0543	0.1618
Carbon sequestration (tonnes)	Total impact	587.06	587.06	880
	Per tonne of product	0.0706	0.0706	0.0706
	Per tonne of garden organics	0.0543	0.0543	0.0543
Soil structure (% decrease in bulk density)	For total application rate	11.42	11.42	11.82
Plant productivity (% increase in yield)	Total product per hectare	27	27	27

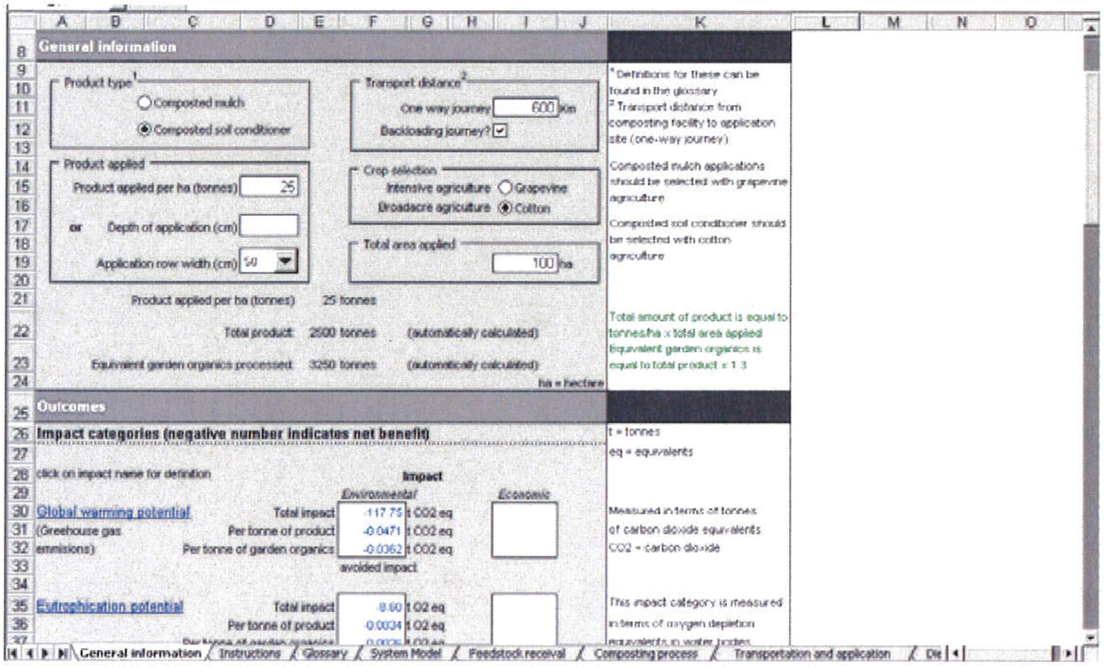
8.2.3 Production of composted soil conditioner and application to cotton production

The results of LCIA modelling of environmental impacts and avoided loads for production and application of composted soil conditioner to cotton are presented in Figure 8.2 and Table 8.3.

Please note LCI data of many post-application environmental performance benefits could not be assessed because relevant relationships and estimates of data were not available. Therefore some post-application performance benefits have been presented as un-characterized avoided loads and not category impact potentials.

Figure 8.2 Environmental impacts and avoided loads of production and application of composted soil conditioner to cotton production as per scenarios 4-7. Note the relevant application rate and transport distance are equal but backloading and non-backloading scenarios are modelled for comparison. (8.2a) backloading and (8.2b) without backloading.

Figure 8.2a – Scenario 4-Application of 25 t/ha of composted soil conditioner to cotton at Narrabri and trucks returning with backloading (Note, Scenario 6 is same but application rate of 50 t/ha)



Product type[1]
○ Composted mulch
● Composted soil conditioner

Transport distance[2]
One way journey: 600 Km
Backloading journey? ☑

Product applied
Product applied per ha (tonnes): 25
or Depth of application (cm):
Application row width (cm): 50

Crop selection
Intensive agriculture ○ Grapevine
Broadacre agriculture ● Cotton

Total area applied: 100 ha

Product applied per ha (tonnes) 25 tonnes
Total product 2500 tonnes (automatically calculated)
Equivalent garden organics processed 3250 tonnes (automatically calculated)
ha = hectare

[1] Definitions for these can be found in the glossary
[2] Transport distance from composting facility to application site (one-way journey)

Composted mulch applications should be selected with grapevine agriculture

Composted soil conditioner should be selected with cotton agriculture

Total amount of product is equal to tonnes/ha x total area applied
Equivalent garden organics is equal to total product x 1.3

Impact categories (negative number indicates net benefit)

click on impact name for definition

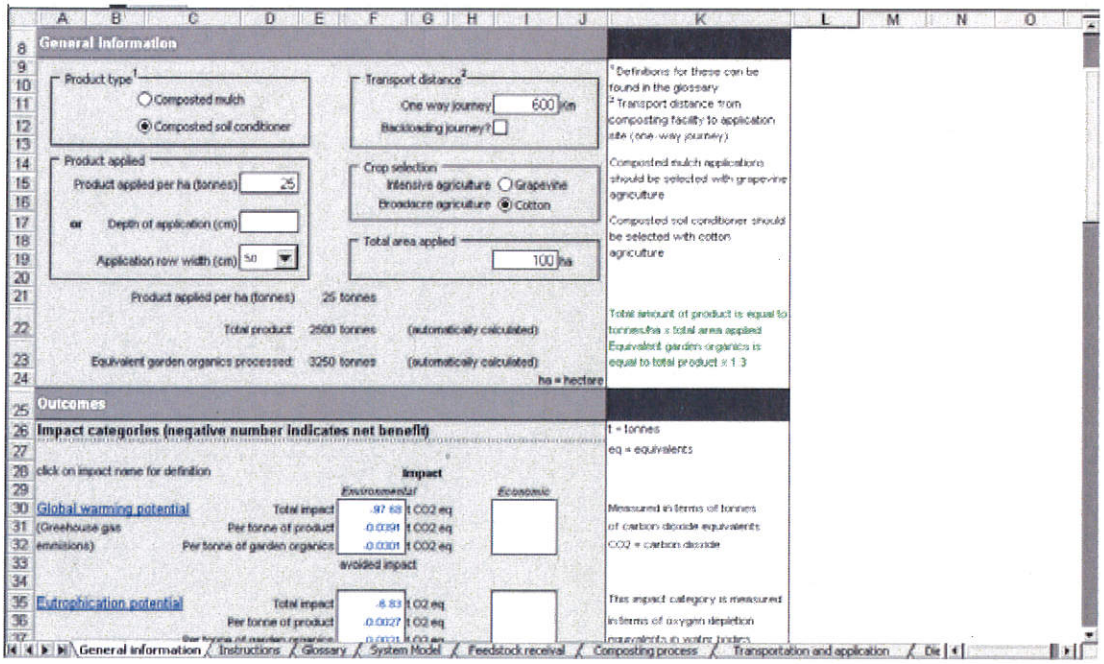
		Impact Environmental	Economic
Global warming potential (Greehouse gas emmisions)	Total impact	-117.75 t CO2 eq	
	Per tonne of product	-0.0471 t CO2 eq	
	Per tonne of garden organics	-0.0362 t CO2 eq	
		avoided impact	
Eutrophication potential	Total impact	-8.60 t O2 eq	
	Per tonne of product	-0.0034 t O2 eq	

t = tonnes
eq = equivalents

Measured in terms of tonnes of carbon dioxide equivalents
CO2 = carbon dioxide

This impact category is measured in terms of oxygen depletion equivalents in water bodies

Figure 8.2b – Scenario 5-Application of 25 t/ha of composted soil conditioner to cotton at Narrabri and trucks returning without backloading (Note, Scenario 7 is same but application rate of 50 t ha^{-1})



General Information

Product type[1]

○ Composted mulch

● Composted soil conditioner

Transport distance[2]

One way journey: 600 Km

Backloading journey? ☐

Product applied

Product applied per ha (tonnes): 25

or Depth of application (cm):

Application row width (cm): 50

Crop selection

Intensive agriculture ○ Grapevine

Broadacre agriculture ● Cotton

Total area applied: 100 ha

Product applied per ha (tonnes) 25 tonnes

Total product: 2500 tonnes (automatically calculated)

Equivalent garden organics processed: 3250 tonnes (automatically calculated)

ha = hectare

[1] Definitions for these can be found in the glossary

[2] Transport distance from composting facility to application site (one-way journey)

Composted mulch applications should be selected with grapevine agriculture

Composted soil conditioner should be selected with cotton agriculture

Total amount of product is equal to tonnes/ha x total area applied

Equivalent garden organics is equal to total product x 1.3

Outcomes

Impact categories (negative number indicates net benefit)

t = tonnes

eq = equivalents

click on impact name for definition

		Impact Environmental		Economic
Global warming potential (Greenhouse gas emissions)	Total impact	-97.68	t CO2 eq	
	Per tonne of product	-0.0391	t CO2 eq	
	Per tonne of garden organics	-0.0301	t CO2 eq	
		avoided impact		
Eutrophication potential	Total impact	-6.83	t O2 eq	
	Per tonne of product	-0.0027	t O2 eq	

Measured in terms of tonnes of carbon dioxide equivalents

CO2 = carbon dioxide

This impact category is measured in terms of oxygen depletion

Table 8.3 LCIA modelling outcomes for scenarios 4-7. Environmental impacts and avoided loads for production and application of composted soil conditioner to cotton at same transport distance (Narrabri), for different application rates and backloading and without backloading.

Impact Categories		Environmental impacts of soil conditioner application rates and backloading and non-backloading			
		25 t ha^{-1}		50 t ha^{-1}	
		Scenario 4	Scenario 5	Scenario 6	Scenario 7
		Backloading	Without Backloading	Backloading	Without Backloading
Global warming potential (t CO_2 eq)	Total impact Per tonne of product Per tonne of garden organics	-117.75 -0.0471 -0.0362	-97.68 -0.0391 -0.0301	-231.19 -0.0462 -0.0356	-191.06 -0.0382 -0.0294
Eutrophication potential (t CO_2 eq)	Total impact Per tonne of product Per tonne of garden organics	-8.60 -0.0034 -0.0026	-6.83 -0.0027 -0.0021	-17.20 -0.0034 -0.0026	-13.65 -0.0027 -0.0021
Human toxicity potential (t DCB eq)	Total impact Per tonne of product Per tonne of garden organics	-0.38 -0.0002 -0.0001	-0.36 -0.0001 -0.0001	-0.76 -0.0002 -0.0001	-0.73 -0.0001 -0.0001
Ecotoxity potential-Terrestrial (t DCB eq)	Total impact Per tonne of product Per tonne of garden organics	-6.39 -0.0026 -0.0020	-6.06 -0.0024 -0.0019	-12.77 -0.0026 -0.0020	-12.13 -0.0024 -0.0019

Impact Categories		**Environmental impacts of soil conditioner application rates and backloading and non-backloading**			
		25 t ha^{-1}		**50 t ha^{-1}**	
		Scenario 4	**Scenario 5**	**Scenario 6**	**Scenario 7**
		Backloading	**Without Backloading**	**Backloading**	**Without Backloading**
Ecotoxity potential-Freshwater (t DCB eq)	Total impact	-0.1360	-0.1360	-0.2720	-0.2720
	Per tonne of product	-0.00005	-0.00005	-0.00005	-0.00005
	Per tonne of garden organics	-0.00004	-0.00004	-0.00004	-0.00004
Ecotoxity potential-Marinewater (t DCB eq)	Total impact	0.00018	0.00018	0.00037	0.00037
	Per tonne of product	0.00000	0.00000	0.00000	0.00000
	Per tonne of garden organics	0.00000	0.00000	0.00000	0.00000
Photochemical oxidation potential (t ethylene eq)	Total impact	0.23443	0.28672	0.46887	0.57344
	Per tonne of product	0.00009	0.00011	0.00009	0.00011
	Per tonne of garden organics	0.00007	0.00009	0.00007	0.00009
Resource depletion potential (t antimony eq)	Total impact	0.06326	0.07791	0.12652	0.15582
	Per tonne of product	0.00003	0.00003	0.00003	0.00003
	Per tonne of garden organics	0.00002	0.00002	0.00002	0.00002
Avoided loads					
Water (ML)	Total impact	13.82	13.82	16.94	16.94
	Per tonne of product	0.00553	0.00553	0.00339	0.00339
	Per tonne of garden organics	0.00425	0.00425	0.00261	0.00261
Nitrogen (tonnes)	Total impact	3.375	3.375	6.75	6.75
	Per tonne of product	0.00135	0.00135	0.00135	0.00135
	Per tonne of garden organics	0.00425	0.00425	0.00104	0.00104
Phosphorus (tonnes)	Total impact	2.85	2.85	5.70	5.70
	Per tonne of product	0.00114	0.00114	0.00114	0.00114
	Per tonne of garden organics	0.00088	0.00088	0.00088	0.00088
Potassium (tonnes)	Total impact	2.40	2.40	4.80	4.80
	Per tonne of product	0.0010	0.0010	0.0010	0.0010
	Per tonne of garden organics	0.0007	0.0007	0.0007	0.0007
Erosion (tonnes)	Total impact	35.01	35.01	62.40	62.40
	Per tonne of product	0.0140	0.0140	0.0125	0.0125
	Per tonne of garden organics	0.0108	0.0108	0.0096	0.0096
Carbon sequestration (tonnes)	Total impact	176.40	176.40	352.80	352.80
	Per tonne of product	0.0706	0.0706	0.0706	0.0706
	Per tonne of garden organics	0.0543	0.0543	0.0543	0.0543
Soil structure (% decrease in bulk density)	For total application rate	4.08	4.08	7.60	7.60
Plant productivity (% increase in yield)	Total product per hectare	19.5	19.5	21.52	21.52

8.3 Interpretation and sensitivity analysis

Life cycle inventory data for production and application of composted products for windrow composting systems in Australia has been derived from a detailed literature review which yielded sufficient data to parameterise relationships between application rates and post-application performance benefits (e.g. weed suppression) using a line of best fit method (Sections 5, 6 and 7). The literature review did not provide a range for LCI data (e.g. low to high) for impact categories and post-application performance benefits. Generally sensitivity analysis requires a data range. Due to unavailability of such data, sensitivity analysis in this study was undertaken to consider the affects of major factors on LCIA modelling. These factors are as follows:

- Compost product application rate,
- Compost product type,
- Transport distance to application sites,
- Backloading and without backloading for vehicles transporting composted products to application sites, and
- Total application area.

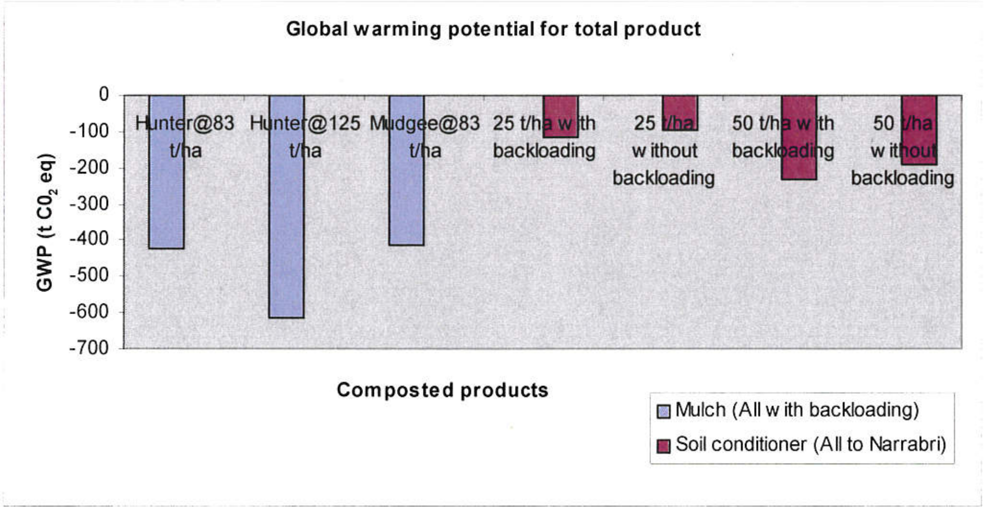
These factors have been varied in modelling scenarios and their affect on each impact category is presented and discussed below.

8.3.1 Impact categories

8.3.1.1 Global warming potential

Given Australians are among the highest emitters of greenhouse gases per capita in the world, global warming potential (GWP) is an important issue at national, state and local levels. As discussed in Section 3.3.2 global warming refers to the rise in temperature of the Earth's atmosphere as a result of a build up of greenhouse gases in the atmosphere, which absorb heat re-transmitted from the Earth's surface.

Figure 8.3. Global warming potential for production and application of total composted product on 100 ha of agricultural land (Note: Negative numbers represent a net environmental benefit. More negative results represent a greater environmental benefit).



Greenhouse gases are released during production and use of diesel fuel and electricity for production, transportation and application of composted products. The results of all modelling scenarios (1 to 7) for GWP show that production and application of composted products produced negative GWP (Figure 8.3), representing a net environmental benefit or beneficial environmental impact. The reason being that post-application benefits of composted products reduce requirements for fertilisers, electricity, and herbicides and increase carbon sequestration. These benefits reduce the release of greenhouse gases involved in the production and use of these items for agriculture resulting in net reduction in GWP. A net greenhouse benefit is realised even when composts are transported the most extreme distances for application, and the trucks return empty.

Higher application rates of composted products deliver a greater net environmental benefit with regard to GWP (Figure 8.3). Higher application rates per unit area produce a greater net environmental benefit. However, it should be mentioned that post-application benefits do not increase linearly with increasing application rates. At certain application rates, the rate of post-application benefits will tend to maximise suggesting a limit for reduction in GWP. This is logical, for example compost applications can only displace a limited quantity of fertiliser, as relevant to crop requirements.

Varying the transport distance by 100 km produced negligible impact on GWP (Figure 8.3). However, there will be a limit for transport distance beyond which GWP from transport outweighs the benefits of compost applications. Note, however, that transport of soil conditioner to the most distant area of the state still resulted in a net GWP benefit.

Backloading or without backloading scenarios hardly have any impact on GWP (Figure 8.3). Backloading scenarios for both composted soil conditioner rates deliver slightly greater net benefit for GWP.

These results identify that post-application environmental impacts of composted products deliver GWP benefit, and are responsible for producing a net GWP environmental benefit from a composting system.

Conclusions for global warming potential impact category:

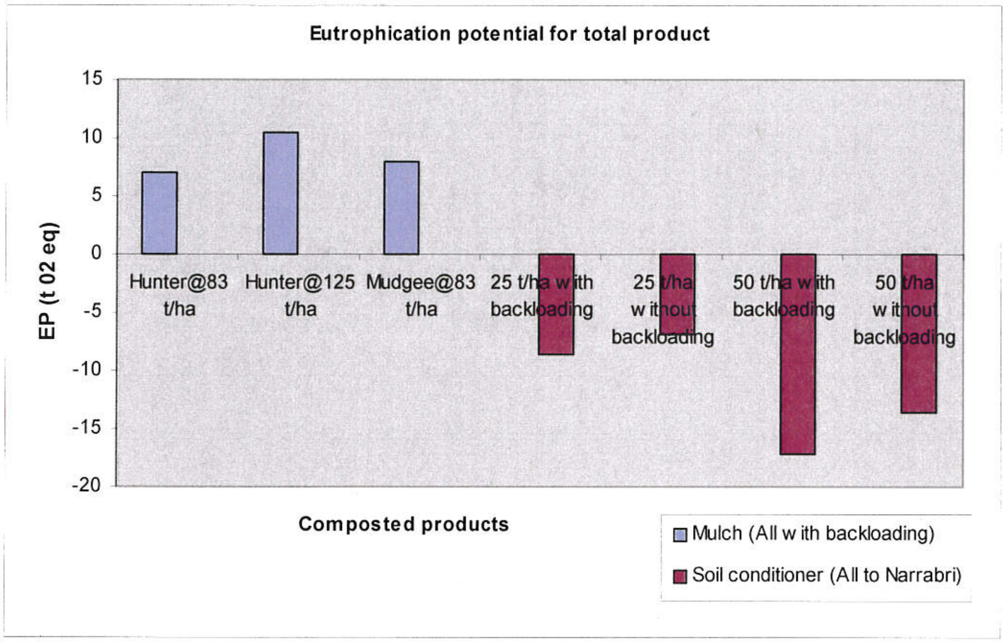
Production and land application of composted mulches and composted soil conditioners produced a net beneficial environmental impact with regard to global warming potential (GWP).

Application rates of composted products had far greater influence on GWP than variation in transport distance, and whether or not backloading was practiced.

8.3.1.2 Eutrophication potential

As defined in Section 3.3.6, eutrophication potential (EP) refers to the enrichment of ecosystems with nutrients via water and air, which may cause an undesirable shift in species composition and elevated biomass production in both aquatic and terrestrial ecosystems. Eutrophication of water resources results in oxygen depletion hence reduction in the diversity of species. The reference substance for eutrophication is kilograms (kg) of oxygen depleted.

Figure 8.4. Eutrophication potential for production and application of total composted product on 100 ha of agricultural land (Note: Negative numbers represent a net beneficial environmental impact and more negative results represent a greater environmental benefit. Positive numbers represent detrimental environmental impact and more positive results represent more detrimental environmental impact).



Nutrients, particularly N and P, are released into air and water as their oxides or ammonia gas (of nitrogen) during production and consumption of diesel fuel and electricity. Post-application benefits of composted products reduce the use of electricity, fertilisers and herbicides in agriculture resulting in reduced production of these items, hence reduced release of nutrients to air and water. The results show that all mulch scenarios (1 to 3) produced detrimental environmental impact (Figure 8.4). However, all soil conditioner scenarios (4 to 7) generated a net environmental benefit (Figure 8.4). Mulches as specified in this study are relatively low nutrient products compared to composted soil conditioners, and therefore reduce requirements for fertilisers to lesser extent than soil conditioners per tonne of application. Therefore the nutrients released into the environment associated with diesel fuel and electricity consumption during production of mulches are not completely offset by avoided releases arising from reduced fertiliser use and the releases arising from the production of higher quantities of fertilisers.

Higher application rates of composted products, particularly higher nutrient soil conditioners have a greater net benefit in relation to EP (Figure 8.4). It should be noted that this trend would only apply to the point where crop nutrient requirements are fully provided by the compost product; supply of nutrients in excess of those required by crops, whether from composts or fertilisers may result in greater negative EP environmental impact. The reduction of fertiliser use and the associated negative environmental impacts related to fertiliser production and distribution is significant in a range of environmental impact categories. Maximising the beneficial environmental impacts associated with the entire composting system therefore requires (amongst other things) maximising the nutrient value to the crop from compost application, across the life of the application. Applying composts with reference soil testing and nutrient budgeting for the particular crop/site is therefore important to maximising net environmental impact from the composting system.

Varying the transport distance (by 100 km) and/or backloading or without backloading situations have little impact on EP (Figure 8.4).

It is noteworthy that eutrophication potential from diesel production will be mainly confined to coastal water resources and not to inland water resources (river systems) because refineries that produce diesel are generally located in coastal areas associated with significant port cities. Eutrophication arising from diesel production in such contexts is likely be negligible compared to eutrophication caused by the urban environment. It may be argued that such EP emissions from urban and metropolitan settlements are more readily reduced by reducing personal and discretionary use of vehicle fuels and electricity rather than at the expense of sustainable soil management.

Conclusions for eutrophication potential impact category:

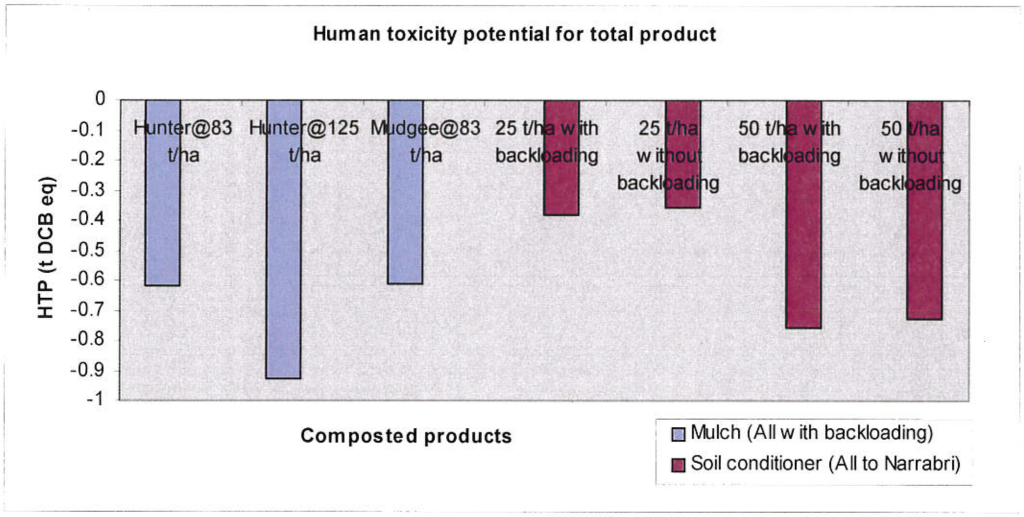
Production and land application of composted soil conditioners produced a net beneficial environmental impact but composted mulches produced a net detrimental environmental impact with regard to eutrophication potential (EP).

Application rates of composted products had far greater influence on EP than variation in transport distance, and whether or not backloading was practiced.

8.3.1.3 Human toxicity potential

Toxic substances released during production and use of diesel, electricity, fertilisers, pesticides, biocides and compost may be toxic to humans (Section 3.3.4). Toxic substances (such as oxides of nitrogen and sulphur, ammonia, ammonium, hydrochloric acid, and heavy metals) may be released to air, water and soil presenting human health problems.

Figure 8.5. Human toxicity potential for production and application of total composted product on 100 ha of agricultural land (Note: Negative numbers represent a net beneficial environmental impact and more negative results represent a greater environmental benefit).



The results of modelling for HTP are presented in Figure 8.5, and all scenarios (1 to 7) result reduced human toxicity potential, hence produce a net environmental benefit. The reduction in use of fertilisers, herbicides, water and electricity as a result of post-application benefits of applying composted products to agriculture substantially reduces the release of toxic substances during their production, which negates the impacts of toxic substances released during production and use of diesel and electricity required for production and application of composted products. Higher application rates of composted products produced a greater net environmental benefit. Increasing transport distance (by 100 km) and/or backloading or without backloading scenarios had very little impact on HTP.

Conclusions for human toxicity potential impact category:

Production and land application of composted soil conditioners and composted mulches produced a net beneficial environmental impact with regard to human toxicity potential (HTP).

Application rates of composted products had greater influence on HTP than variation in transport distance, and whether or not backloading was practiced.

8.3.1.4 Ecotoxicity potential

Toxic substances released during production and use of diesel, electricity, fertilisers, herbicides and compost may not only be toxic to humans, but also have ecological impacts on aquatic (marine as well freshwater), and terrestrial ecosystems and can result in decreased biodiversity (Section 3.3.4).

Figure 8.6. Ecotoxicity potential for production and application of total composted product on 100 ha of agricultural land. 8.6a-Terrestrial; 8.6b-Freshwater; 8.6c-Marinewater

Figure 8.6a Ecotoxicity potential for terrestrial ecosystems (Note: Negative numbers represent a net beneficial environmental impact and more negative results represent a greater environmental benefit).

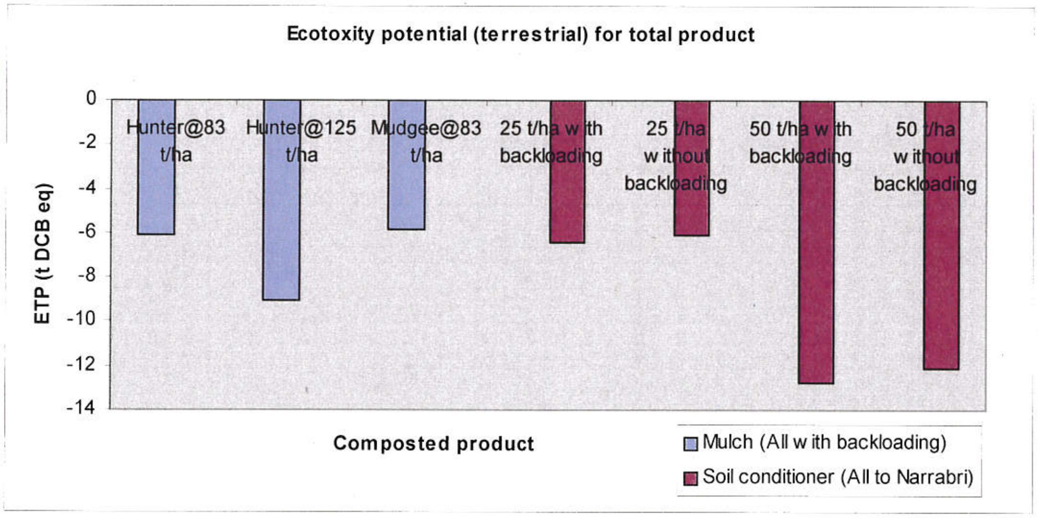


Figure 8.6b Ecotoxicity potential for freshwater ecosystems (Note: Negative numbers represent a net beneficial environmental impact and more negative results represent a greater environmental benefit).

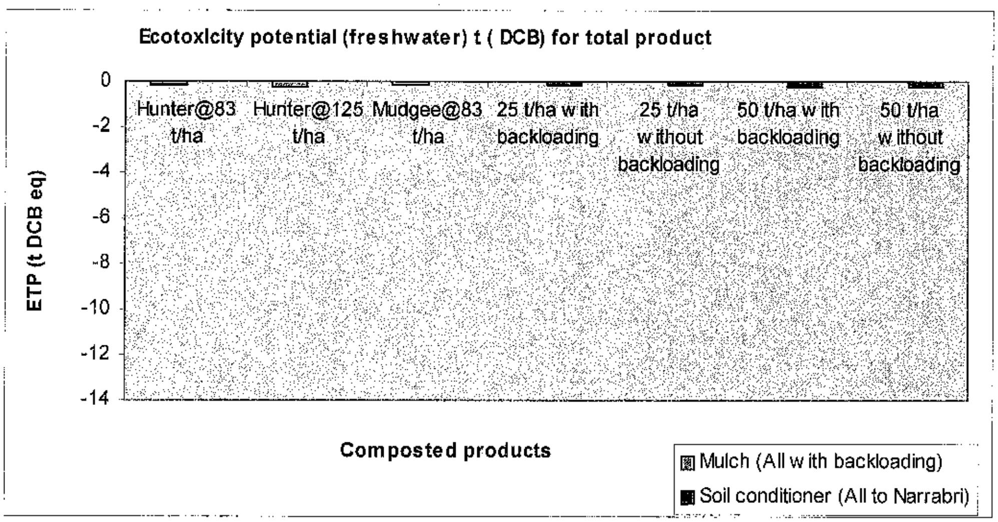
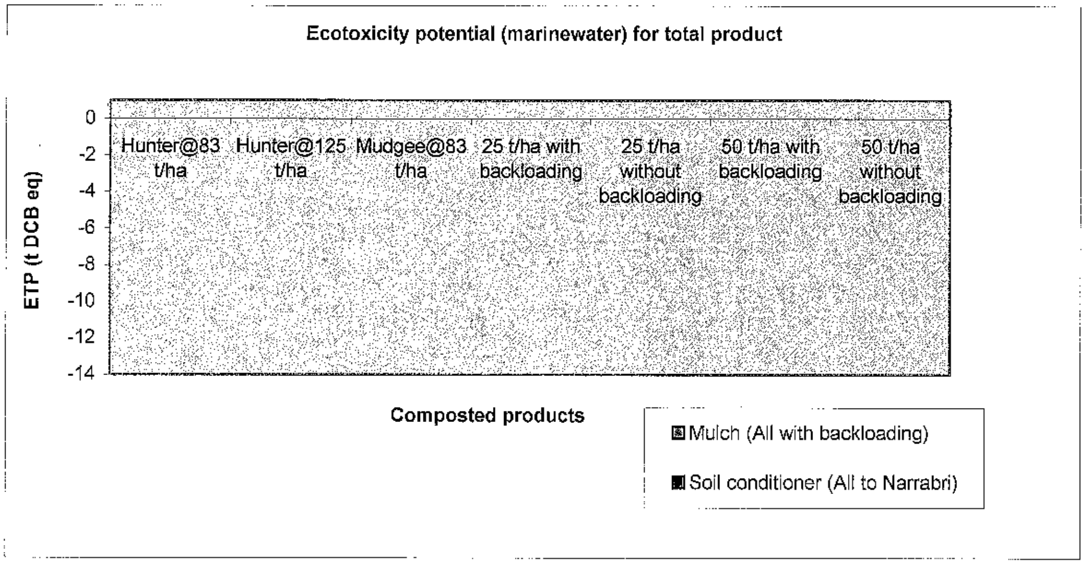


Figure 8.6c Ecotoxicity potential for marinewater ecosystems (Note: Positive numbers represent detrimental environmental impact and more positive results represent more detrimental environmental impact).



The results of modelling for ecotoxicity potential (ETP) show that production and application of composted products reduce ETP for terrestrial and freshwater ecosystems i.e. produce a net environmental benefit for all scenarios (1 to 7) (Figures 8.6a and 8.6b). The reduction in use of fertilisers, herbicides, water, and electricity as a result of post-application benefits to agriculture avoids substantial releases of toxic substances in terrestrial and freshwater ecosystems during production of these displaced farm inputs compared to toxic substances released during production and use of diesel and electricity required for production and application of composted products. Soil conditioners produced higher net environmental benefit compared to mulches, due to greater reduction in fertiliser use. Higher application rates of composted products produced a greater net environmental benefit. Increasing transport distance (by 100 km) and backloading did not produce substantial difference in ETP.

Whilst ETP for terrestrial and freshwater environments reduced, producing a net environmental benefit (Figures 8.6a and 8.6b), ETP for marine water environment increased by 0.001 t DCB eq. i.e. produced a negligible net detrimental environmental impact compared to beneficial environmental impact of ETP for terrestrial and freshwater environments, which cannot even be seen in Figure 8.6c when presented in the same scale as the benefits identified in Figures 8.6a and 8.6b. It is most likely that the production of diesel, electricity, fertilisers, herbicides, etc. occurs near coastal areas releasing toxic chemicals in coastal water resources, thereby causing a negligible net detrimental environmental impact. Mulches resulted in comparatively greater detrimental environmental impact compared to soil conditioners (net impact is so minute even can not be seen in Figure 8.6c). As discussed in Section 8.3.1.2, mulches specified in this study are relatively low nutrient products compared to composted soil conditioners, and therefore reduce requirements for fertilisers to a lesser extent than soil conditioners per tonne of application. Therefore the nutrients released into the environment associated with diesel fuel and electricity consumption during production of mulches is not completely offset by avoided releases arising from reduced fertiliser use and the releases arising from the production of higher quantities of fertilisers.

Conclusions for ecotoxicity potential impact category:

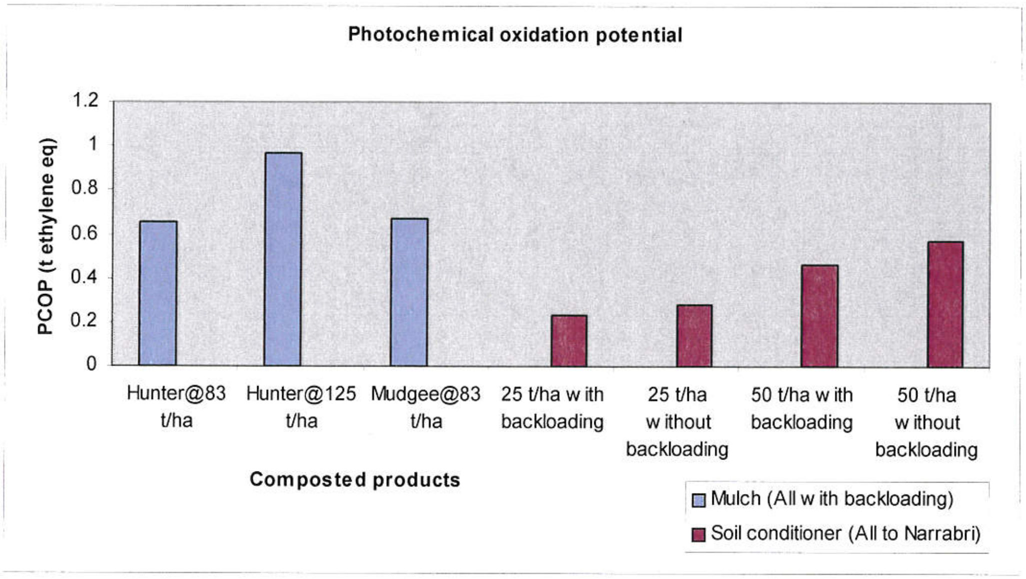
Production and land application of composted soil conditioners and composted mulches produced a net beneficial environmental impact with regard to ecotoxicity potential (ETP) for terrestrial and freshwater environment, however produced a negligible net detrimental environmental impact for marinewater environment.

Application rates of composted products had greater influence on HTP than variation in transport distance, and whether or not backloading was practiced.

8.3.1.5 Photochemical oxidation potential

As discussed in Section 3.3.3, photochemical oxidants are trace species that are formed during the photo-oxidation of volatile organic compounds (VOCs), carbon monoxide (CO) and oxides of nitrogen released during various human activities including compost production and application. For example ozone is one of the trace chemicals photochemically produced through oxidation of CO, VOCs and oxides of nitrogen.

Figure 8.7. Photochemical oxidation potential for production and application of total composted product on 100 ha of agricultural land (Note: Positive numbers represent detrimental environmental impact and more positive results represent more detrimental environmental impact).



The results of modelling for photochemical oxidation potential (POCP) presented in Figure 8.7 identify that all scenarios (1 to 7) show small positive POCP, i.e. production and application of composted products increase the potential for photochemical oxidation hence produce a small net detrimental environmental impact. This is due to the oxides of nitrogen, sulphur, and carbon (that produce photochemical oxidants) released during production and consumption of diesel fuel and electricity are substantially higher than the reduction in release of these oxides during production and use of fertilisers, herbicides, and electricity resulting from beneficial impacts of application of composted products in agriculture. Higher application rates of composted products again result in greater negative POCP environmental impact.

Variation in transport distance (by 100 km) or having backloading or without backloading scenarios did not make much difference to POCP.

Conclusions for photochemical oxidation potential impact category:

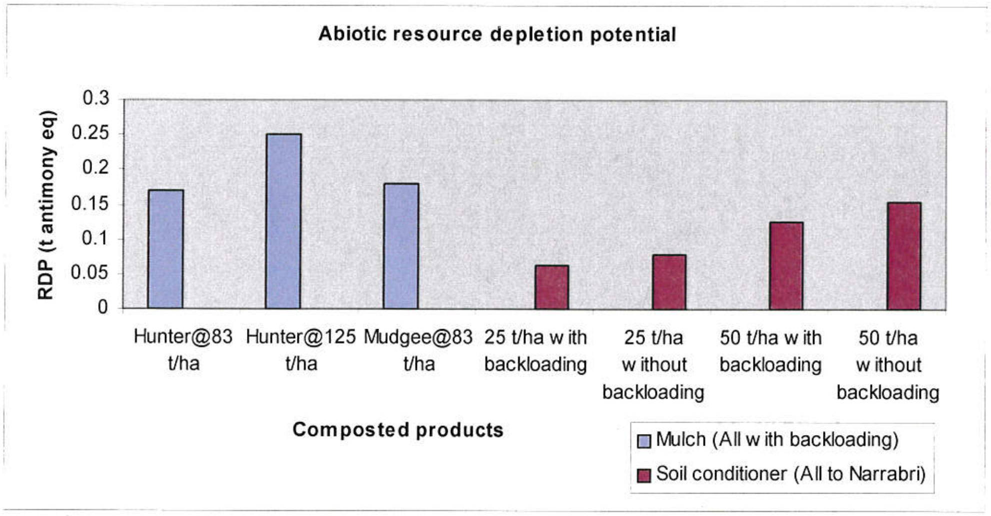
Production and land application of composted soil conditioners and composted mulches produced a small net detrimental environmental impact with regard to photochemical oxidation potential (POCP).

Application rates of composted products had greater influence on POCP than variation in transport distance, and whether or not backloading was practiced.

8.3.1.6 Abiotic resource depletion potential

Natural resources are being depleted at increasing rates as a result of increasing population and industrial development. As discussed in Section 3.3.5, most environmental crises facing humanity at present are directly related to resource use patterns and depletion, for example depletion of oil, coal, reserves of limited minerals, etc.

Figure 8.8. Abiotic resource depletion potential for production and application of total composted products on 100 ha of agricultural land (Note: Positive numbers represent detrimental environmental impact and more positive results represent more detrimental environmental impact).



Modelling results for abiotic resource depletion potential (ARDP) are presented in Figure 8.8. Results identify a small net detrimental impact in relation to ARDP in all modelling scenarios (1 to 7), indicating that production and application of composted products causes resource depletion. Please note the ARDP presented here is mainly related to depletion of natural resources such as coal, natural gas, and crude oil required for the production of diesel and electricity used for the production and application of composted products.

This analysis does not include reduction in use of resources such as fertilisers, water, and gypsum that result from the application of composted products in agriculture, nor does it include reduction in land degradation, which would consider healthy, productive soil as a resource. The reason these impacts are excluded is that characterisation/equivalency factors required for the impact category resource depletion for fertilisers and water were not available.

Composts have the potential to reduce sodicity of the soil and can avoid the use of gypsum. Relevant LCI data for gypsum production in Australia was not available and there is no data upon which to quantify the relationship between compost application and reduced gypsum use, so this could not be incorporated into LCIA impact categories such as ARDP.

The LCIA calculator for compost systems contains options for including calculations to convert fertilisers, water and gypsum LCI data to this impact category when and if relevant data becomes available.

The results of ARDP will remain positive, i.e. reduction in ARDP will not occur until such post-application benefits are quantified and included in these calculations.

Higher application rates of composted products have greater impact on ARDP than variation in transport distance (by 100 km) for both backloading or without backloading scenarios.

Conclusions for abiotic resource depletion impact category:

Production and land application of composted soil conditioners and composted mulches produced a small net detrimental environmental impact with regard to ARDP. Please note many resource depletion categories could not be included in the calculations due to unavailability of data. Once such data is included, the LCIA results for ARDP should produce beneficial environmental impact.

Application rates of composted products had greater influence on ARDP than variation in transport distance, and whether or not backloading was practiced.

8.3.1.7 Land use

As discussed in Section 3.3.7, land use impacts in LCIA are related to the area of land use, generally in combination with the time required to produce a certain output. Application of composted products, as discussed in Section 7, have the potential to increase soil organic carbon, improve soil structure, reduce sodicity, reduce soil erosion hence improve land productivity/quality, and to increase crop productivity. Although LCI data for these soil qualities is available, relevant characterisation/equivalency factors for land use are not available for LCIA. Information currently available on LCIA for land use is related to preserving the natural environment (i.e. wilderness), which is not applicable to this study. In future, if relevant information becomes available, land use impacts with application of composted products should be assessed.

Conclusions for Land use impact category:

Relevant data was not available to calculate impact on land use.

8.3.2 Avoided loads

Environmental life cycle analysis aggregates life cycle inventory data within the specified system boundary. The aggregated flows are then characterized in impact category potential indicators. Life cycle inventory data of many post-application environmental performance benefits could not be assessed because relevant relationships and estimates of data could not be established from available data. Therefore, some post-application performance benefits have been presented as un-characterized avoided loads and not as category impact potentials.

Avoided loads are related to post-application performance benefits, which are mainly affected by compost product type and application rates. Therefore results of avoided loads have been presented for varying application rates of composted mulches and soil conditioners. Changing transport distances, production inputs, and backloading or non-backloading scenarios have no relevance in relation to such avoided loads.

8.3.2.1 How to read results

Please note, results are read and interpreted in exactly the opposite manner to those presented in the previous section for environmental impact categories.

When reading results of avoided loads, positive numbers represent a net environmental benefit, or beneficial environmental impact. Therefore greater, or more positive results represent a greater environmental benefit.

Negative numbers represent no environmental benefit, or detrimental environmental impact. Therefore more negative results represent a greater negative environmental impact.

The results refer to total avoided loads arising from total product application to 100 ha of agricultural land.

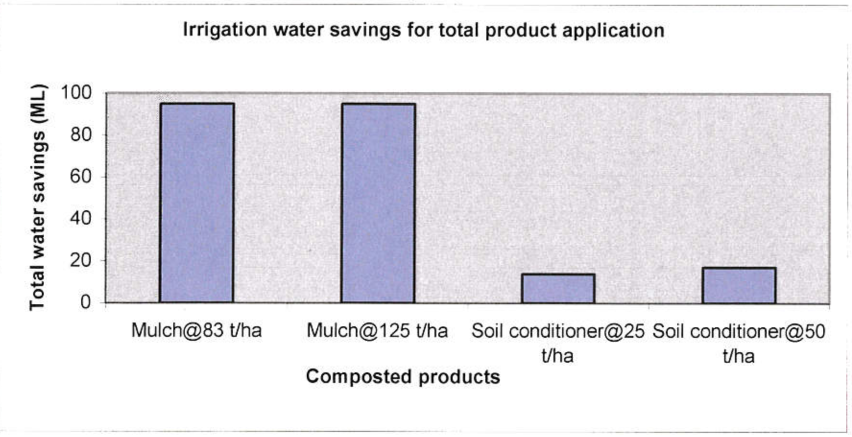
8.3.2.2 Avoided irrigation water potential

As discussed in Section 7.2, application of composted products has the potential to reduce irrigation water requirements in irrigated agriculture. In non-irrigated or dryland conditions, the application of composted products has the potential to conserve soil moisture hence reduce crop stress and potential crop failures due to lack of soil moisture.

It should be mentioned here (as discussed in Section 7.2) that irrigation water saving potential for soil conditioners has been derived from increases in plant available water, which provides an indication of potential irrigation water savings. Data on plant available water for mulches were not available therefore most relevant data, i.e. increase in soil moisture content was used to calculate irrigation water savings. Increase in soil moisture content does not translate directly to potential reduced irrigation water requirements, therefore irrigation water savings in the case of mulches may not provide an accurate indication of actual irrigation water savings (see Section 7.2).

Irrigation water savings in the case of composted soil conditioners results mainly from improved soil physical conditions such as soil aggregation and porosity. In the case of composted mulches, irrigation water savings initially arise from reduced water evaporation from the surface of the soil, and eventually through improvement in soil physical conditions over time as mulch decomposes and interacts with the soil.

Figure 8.9. Avoided irrigation water potential from application of total composted products on 100 ha of agricultural land.



The results of irrigation water savings potential are presented in Figure 8.9. Positive numbers identify that application of composted products reduce requirements of irrigation water producing a net environmental benefit in terms of water conservation and potential for increased environmental flows in river systems. Irrigation water savings are higher for mulches than for soil conditioners. As discussed earlier, irrigation water savings for mulches have been derived from increase in soil moisture content and this irrigation water savings potential may be realized only if the increased soil moisture content becomes available for plant use. The relationship between increase in soil moisture content and plant available moisture cannot be determined from existing data.

Higher application rates of soil conditioners generated more positive results, representing greater environmental benefit with respect to water savings. However, variation in mulch application rates (due to varying mulch width) did not change the calculation of potential irrigation water savings, this does not indicate that mulch application rate does not impact on soil moisture level. The reasons for no change in result is that mulch application rates in the modelling scenarios of this study varied due to mulch application width for same mulch application depth of 10 cm. Life cycle inventory data for irrigation water savings has been derived for mulch application depth only. That is why irrigation water savings for different mulch application rates were same. Amendments should be made to the LCIA calculator once such data relating to width of mulch application becomes available.

Savings in irrigation water have the potential to increase environmental flows in river systems and improve water quality hence maintain ecological integrity and biodiversity. This impact category remains unquantified.

Conclusions for avoided use of irrigation water:

Land application of composted soil conditioners and composted mulches produced a net environmental benefit in terms of savings in irrigation water.

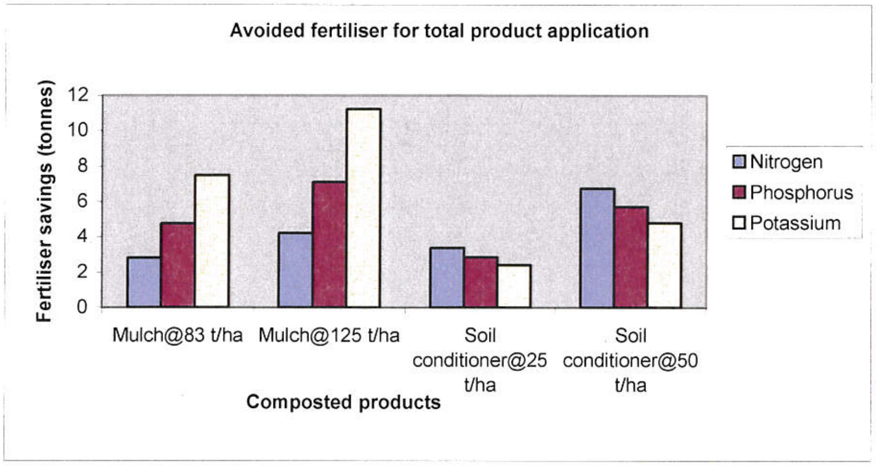
Composted mulches produced greater irrigation water savings than composted soil conditioners.

8.3.2.3 Avoided fertilizer potential

Composted products are a source of nutrients, the primary plant nutrients being N, P and K. Concentration of nutrients varies considerably in composted products. Generally mulches are lower nutrient products per unit quantity when compared to soil conditioners. For composts of the same nutrient value, higher application rates of the composted products will have greater fertilizer savings. This should not be read as a recommendation for high nutrient value composts, compost product and application rates must take into account the nutrient requirements of the relevant crop. Excess application of nutrients via fertilizers or compost is undesirable both for crop production and for resulting negative environmental impacts.

Positive results for fertilizer savings potential represent a net beneficial environmental impact (Figure 8.10) because avoided use of fertilisers reduces negative environmental impacts of producing and using fertilisers such as GWP, RDP, EP, HTP and ETP.

Figure 8.10. Avoided fertilizer potential in first year from application of total composted products on 100 ha of agricultural land.



Nutrients particularly N and P in composted products are present in organic form, which become available for plant use after mineralisation of organic matter over a 3 to 5 years period (see Section 7.3.7). Fertiliser savings shown in Figure 8.10 are for the first year of application and for the concentration of nutrients in the composted products specified in this report (see Section 2.3). Total fertiliser savings from this application will be much higher over the 3-5 year application cycle.

Conclusions for avoided fertiliser use:

Land application of composted soil conditioners and composted mulches produced a net environmental benefit in terms of reduction in fertiliser use.

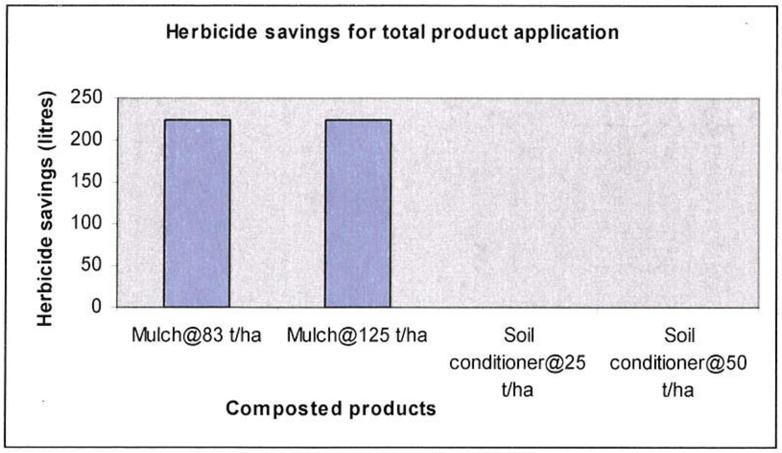
Composted soil conditioners produced greater benefit to reduce fertiliser use than composted mulches.

8.3.2.4 Avoided herbicides/pesticides potential

Application of composted mulches has demonstrated significant reduction in the growth of weeds resulting reduction in use of herbicide applications to destroy weeds (Section 7.4). Composted soil conditioners do not suppress weeds (Section 7.4).

The results of herbicide savings potential for composted products are presented in Figure 8.11. Positive numbers for mulches represent a net environmental benefit in terms of avoided use of herbicides, which reduce negative environmental impacts of producing and using herbicides such as GWP, RDP, EP, HTP and ETP. Depth of mulch application is considered most important in terms of weed suppression, although maximum application depth is inherently related to particle size, compost maturity and chemical properties of the mulch (e.g. electrical conductivity, nutrient content). Thicker mulches (of equivalent maturity and particle size) suppress a higher percentage of weeds for a longer duration. Life cycle inventory data used for modelling was based on 10 cm mulch depth, therefore different mulch application rates (varied due to mulch application width) (Figure 8.11) show the same herbicide savings potential as mulching depth in modelling scenarios is the same (10 cm).

Figure 8.11. Avoided herbicide use potential from application of total composted products on 100 ha of agricultural land (in this instance the relevant herbicide is glyphosate).



Conclusions for avoided use of herbicides/pesticides:

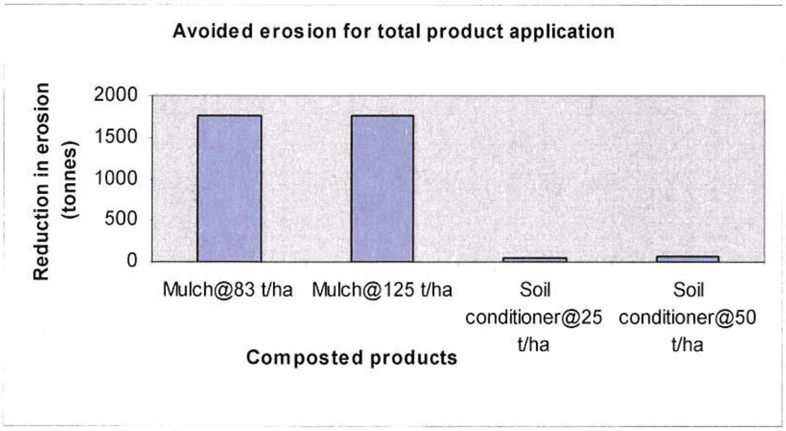
Land application of composted mulches produced a net environmental benefit in terms of reduction in herbicide use.

Composted soil conditioners do not suppress weeds therefore cannot reduce use of herbicides.

8.3.2.5 Avoided erosion potential

Application of composted products has the potential to reduce soil erosion. As mulches are used to cover the soil surface, they can significantly reduce soil erosion particularly on sloping steeper landforms (Section 7.8). Soil conditioners on the other hand (see Section 7.8) reduce soil erosion through improvement in soil structure. Both types of compost product increase the threshold of the soil.

Figure 8.12. Avoided erosion potential for application of total composted product on 100 ha of agricultural land.



Positive results for avoided soil erosion potential presented in Figure 8.12 show a net beneficial environmental impact. Mulch application produced more positive results, which represent greater (immediate) environmental benefit. Higher application rates of soil conditioners produced greater environmental benefit. However higher mulch application rate (varied due to mulch application width) provided the same benefit because LCI data for reduced erosion has been based on mulch application depth only.

Reduction in soil erosion has the potential to reduce runoff generation and accession of sediments and nutrients to waterways, and consequently to reduce eutrophication, and to maintain or improve ecological integrity and biodiversity relating to improved water quality.

Conclusions for avoided erosion:

Land application of composted mulches and composted soil conditioner produced a net environmental benefit in terms of reducing soil erosion.

Composted mulches produced this benefit greater than soil conditioner.

8.3.2.6 Carbon sequestration potential

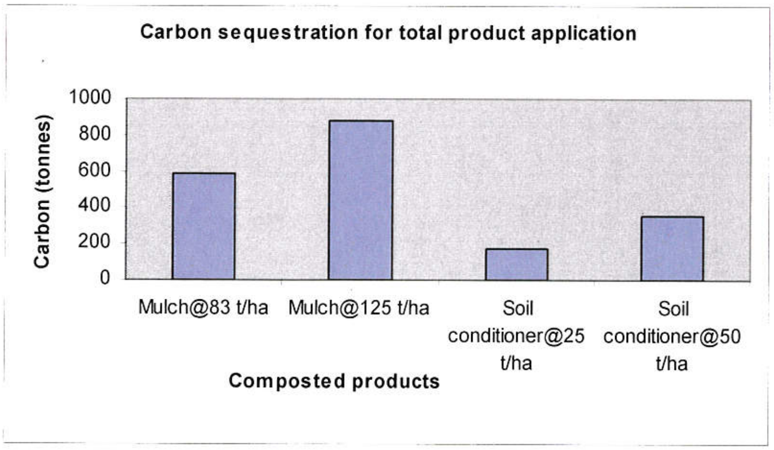
Composted products are rich source of organic carbon. When applied to the soil, this organic carbon decomposes and interacts with the mineral component of the soil, and a small proportion of it becomes part of the soil organic carbon.

Positive results of carbon sequestration represent a net beneficial environmental impact (Figure 8.13). Increased soil carbon also improves soil quality, particularly soil structure, which has the potential to maintain and/or improve land productivity and consequently productivity of crops.

Higher application rates of composted products produce more positive numbers hence greater environmental benefit because higher amounts of organic carbon being added via higher compost application rates result in greater soil carbon sequestration (Figure 8.13).

Carbon sequestration in case of soil conditioners will be realised quickly because the product is incorporated with the soil. In case of mulches, soil carbon sequestration will happen slowly over a longer time when organic carbon applied via mulches decomposes and interacts with the soil.

Figure 8.13. Carbon sequestration potential for application of total composted product on 100 ha of agricultural land.



Conclusions for carbon sequestration:

Land application of composted mulches and composted soil conditioner produced a net environmental benefit in terms of carbon sequestration in the soil.

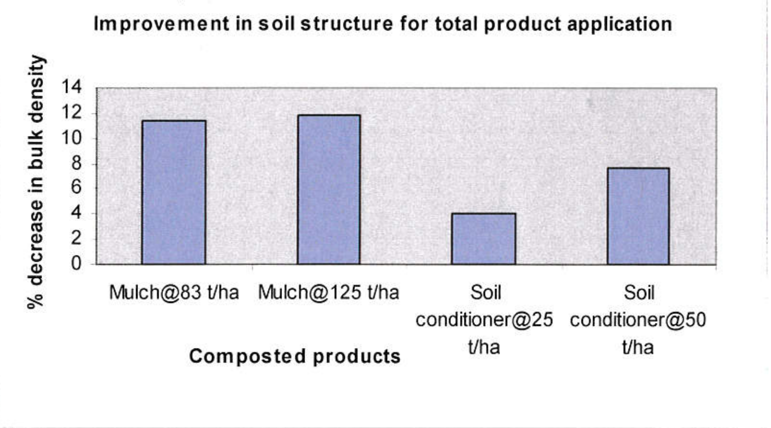
Composted mulches sequester more carbon in the soil than composted soil conditioner but soil conditioner produced this beneficial impact faster than mulches.

8.3.2.7 Soil structure improvement potential

As discussed in Section 8.3.2.6, application of composted products increase soil organic carbon, which improves soil aggregation and porosity, and hence reduces bulk density and improves soil structure facilitating tillage; seedling emergence; water retention; reduces runoff generation and soil erosion; and reduces eutrophication thereby maintaining ecological integrity and biodiversity. These beneficial outcomes also improve land productivity and consequently productivity of crops.

Positive results for soil structure improvement potential represent a net environmental benefit (Figure 8.14). Higher application rates of composted products generated greater environmental benefit. Soil conditioners will bring changes in soil structure more quickly than mulches for the same reason as discussed for soil carbon sequestration (Section 8.3.2.6).

Figure 8.14. Soil structure improvement potential for application of total composted product on 100 ha of agricultural land.



Conclusions for improvement in soil structure:

Land application of composted mulches and composted soil conditioner produced a net environmental benefit in terms of decreasing soil bulk density consequently improving soil structure.

Composted mulches decreased soil bulk density more than did soil conditioners but soil conditioner produced this beneficial impact faster than did mulches.

8.3.2.8 Remediation of saline-sodic soils

As discussed in Section 7.7, incorporation of soil conditioner has the potential to improve conditions of saline-sodic soils via soil structure improvement and leaching of excessive salts from the topsoil, this can result in reduced gypsum requirements (a common treatment for such soils) and avoid/displace environmental impacts arising from production and use of gypsum leading to net beneficial environmental impact. There are no data available to quantify how much gypsum can be replaced through the application of soil conditioners. Information on ability of mulches to reclaim saline-sodic soils is also not available. Life cycle inventory data for gypsum production relevant to Australian gypsum is also not available. Life cycle impact assessment for saline-sodic soils should be assessed when relevant information on gypsum production and potential replacement via compost applications becomes available.

Conclusions for remediation of saline-sodic soils:

Land application of composted soil conditioner has the potential to remediate saline-sodic soils through improvement in soil structure but such information is not known for composted mulches.

8.3.2.9 Increased plant productivity potential

Application of composted products has the potential to increase crop yields due to supply of required nutrients, improving soil conditions and increasing available moisture in the soil.

The results for plant productivity have been presented in Figure 8.15. Positive numbers for plant productivity represent a net benefit. Higher application rates produced greater benefit. In case of mulches, plant productivity was same for both application rates because mulch application rates for modeling scenarios in this study were varied due to mulch application width keeping application depth constant and equal to 10 cm. Increasing application rates beyond a certain range may reduce crop yields and produce negative environmental impacts as previously discussed. Local climate, soil types and crop varieties will also affect crop yields.

Figure 8.15. Increased plant productivity for application of total composted products on 100 ha of agricultural land.

Figure 8.15a Mulch application to grapevine.

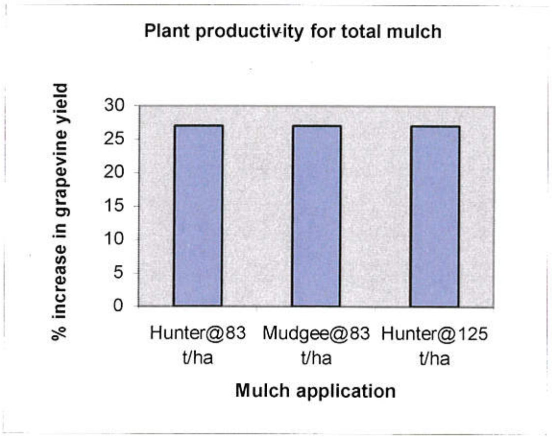
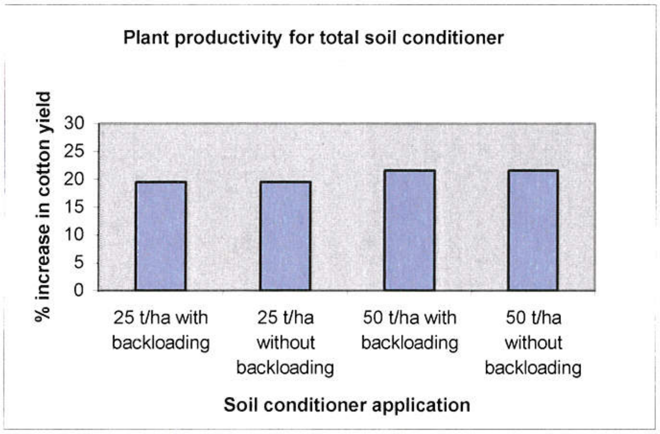


Figure 8.15b Soil conditioner application to cotton.



Conclusions for increased plant productivity:

Land application of composted mulches and composted soil conditioner produced a net environmental benefit in terms of increasing plant productivity.

Ecological integrity and biodiversity

The results of LCIA for all impact categories and avoided loads discussed above suggest that production and application of composted products in agricultural production systems has the potential to improve, or at the very least reduce the rate of decline in ecological integrity and biodiversity.

Ecological integrity and biodiversity is the degree to which the fundamental ecological processes (e.g. water and nutrient cycling, the flow of energy and biodiversity of species) are maintained (Environment Australia, 2001). An ecosystem includes the organisms of a particular area and their non-living environment, while biodiversity is the variability among living organisms from all sources (including terrestrial, marine and other ecosystems and ecological complexes of which they are part) and includes: diversity within species and between species and diversity of ecosystems (Environment Australia, 2001). Ecosystem integrity provides a continuous flow of current benefits and also maintains the capacity to respond to future needs and values (Environment Australia, 2001).

This study could not present the data for ecological integrity and biodiversity due to lack of data to quantify direct relationships between post-application benefits to ecological integrity and biodiversity. There is also no agreed methodology to quantify ecological integrity in the LCA paradigm. However the LCIA calculator for composting systems contains the capacity for including this impact category in future as quantification of such relationships become available.

For example, reduction in irrigation water requirements has the potential to increase environmental flows in river systems and consequently improve water quality, thereby improving the biodiversity of plant and animal species, however this study could not establish this relationship via either literature or expert advice from CSIRO researchers working in the specific area.

It has been established that reduction in soil erosion resulting from application of composted products has the potential to reduce transport of sediments and nutrients to water resources, reduce eutrophication potential and consequently to reduce algal productivity and production of algal detritus, the decomposition of which depletes oxygen from the system causing die-off of animals. Similarly, however, this study could not establish a direct relationship between these outcomes and ecological integrity and biodiversity.

8.3.3 Summary of LCIA results

The summary of LCIA results for production and application of composted products are given in Table 8.4. In the LCIA, impacts of product types, varying application rates of composted products, transport distances to application sites, return journey with or without backloading, and total application area are modelled and the results of modelling reported in the context of relevant environmental issues (impact categories).

Table 8.4 Summary of LCIA results for production and application of total composted products on 100 ha of land (▤-Beneficial impact; ×-Detrimental impact; **Note**- General comment).

Impact Categories	**Environmental impacts**	
Global warming potential (t CO_2 eq)	▤	Beneficial environmental impact for composted mulches and soil conditioners.
	Note	Application rates of composted products had greater influence on GWP than transport distance difference of 100 km and backloading or non-backloading.
Eutrophication potential (t O_2 eq)	▤	Beneficial environmental impact for soil conditioners.
	×	Detrimental environmental impact for mulches. Note this impact will be mainly confined to coastal water resources and not to inland river systems.
	Note	Application rates of composted products had greater influence on EP than varying transport distance by 100 km and backloading or non-backloading.
Human toxicity potential (t DCB eq)	▤	Beneficial environmental impact for composted mulches and soil conditioners.
	Note	Application rates of composted products had greater influence on HTP than transport distance difference of 100 km or backloading and non-backloading.
Ecotoxity potential (t DCB eq)	▤	Beneficial environmental impact for composted mulches and soil conditioners with respect to ETP for terrestrial and freshwater environment.
	×	Negligible detrimental environmental impact for composted mulches and soil conditioners with respect to ETP (marine water) compared to beneficial environmental impacts in regard to ETP for terrestrial and freshwater systems.
	Note	Application rates of composted products had greater influence on ETP than variation in transport distance by 100 km and backloading or non-backloading.
Photochemical oxidation potential (t ethylene eq)	×	Small detrimental environmental impact for composted mulches and soil conditioners.
	Note	Application rates of composted products had greater influence on HTP than transport distance difference of 100 km and backloading or non-backloading.
Abiotic Resource depletion potential (t antimony eq)	×	Small detrimental environmental impact for composted mulches and soil conditioners. Note a number of reductions in agricultural inputs and improvements in land conditions could not be included for this category in LCIA due to absence of data.
	Note	Application rates of composted products had greater influence on ARDP than transport distance difference of 100 km and backloading or non-backloading.
Land use	**Note**	Data not available

Table 8.4 Continued Summary of LCIA results for production and application of total composted products on 100 ha of land (▤-Beneficial impact; ×-Detrimental impact; **Note**- General comment).

Impact Categories	**Environmental impacts**	
Avoided loads		
Irrigation water (ML)	▤	Net environmental benefit from composted mulches and soil conditioners.
	Note	Mulches produced greater benefit than soil conditioners.
Fertilisers (tonnes)	▤	Net environmental benefit from composted mulches and soil conditioners.
	Note	Soil conditioners produced greater benefit than mulches.
Herbicides/pesticides savings (litres)	▤	Net environmental benefit from composted mulches.
	Note	Soil conditioners do not produce this benefit.
Reduced erosion (tonnes)	▤	Net environmental benefit from composted mulches and soil conditioners.
	Note	Mulches produced greater benefit than soil conditioners.
Carbon sequestration (tonnes)	▤	Net environmental benefit from composted mulches and soil conditioners.
	Note	Mulches produced greater benefit than soil conditioners.
	Note	Soil conditioners produce this benefit faster than mulches.
Soil structure improvement (% decrease in bulk density)	▤	Net environmental benefit from composted mulches and soil conditioners.
	Note	Mulches produced greater benefit than soil conditioners.
	Note	Soil conditioners produce this benefit faster than mulches.
Increased plant productivity (% increase in yield)	▤	Net environmental benefit from composted mulches and soil conditioners.

8.4 Limitations and gaps of LCIA

The main limitation of this LCIA is that it does not present total environmental impact assessment as it could not include a number of reductions in agricultural inputs (such as fertilisers and water) due to absence of characterisation/equivalency factors required for LCIA, and absence of a range of LCI data which would be expected to result in improved environmental performance of the composting system for a number of impact categories. The gaps identified for LCIA are listed below:

- There was lack of LCI data relevant to Australian gypsum production.
- CHARACTERISATION/EQUIVALENCY FACTORS FOR ABIOTIC RESOURCE DEPLETION POTENTIAL FOR FERTILISERS AND WATER WERE NOT AVAILABLE.
- Land use life cycle inventory assessment applicable to this study was not available.
- Lack of relationships to estimate and quantify direct relationships between post-application benefits to ecological integrity and biodiversity.

Section 9 Conclusions and options for further work

9.1 Conclusions

Life cycle inventory (LCI) of the windrow composting system including production of composted products, transport, and application to agricultural production systems in Australia has been developed and documented. Life cycle impact assessment (LCIA) of relevant environmental issues (in the scenarios modelled) has demonstrated overall net environmental benefits from production and application of composted products.

Life cycle inventory of windrow composting systems, including post-application impacts has identified relevant environmental issues (impact categories), and quantified data for production and application of composted products. Life cycle inventory data suggests that most environmental issues arising from production and transportation to application sites and application on agricultural land are related to production and consumption of diesel fuel and electricity required during all stages of composting process. In contrast, LCI data relating to a post compost application relate to wider range of issues including: reduction in use of irrigation water, electricity, fertilisers, herbicides for agriculture; reduction in soil erosion; carbon sequestration in the soil; increased plant productivity; improvement in soil structure; and the potential to reclaim saline-sodic soils.

Life cycle impact assessment of environmental issues arising from production and application of composted products has shown a net environmental benefit for impact categories including global warming potential (GWP), human toxicity potential (HTP), ecotoxicity potential (ETP) (terrestrial and freshwater) and eutrophication potential (EP) (in case of soil conditioners only). The beneficial results are largely due to the reduction in use of fertilisers, herbicides, water, and electricity resulting from compost applications, and therefore reducing release of GHGs, nutrients and toxic chemicals to environment (air, water, and soil) during production and use of these avoided inputs. These beneficial impacts offset the GHGs, nutrients and toxic substances released into environment during production and use of diesel and electricity required for production and application of composted products.

Life cycle impact assessment however has shown either a negligible or small net detrimental environmental impact from the composting system for impact categories such as ecotoxicity potential (ETP) (marine water), photochemical oxidation potential (POCP) and abiotic resource depletion (ARDP). The oxides of nitrogen, sulphur, and carbon (that produce photochemical oxidants) released during production and consumption of diesel fuel and electricity used in compost production and transport are currently calculated to be greater than the reduction in release of these oxides during production and use of fertilisers, herbicides, and electricity resulting from post compost application to agriculture. This study could not however include a number of reductions in agricultural inputs (such as fertilisers and water) and improvements in soil conditions due to absence of characterisation/equivalency factors required for LCIA, and absence of a range of data which would be expected to result in improved environmental performance of the composting system in these detrimental impact categories.

Composted mulches in the scenarios modelled have been shown to cause detrimental environmental impact with respect to EP. Mulches, being lower nutrient products compared to soil conditioners, can reduce requirements of fertilisers to a lesser extent than soil conditioners consequently more nutrients are released into environment during the production of fertilisers in the case of mulches. Reduction in fertiliser and other farm input use is key to realising a net environmental benefit from compost application. This highlights the importance of consideration of nutrient budgeting in specifying suitable composts for agriculture. Where compost products are selected that provide maximum nutrient value to a crop (over the life of the application) this will result in maximum environmental benefit arising from reduced fertiliser input. This principle applies also to avoided irrigation, herbicide and biocide application.

The results of the LCIA with respect to ETP have shown beneficial environmental impacts for terrestrial and freshwater ecosystems but a negligible detrimental environmental impact for ETP (marine water) suggesting that ecotoxicity such as eutrophication will be mainly confined to coastal water resources and not to inland water resources (river systems). It is most likely that production of diesel fuel occurs near coastal areas releasing toxic chemicals in marine water, thereby causing a minor net detrimental environmental impact with respect to ETP (marine water).

Higher application rates of composted products have been shown to produce greater net environmental benefits as higher application rates reduce the need for other farm inputs. Higher application rates per unit area produce higher post-application benefit, consequently greater net environmental benefit. However, it should be mentioned that post-application benefits do not increase linearly with increasing application rates. At certain application rates, most post-application benefits will tend to maximise or excessive application rates may cause detrimental affects suggesting a limit for net environmental benefits. Therefore, to achieve maximum environmental benefit, composts should be applied at a rate so as to maximise weed suppression and plant available moisture, whilst being manufactured to suppress crop/region specific pest and diseases. Lastly compost should be selected to maximise contribution to crop nutrient requirements when applied at the appropriate application rate.

Varying the transport distance by 100 km produced negligible impact on all impact categories, in particular GWP. However, there will be a limit for transport distance beyond which environmental impact from transport outweighs benefits of compost applications. Note, however, that transport of soil conditioner to the most distant area of the state still resulted in a net environmental benefit including with regard to GWP.

Backloading or without backloading scenarios hardly have any impact on all impact categories particularly GWP. The reason being greenhouse gases released during backloading journeys have not been accounted in this system because these are not considered part of this system. However, greenhouse gases released during non-backloading journeys have been included as a part of this system.

9.2 Options for further work

The recommendations of this study are:

a) It is highly recommended that an economic assessment should be conducted to calculate dollar values associated with identified environmental benefits to inform and justify future policies and strategies for organics recovery, management, and agricultural compost use. The Policy and Economic Research Unit of CSIRO Land and Water have highly reputable expertise in conducting agricultural and environmental economic assessment and are recommended for this task.

b) That this report be distributed to ensure that LCI data and LCA results are available for future studies, and for related policy and strategy development across Australia.

c) That the results and identified gaps of this study should be integrated into other Department of Environment and Conservation NSW projects, including field research trials to address gaps identified in LCI data for post-application impacts of composted products (Section 7.12) and life cycle impact assessment (Section 8.4).

d) That future LCA studies should be commissioned in a manner such that LCI data is transparently documented and is made available in a form suitable for use in future LCA studies, as has been the case with this study.

e) As reduction in fertiliser, irrigation water and biocide use due to compost application has significant influence on the realisation of net beneficial environmental impact from the composting system, maximising the nutritional, irrigation efficiency and disease/weed suppression value of composts to a particular crop across the life of a compost application are significant in terms of maximising beneficial environmental impacts. It is therefore recommended that soil testing, nutrient budgeting and other related crop requirements be integrated into the development, specification and selection of compost products for crop specific applications, not simply for direct agronomic benefit, but also for indirect net environmental benefit.

f) That future compost production application trials should assess the performance of products that are specifically designed to best meet the crop and environment performance requirements of the particular crop and growing region.

g) That the DEC Alternative Waste Technology Selector and Handbook should be updated to incorporate data and information arising from this study.

Section 10 References

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Section 11 Glossary

All terms defined in this glossary are based upon definitions given in the Recycled Organics Industry Dictionary and Thesaurus, 2nd Edition (2002) unless otherwise noted.

Term	Definition
Agricultural organics	Any residual organic materials produced as by-products of agricultural and forestry operations, including: weeds (woody and non-woody); animals (processing residuals, stock mortalities, pests), and crop residuals (woody and non-woody), and manures.
Australian Standard	A Standard is a published document, which sets out specifications and procedures designed to ensure that a material, product, method or service is fit for its purpose and consistently performs the way it was intended to.
Backloading	Backloading in this study means trucks transporting composted products to application sites return loaded with other products, which may or may not be related to composting system.
Bacteria	Microscopic single celled organisms occurring everywhere in nature. They can be harmful (cause diseases in plants and animals) and beneficial (decompose organic matter in the soil or composting pile or fix nitrogen on legumes from atmosphere).
Biodiversity	Variability among living organisms from all sources (including terrestrial, marine and other ecosystems and ecological complexes of which they are part) and includes: diversity within species and between species and diversity of ecosystems
Biosolids	Organic solids or semi-solids produced by municipal sewage treatment processes. Solids become biosolids when they come out of an anaerobic digester or other treatment process and can be beneficially used. Until such solids are suitable for beneficial use they are defined as waste-water solids. The solids content in biosolids should be equal to or greater than 0.5% weight by volume (w/v). Biosolids are commonly co-composted with garden organics and/or residual wood and timber to produce a range of recycled organics products.
Bulk density	It is a measure of soil porosity, with low values meaning a highly porous soil and vice versa.
Carbon dioxide equivalent	An atmospheric carbon dioxide concentration that would have the same radiative forcing effect as all of the human-produced greenhouse gases combined
Carbon sequestration	Natural or man-made processes that remove carbon from the atmosphere and store it for extended periods or permanently. A store of sequestered carbon (e.g. forest or soil) is known as a carbon sink.
Carbon sink	A pool (reservoir) that absorbs or takes up released carbon from another part of the carbon cycle
Carbon to nitrogen ratio (C:N ratio)	The ratio of the weight of organic carbon (C) to that of total nitrogen (N) in an organic material. Material with a large ratio (20:1) will break down slowly, while a small ratio (8:1) will quickly breakdown. Composted material with high C:N ratio when incorporated into the soil may cause a temporary nitrogen deficiency.
Cation exchange capacity	The ability of a soil to hold cations, preventing them from being leached. Soils with high clay and organic matter and low sand have high cation exchange capacity.
Characterisation/equivalency factors	Characterisation/equivalency factors are derived from characterization/mechanism model specific for each impact category, which are applied to convert the assigned LCI results to the common unit of the category indicator (AS/NZS ISO 14042:2001)
Clay	Soil particles less than 0.002 mm in diameter.
Compost	An organic product that has undergone controlled aerobic and thermophilic biological transformation to achieve pasteurisation and a specified level of maturity. Compost is suitable for the use as soil conditioner or mulch and can improve soil structure, water retention, aeration, erosion control, and other soil properties.

Term	Definition
Compostable organics	Compostable organics is a generic term for all organic materials that are appropriate for collection and use as feedstocks for composting or in related biological treatment systems (e.g. anaerobic digestion). Compostable organics is defined by its material components: residual food organics; garden organics; wood and timber; biosolids, and agricultural organics.
Composted mulch	Any pasteurised product, which has undergone composting for a period of not less than 6 weeks (excluding polymers which do not degrade such as plastics, rubber and coatings) that is suitable for placing on soil surfaces. Composted mulch has at least 70% by mass of its particles with a maximum size of greater than 15 mm.
Composted products	An organic product that has undergone controlled aerobic and thermophilic biological transformation to achieve pasteurization and a specified level of maturity.
Composted soil conditioners	Any composted product, including vermicast, manure and mushroom substrate, that is suitable for adding to soils. This term also includes 'soil amendment', 'soil additive', 'soil improver' and similar terms, but excludes polymers which do not biodegrade, such as plastics, rubber and coatings. Soil conditioner has not more than 20% by mass of particles with a maximum size above 16 mm.
Composting	The process whereby organic materials are pasteurised and microbially transformed under aerobic and thermophilic conditions for a period not less than 6 weeks. By definition, it is a process that must be carried out under controlled conditions yielding mature products that do not contain any weed seeds or pathogens.
Cover crop	A crop that is grown to reduce soil erosion, conserve nutrients, and provide organic matter. A cover crop may also be grown to nurse or protect another tender species until it is established.
Cultivation	Working the soil with implements in order to prepare it for sowing of crops.
Decomposition	The breakdown of organic waste materials by micro-organisms.
Denitrification	An anaerobic biological process which converts nitrogen compounds to nitrogen gas or nitrous oxide.
Dry matter	The portion of a substance that is not comprised of water. The dry matter content (%) is equal to 100% minus the moisture content (%).
Ecological integrity and biodiversity	the degree to which the fundamental ecological processes (e.g. water and nutrient cycling, the flow of energy and biodiversity) are maintained
Ecological sustainable development	Using, conserving and enhancing the community's resources so that ecological processes, on which life depends, are maintained and the total quality of life, now and in the future, can be increased
Ecotoxicity	Eco-toxicity deals with effects of toxic substances on terrestrial and aquatic ecosystems. The potential effects on ecosystems depend on the actual emission, the exposure to these emissions and the fate of specific substances in terrestrial and aquatic ecosystems.
Electrical conductivity	A measure of a solution's ability to carry an electrical current; varies both with the number and type of ions contained in the solution. Usually measured in deci-Siemens per metre (dS m^{-1}).
Erosion	The wearing away of the soil by water or wind. This may be natural or induced by land management practices.
Eutrophication	Process by which waters become enriched with nutrients, primarily nitrogen and phosphorus which stimulate the growth of aquatic flora and/or fauna.
Evapotranspiration	Removal of moisture from soil by evaporation and by transpiration by plants growing in the soil.
Exchangeable cations	Cations that are held by electrical charge to negatively charged clay or humus surfaces in the soil and can be exchanged with other cations in the soil solution.

Term	Definition
Extensive agriculture	Refers to the market segment within the recycled organics sector which incorporates: Pasture Farming, Broadacre Farming, and Forestry Farming.
Fallow	A farming practice in which land is left without a crop or weed growth for extended periods to accumulate soil moisture.
Feedstock	Organic materials used for composting or related biological treatment systems. Different feedstocks have different nutrient concentrations, moisture, structure and contamination levels (physical, chemical and biological).
Fertiliser	A substance that is added to the soil to supply essential nutrients for plant growth. Fertilisers may be natural or manufactured.
Field capacity	The amount of water remaining in the soil after free draining due to gravitation. It is the maximum amount of water that a soil can hold under free drainage conditions.
Food organics	The food organics material definition is defined by its component materials, which include: fruit & vegetable material; meat & poultry; fats & oils; seafood (including shellfish, excluding oyster shells); recalcitrants (large bones >15mm diameter, oyster shell, coconut shells); dairy (solid and liquid); bread, pastries & flours (including rice & corn flours); food soiled paper products (hand towels, butter wrap); biodegradables (cutlery, bags, polymers). Such materials may be derived from domestic or commercial and industrial sources. The definition does not include grease trap waste. Food organics is one of the primary components of the compostable organics stream.
Garden organics	The garden organics material definition is defined by its component materials including: Putrescible garden organics (grass clippings); non-woody garden organics; woody garden organics; trees and limbs; stumps and rootballs. Such materials may be derived from domestic, commercial and industrial and commercial and demolition sources. Garden organics is one of the primary components of the compostable organics stream.
Global warming	A term used to describe the role of atmospheric trace gases - water vapour, carbon dioxide, methane, nitrous oxide, ozone, in keeping the earth's surface warmer than it would be otherwise
Global warming potential	Heat trapping potential of a gas.
Greenhouse effect	The addition to the natural greenhouse effect resulting from human activities such as the burning of fossil fuels and land clearing, which increase the atmospheric levels of greenhouse gases such as carbon dioxide, methane, nitrous oxide, ozone and CFCs (see also global warming)
Greenhouse gas emissions	Emission of those gases that, by affecting the radiation transfer through the atmosphere, contribute to the greenhouse effect (see enhanced greenhouse effect)
Heavy metals	A group of metallic elements that include lead, cadmium, zinc, copper, mercury, and nickel. Can be found in considerable concentrations in sewage sludge and several other waste materials. High concentrations in the soil can lead to toxic effects in plants and animals ingesting the plants and soil particles.
Herbicides/pesticides	Agents used to inhibit plant/animal growth or kill specific plant/animal types.
Human toxicity	The impact category human toxicity contains effects of toxic substances on humans. The potential effect on humans depends on the actual emissions, the fate of the specific substance emitted to the environment and the time of exposure.
Humic acids	The chemical or biological compounds composed of dark organic substances that are precipitated upon acidification of a basic extract from soil or compost.
Humus	The dark or black carbon-rich relatively stable residue resulting from the decomposition of organic matter.

Term	Definition
Immobilisation, nitrogen Or Nitrogen Drawdown Index or Nitrogen Drawdown	Conversion of nutrient compounds from an inorganic form, available to plants, into the organic tissue of microorganisms (or other plants). The nutrients are unavailable until the microorganisms die and the microbial tissues containing these nutrients decompose. Nitrogen immobilisation occurs when materials with a high C:N ratio are land applied. The microorganisms that use the carbon also assimilate the available nitrogen, rendering it unavailable to plants.
Infiltration	The downward movement of water into soil. It is largely governed by the structural condition of the soil, the nature of the soil surface including presence of vegetation, and the antecedent moisture content of the soil.
Intensive agriculture	Refers to the market segment within the recycled organics sector which incorporates: Nurseries – production, Nurseries – wholesale, Fruit & Orchard Growing, Market Gardening, Mushroom Farming, Turf Grass Growing, and Viticulture.
Land application	The spraying or spreading of solid, semi-solid or liquid organic products onto the land surface, or their injection below the land surface.
Land degradation	The decline in land quality caused by improper use of the land.
Macronutrient or major nutrient	An essential nutrient that is needed in relatively large amounts. For example nitrogen, phosphorus, potassium.
Micronutrients or trace elements	An essential nutrient that is needed in small amounts. Foe example zinc, copper, boron, molybdenum.
Manure	Refers to all faecal and urinary excretion of livestock and poultry that are appropriate for collection and use as feedstock materials for composting or in related biological treatment systems. This material may also contain bedding, spilled feed, water or soil. See also agricultural organics.Such material may be derived from agricultural sources. These materials form one of the material description subcategories within the Agricultural Organics material description.
Maturation	Final stage of composting where temperatures remain steady below 45°C, and the compost becomes safe to use with plants due to the absence of toxins.
Maturity of compost	Is related to the level of composting feedstock material receives. A mature product is stable and does not cause toxicity to plants. See also Maturation and Stability.
Mineralisation	The breakdown of organic matter into its constituent inorganic components, carried out chiefly by decomposer microorganisms, and, for carbon, during respiration when carbon dioxide is returned to the environment.
Mulch	Any composted and pasteurised product (excluding polymers which do not degrade such as plastics, rubber and coatings) that is suitable for placing on soil surfaces. Mulch can be either composted mulches or pasteurised mulches. Mulch has at least 70% by mass of its particles with a maximum size of equal or greater than 16 mm.
Municipal solid waste	The solid component of the waste stream arising from all sources within a defined geographic area.
Nitrate nitrogen	A negatively charged ion comprised of nitrogen and oxygen (NO_3^-). Nitrate is a water soluble and mobile form of nitrogen. Because of its negative charge, it is not strongly held by soil particles (also negative) and can be leached away.
Nitrification	The biochemical oxidation of ammonia-nitrogen to nitrate.
Nitrogen (N)	Chemical symbol for nitrogen.
Nitrogen Drawdown Index	A measure of the ability of a composted organic product to supply soils and/or plants with soluble nitrogen. See also immobilisation, nitrogen.
Nutrient availability	The relative proportion of a nutrient in the soil (or compost) that can be absorbed and assimilated by growing plants.
Organic matter	Chemical substances of animal or vegetable origin, consisting of hydrocarbons and their derivates.

Term	Definition
Pasteurisation	An organic product that has undergone controlled aerobic and thermophilic biological transformation to achieve pasteurisation, but is relatively immature and lacking in stability compared to compost.
Pasteurised mulch	Any pasteurised organic product (excluding polymers which do not degrade such as plastics, rubber and coatings) that is suitable for placing on soil surfaces. Mulch has at least 70% by mass of its particles with a maximum size of greater than 15 mm.
Pasteurised products	An organic product that has undergone pasteurisation (a process whereby organic materials are treated to significantly reduce the numbers of plant and animal pathogens and plant propagules) but is relatively immature and lacking stability.
Pasteurised soil conditioner	Any pasteurised organic material that is suitable for adding to soils. This term also includes 'soil amendment', 'soil additive', 'soil improver' and similar terms, but excludes polymers which do not degrade such as plastics, rubber and coatings. Soil conditioner has not more than 15% by mass of particles with a maximum size above 15 mm.
Pathogens	Microorganisms capable of producing disease or infection in plants or animals. Pathogens can be killed by heat produced during thermophilic composting.
Permanent wilting point	The amount of water still held in the soil when plants wilt to the extent that they won't recover even if water is added. Soils differ in the amount of water still held at permanent wilting point.
pH	A measure of the concentration of hydrogen ions in a solution. pH is expressed as a negative exponent. Material that has a pH of 8 has ten times fewer hydrogen ions than a material with a pH of 7. The lower the pH, the more hydrogen ions are present, and the more acidic the material is. The higher the pH, the fewer hydrogen ions present, and the more basic it is. A pH of 7 is considered neutral.
Phosphorus (P)	Chemical symbol for phosphorus.
Photochemical oxidation	This category considers releases of organic compounds contributing to photochemical ozone formation (mostly hydrocarbons, ie. ethylene, propene, higher alkenes, ethane, propane, butane, benzene, toluene, xylene, formaldehyde and aromatic aldehydes, acetone and acetylene)
Plant available nutrient	A nutrient that can be easily taken up by plants from the soil solution.
Plant available water	Soil moisture that can be used by plants and is held at soil moisture potential between field capacity and permanent wilting point.
Plant productivity	It is the total quantity of harvested crop.
Plant propagule	Plant or part of a plant that could generate a new plant, e.g. a seed, part of a rhizome, corm, bulb, etc.
Potassium (K)	Chemical symbol for potassium
Processing	Subjecting a substance to a physical, chemical or biological treatment or a combination of treatments. Composting, for example, is a form of processing.
Recycled Organics	The term Recycled Organics has been adopted by NSW Waste Boards and EcoRecycle Victoria as a generic term for a range of products manufactured from compostable organic materials (garden organics, food organics, residual wood and timber, biosolids and agricultural organics).
Resource depletion	Resource depletion can be either abiotic (non-biological) or biotic (biological). Abiotic resources are natural resources such as phosphate ore, crude oil and wind energy which are non-living while biotic resources are living material resources e.g. rainforest.
Runoff	Rainfall that is not immediately absorbed into the soil. Water flows across the soil surface and depending on the soil condition the runoff may pick up soil particles and cause erosion.

Term	Definition
Salinity	The concentration of sodium chloride or dissolved salts in water, usually expressed in electrical
Saturated Extract	A solution derived by saturating a soil sample with water under standard conditions for a period long enough to dissolve any salts present.
Sodic soils/Sodicity	Soils with a high proportion of sodium relative to calcium, potassium and magnesium in the composition of the exchangeable cations on the clay fraction.
Soil aggregate	Groups of soil particles clumped together to form the structure of the soil.
Soil buffering capacity	The ability of a soil to withstand changes in pH. Soils with high levels of clay or organic matter have high buffering capacity.
Soil conditioners	Any composted or pasteurised organic product, including vermicast, manure and mushroom substrate that is suitable for adding to soils (excluding polymers which do not biodegrade, such as plastics, rubber and coatings. Soil conditioners any be either composted soil conditioners or pasteurised soil conditioners. Soil conditioners has not more than 20% by mass of particles with a maximum size above 16 mm and complies with appropriate criteria.
Soil degradation	Soil in which the structure has been damaged, compaction or erosion has occurred. It may also refer to soil acidity and salinity and the loss of nutrients from a soil.
Soil profile	Description of each of the layers (soil horizons) in the soil. Examination of the profile is made to determine depth, drainage, texture, structure and classification of the soil.
Soil structure	The combination or arrangement of primary soil particles into secondary particles, unit, or peds. Compost helps bind primary soil particles to improve the structure of soil.
Soil texture	A characterisation of soil type, based on the relative proportions of sand, silt, and clay in a particular soil.
Soil type	A general term used to describe the features of particular soils in terms of fertility, colour, texture and parent material.
Stability of compost	The rate of change or decomposition of compost. Usually stability refers to the lack of change or resistance to change. A stable compost continues to decompose at a very slow rate and has a low oxygen demand. See also maturation.
Subsoil	Soil in the lower horizons of the soil profile. The nature and depth of the subsoil is important for drainage and the growth of deep-growing plant roots.
Sustainability	In agriculture, sustainable practices are those which are, and will continue to be, profitable for farmers, that will conserve soil, vegetation and water resources and protect the environment, and that will assure adequate and safe food supplies into the future.
Temperate	Plant species that grow best in cool climates, or during the cooler months of the year; (as opposed to tropical plants, which grow in warmer climates).
Topography	The shape of the ground surface, such as hills, mountains, or plains. It also refers to the slope of the land.
Topsoil	The top part of the soil profile that contains the most fertile portion of the soil. It is usually darker than the subsoil, because it contains most organic matter. The topsoil is the most important part of the soil for agriculture, and must be protected by wise management. Original surface layer of soil from grassland, bushland or cultivated land.
Toxicity	The potential or ability of a material to cause adverse affects in an organism.
Viticulture	Viticulture is the term used to describe grape growing for production of various products mainly for wine, dry grapes, fresh consumption (table fruits), juice for non-alcoholic consumption and concentrated juice.
Water holding capacity	The amount of water held in a soil after any excess has drained away following saturation, expressed as a percentage of the oven-dry weight of the soil.

Term	Definition
Waterlogged	The condition of a soil, which is saturated with water and in which most or all of the soil air has been replaced. The condition, which is detrimental to most plant growth, may be caused by excessive rainfall, irrigation or seepage, and is exacerbated by inadequate site an/or internal drainage.
Weeds	A plant growing where it is not wanted.
Windrow composting	Windrow composting is a process for aerobic (aerated) and thermophilic (hot) biological transformation of organic materials. The process generates heat that destroys pathogens and produces stabilized compost products for use as a mulch, soil conditioner and topsoil additive.
Without backloading	Without backloading means trucks transporting composted products to application sites return without any load or empty.

Appendix 1: Detailed description of phases in an LCA

Stage of LCA	Component	Description	Section
Goal and scope definition	**Goal of study**	Intended application of the study and the reasons for carrying out the study and the intended audience are clearly identified	**1.1**
	Scope of study	**Components of the study** are described (see glossary for definitions): 1. The functions of the product system; 2. The functional unit – e.g. 1 tonne of source separated recycled organics/food organics/garden organics; 3. The product system to be studied (e.g. composting system); 4. The product system boundaries (e.g. collection to post application); 5. Allocation procedures (allocating data to unit processes); 6. Types of impact and methodology of impact assessment; 7. Data requirements; 8. Assumptions; 9. Limitations	**1.2**
Life cycle inventory analysis (LCI)	**Preparation prior to data collection**	**Flow diagrams** are required, outlining all unit processes to be modelled, including interrelationships	**2.1**
		Detailed descriptions of each unit process are made. All data categories associated with each unit process are identified	
		Must specify the units of measurement for each unit process	
		Description of data collection techniques and calculation techniques for each data category	
		Document any special cases, irregularities or other issues associated with the data provided from different sources	
	Collecting the data	The steps used for data collection may vary with each unit process. Procedures used and reasons are documented	**2.2**
		Avoid double counting or gaps by describing both qualitatively and quantitatively: the inputs and outputs of a system; where a process begins and ends, and the function of each unit process	
		Inputs and outputs need to be quantified using appropriate units of measurement (e.g. MJ). Where applicable, record the mass or volume of fuels used.	
		Reference the source of the collected data collected. For data that impacts significantly on the conclusions of the study, provide additional information (e.g. time when data collected, additional data quality indicators)	
	Calculation procedures	**Consideration should be made for the resources** (elementary flows- coal, oil etc) associated with the production of electricity	**2.3**
		Inputs and outputs related to combustible material e.g. oil, gas or coal, can be transformed into an energy equivalents by multiplying the material by its heat of combustion	
		Check data validity during the collection process	

Stage of LCA	Component	Description	Section
Life cycle inventory analysis (continued)	Calculation procedures (cont)	Address gaps in data or missing data – justification should be made when using 'non-zero' data values; or 'zero' data values to fill in the missing data. Calculated values based on the reported values from unit processes employing similar technologies can be used. Document all procedures used to fill in the missing data.	2.3
		For each unit process, an appropriate reference flow needs to be determined (e.g. 1 kg of material or 1 MJ of energy)	
		Data aggregation - data categories should only be aggregated if they are related to equivalent substances and to similar environmental impacts	
		Iterative refinement of data via sensitivity analysis – refinement of system boundaries. This process allows for the exclusion of life cycle stages or unit processes when data is shown to not be significant and vice versa for inputs and outputs that are shown to be significant. This will limit further data handling to data that is significant	
	Allocation of flows and releases	LCI analysis relies on linking unit processes within a product system by simple material or energy flows	2.5
		Document the allocation procedure (of inputs and outputs) used for each unit process	
	Limitations and conclusions	Identify all limitations associated with LCI analysis, including: 1) Data quality and assessment and sensitivity analysis; 2) The system functions and functional units 3) The system boundaries 4) Uncertainty analysis; 5) Limitations identified by the data quality and assessment and sensitivity analysis	2.6
		Conclusions and recommendations	
Life cycle Impact Assessment (LCIA)	**Background**	LCIA is used to examine a product system from an environmental perspective using impact categories and category indicators connected with the LCI results. The LCIA phase also provides information for the life cycle interpretation phase.	3.1
	SELECTION of impact categories (classification/categorisation)	An impact category is used to represent environmental issues of concern to which LCI results may be assigned (e.g. acidification)	3.2
		Category indicators are quantifiable representations of an impact category (e.g. global warming potential)	
		Characterization models reflect the environmental mechanism by describing the relationship between the LCI results, category indicators and in some cases category endpoints. The characterization model is used to derive the characterization factors used to calculate the category indicator results	
		For each impact category, a category indicator is selected and the category indicator result (indicator result) is calculated. The collection of indicator results (LCIA profile) provides information on the environmental issues (category endpoints) associated with the inputs and outputs of the product system.	

Stage of LCA	Component	Description	Section
Life cycle Impact Assessment (LCIA)	SELECTION of impact categories (classification/ categorisation)	Necessary components for each impact category include: 1. Identification of the category endpoints (areas affected by the impact e.g. forests, rivers etc.); 2. Definition of the category indicator for given category endpoints; 3. Identification of appropriate LCI results that can be assigned to the impact category, taking into account the chosen category indicator and identified category endpoint(s); 4. Identification of the characterization model and the characterization factors.	3.3
		Steps that need to be taken during the classification and characterisation process: 1. Select impact categories, category indicators and characterization models that are consistent with the goals and scope of study 2. Reference all sources for impact categories etc 3. Justify the selection of impact categories etc 4. Use accurate and descriptive names for the impact categories and category indicators 5. Describe the environmental mechanism and characterization model which relate the LCI results to the category indicator and provide the basis for characterization factors 6. Describe the appropriateness of the characterization model used for deriving the category indicator in the context of the goal and scope of the study.	
	Other issues to consider during classification/ categorisation	1. The impact categories, category indicators and characterization models should be internationally accepted 2. The impact categories should represent the aggregated emissions or resource use of the product system on the category endpoint(s) through the category indicators 3. Value choices and assumptions made during the selection of impact categories, category indicators and characterization models should be minimized. 4. The impact categories, category indicators and characterisation models should avoid double counting unless required by the goal and scope definition- e.g. when a study includes both human health and carcinogenicity 5. The characterization model for each category indicator should be scientifically and technically valid, and based upon a distinct identifiable environmental mechanism and/or reproducible empirical observation 6. The extent to which the characterization model and the characterization factors are scientifically and technically valid should be identified 7. The category indicators should be environmentally valid	3.4

Stage of LCA	Component	Description	Section
Life cycle Impact Assessment (continued)	**Category indicator details**	Identify the environmental relevance (at least qualitatively) of the category indicator in respect to the category endpoint(s) including: 1. The condition of the category endpoint(s); 2. The relative magnitude of the assessed change in the category endpoints; 3. The spatial aspects, such as duration, residence time, persistence, timing etc; 4. The reversibility of the environmental mechanism, and the uncertainty of the linkages between the characterization model and the changes in the category endpoints.	**3.5**
	Assignment of LCI results during classification	LCI results may be allocated exclusively to one impact category (e.g. carbon dioxide is allocated to Global Warming) or to more than one (e.g. SO_2 is allocated between the impact categories of Human Health and Acidification). If LCI results relate to more than one impact category, then allocations should be made on the basis of the goal and scope of the study (i.e. only allocate LCI results to impact categories that are relevant to the study)	**3.6**
	Calculation of category indicator results (characterization)	Calculations involve the conversion of LCI results to common units and the aggregation of the converted results within the impact category. This conversion uses characterization factors, giving a numerical indicator result. One example of a category indicator is infrared radiative forcing. A characterization factor, in this case global warming potential, is used to calculate the converted LCI results for each gas in units of carbon dioxide equivalents. Their contributions are then aggregated into an indicator result in terms of total carbon dioxide equivalents	**3.7**
	Optional elements (Valuation)	**Normalisation** of indicator results is done to better understand the relative magnitude for each indicator result. This procedure transforms an indicator result by dividing it by a selected reference value, e.g. total emissions or resource use for a given area (global, regional or local); total emissions or resource use for a given area on a per capita basis	**3.8**
		Grouping is assigning impact categories into one or more sets as predefined in the goal and scope definition. It is done by sorting by: 1) sorting impact categories on a nominal basis, e.g. by characteristics such as emissions and resources, or scales, global, regional etc.; 2) ranking the impact categories in a given hierarchy (high medium low etc). Ranking is based on value choices	
		Weighting is the process of converting indicator results of different impact categories by using numerical factors based on value choices. Weighting is achieved by: 1) converting indicator results or normalised results with selected weighting factors; 2) aggregating these converted indicator results or normalised results across impact categories	
	Data quality analysis	Additional techniques and information may be needed to understand the significance, uncertainty and sensitivity of the LCIA results, e.g. gravity analysis (pareto analysis); uncertainty analysis or sensitivity analysis	**3.9**

Stage of LCA	Component	Description	Section
	Limitations of LCIA	Identify all limitations inherent in the LCIA, e.g value choices for the selection of impact categories etc; variation in precision of category indicators; limitations of the LCI phase such as limitations in system boundary etc.	**3.10**
	Critical review of LCIA	It is necessary that a critical review of the report is performed by outside organisations with appropriate expertise in the area of study	**3.11**
Interpretation phase	**Objectives**	The objectives of life cycle interpretation are to analyse results, reach conclusions, explain limitations and provide recommendations based on the findings of the preceding phases (goal and scope, LCI and LCIA) and to report the results of the life cycle interpretation in a transparent manner	**4.1**
	Identification of significant issues	The significance of the results from the LCI and LCIA need to be determined. Significant issues can include: inventory data categories, such as energy, emissions, waste etc.; impact categories, such as resource use, global warming potential etc; essential contributions from results, such as individual unit processes or groups of processes like transportation and energy production	**4.2**
	Evaluation	Establish and enhance the confidence in and the reliability of the results of the study, including the significant issues identified above. During the evaluation, the use of the following should be considered: 1) completeness check 2) sensitivity check 3) consistency check	**4.3**
		Completeness check: ensures that all relevant information and data needed for the interpretation are available and complete.	
		Sensitivity check: assess the reliability of the final results and conclusions by determining whether they are affected by uncertainties in the data, allocation methods or calculation of category indicator results	
		Consistency check: Determine whether assumptions, methods and data are consistent with the goal and scope. Questions asked: Are differences in data quality along a product system life cycle and between different product systems consistent with the goal and scope of the study? Have regional and/or temporal differences been consistently applied? Have allocation rules and system boundaries been consistently applied to all product system? Have the elements of the impact assessment bee consistently applied?	
	Conclusions	Identify the significant issues Evaluate the methodology and results for completeness, sensitivity and consistency; Draw conclusions and check that these are consistent with the requirements of the goal a scope of the study, including, in particular, data quality requirements, predefined assumptions and values, and application oriented requirements. Make recommendations.	**4.4**

Appendix 2: Components of a composting system than can be quantified

	Life cycle stage[1]	LCI results	Impact category (IC)	Associated SOE indicator	Indicator result (assigned to IC)[2]	Time requirements	Location in the report
1.1	Collection	GHG emissions from fuel production and combustion	Global warming	GHG atmospheric concentrations – indicator 1.10 (SOE – atmosphere)	Kg of CO_2 equivalents	Excluded from study	Excluded
1.2		Motor vehicle exhaust emissions	Human toxicity/ Eco-toxicity	Not yet classified	Not yet classified	Excluded from study	Excluded
2.1	Receival	GHG emissions from fuel production and combustion	Global warming	GHG atmospheric concentrations – indicator 1.10 (SOE – atmosphere)	Kg of CO_2 equivalents	Consultation with composting facilities and use of existing LCI data	Not done
2.2		GHG emissions from electricity use	Global warming	GHG atmospheric concentrations – indicator 1.10 (SOE – atmosphere)	Kg of CO_2 equivalents	Use of existing LCI data for electricity use	Not done
2.3		Motor vehicle exhaust emissions	Human toxicity/ Eco-toxicity	Not yet classified	Not yet classified	Modification of research data	Not done
2.4		SO_2 emissions (electricity production)	Human toxicity/ Eco-toxicity	Concentration of SO_2 in urban areas – Indicator 3.2 (SOE – atmosphere)	SO_2 ppm	Use of existing LCI data/ modification of research data	Not done
2.5		Water use	Energy/Resource depletion	Not yet classified	Not yet classified	Consultation with composting facilities and the use of existing LCI data for water use	Not done
2.6		Leachate from stock piling of material (litres/tonne of feedstock)	Eutrophication	Nutrient loads – indicator 3.3 (SOE – water)	Total N and P (ppm)	Use of existing LCI/research data	Not done

SOE = State of the Environment Reporting system developed by Environment Australia

[1] See Appendix 2 – system model – refers to processing stage in composting cycle

[2] See Appendix 1, Table A1, Section 3

	Life cycle stage[1]	LCI results	Impact category (IC)	Associated SOE indicator	Indicator result (assigned to IC)[2]	Task	Location in the report
3.1	Processing	Fuel production and combustion	Global warming	GHG atmospheric concentrations – indicator 1.10 (SOE – atmosphere)	Kg of CO_2 equivalents	Use of existing LCI data (see Section 2.1)	Not done
3.2		Decomposition of feedstock	Global warming	GHG atmospheric concentrations – indicator 1.10 (SOE – atmosphere)	Kg of CO_2 equivalents	Use of existing research data and modification for use as LCI data	Not done
3.3		Leachate (litres/tonne of feedstock)	Eutrophication	Nutrient loads – indicator 3.3 (SOE – water)	Total N and P	Use of existing LCI/research data (see Section 2.6)	Not done
3.4		Water use	Energy/Resource depletion	Not yet classified	Not yet classified	Use of existing LCI data (see Section 2.5)	Not done
3.5		Emissions from electricity use	Global warming	GHG atmospheric concentrations – indicator 1.10 (SOE – atmosphere)	Kg of CO_2 equivalents	Use of existing LCI data (see Section 2.2)	Not done
3.6		SO_2 emissions (electricity production)	Human toxicity/eco-toxicity	Concentration of SO_2 in urban areas – Indicator 3.2 (SOE – atmosphere)	SO_2 ppm	Use of existing LCI data (see Section 2.4)	Not done
3.7		Motor vehicle exhaust emissions	Human toxicity/ Eco-toxicity	Not yet classified	Not yet classified	Modification of research data (see Section 2.3)	Not done
4.1	Transportation	GHG emissions from fuel production and combustion	Global warming	GHG atmospheric concentrations – indicator 1.10 (SOE – atmosphere)	Kg of CO_2 equivalents	Use of existing LCI data (see Section 2.1)	Not done
4.2		Motor vehicle exhaust emissions	Human toxicity/ Eco-toxicity	Not yet classified	Not yet classified	Adaptation of research data (see Section 2.3)	Not done

SOE = State of the Environment Reporting system developed by Environment Australia

[1] See Appendix 2 – system model – refers to processing stage in composting cycle

[2] See Appendix 1, Table A1, Section 3

	Life cycle stage[1]	LCI results	Impact category (IC)	Associated SOE indicator	Indicator result (assigned to IC)[2]	Task	Location in report
5.1	Land application	GHG emissions from fuel production and combustion	Global warming	GHG atmospheric concentrations – indicator 1.10 (SOE – atmosphere)	Kg of CO_2 equivalents	Use of existing LCI data – 10 days (consultation with Agriculture NSW)	Not done
5.2		Motor vehicle exhaust emissions	Human toxicity/ Eco-toxicity	Not yet classified	Not yet classified	Modification of research data (see Section 2.3)	Not done
5.3		Leachate from stockpiling of end-product	Eutrophication	Nutrient loads – indicator 3.3 (SOE – water)	Total N and P	Use of existing LCI/research data	Not done
6.1	Post application	Carbon sequestration	Global warming	GHG atmospheric concentrations – indicator 1.10 (SOE – atmosphere)	Kg of CO2 equivalents	Use of existing research data (consultation with Agriculture NSW)	Section 8.6
6.2		Leachate from application of end-product	Eutrophication	Nutrient loads – indicator 3.3 (SOE – water)	Total N and P	Modification of research data for use as LCI data	Excluded
6.3		Pesticide production (tonnes/ha of avoided product)	Human/ Eco-toxicity	Total immobile contaminant load on land area – Indicator 6.1 (SOE – land)	Toxicity factor/tonne of avoided product	Modification of research data for use as LCI data (consultation with NRA, TGA, NOHSC)	Section 8.5
6.4		Herbicide production (tonnes/ha of avoided product)	Human/ Eco-toxicity	Total immobile contaminant load on land area – Indicator 6.1 (SOE – land)	Toxicity factor/tonne of avoided product	Modification of research data for use as LCI data (consultation with NRA, TGA, NOHSC)	Section 8.4
6.5		Pesticide production GHG emissions (per tonne of avoided product)	Global warming	GHG atmospheric concentrations – indicator 1.10 (SOE – atmosphere)	Kg of CO_2 equivalents	Modification of existing research/LCI data	Section 8.4

SOE = State of the Environment Reporting system developed by Environment Australia; NRA = National Registration Authority; TGA = Therapeutic Goods Administration; NOHSC = National Occupational Health and Safety Commission

[1] See Appendix 2 – system model – refers to processing stage in composting cycle

[2] See Appendix 1, Table A1, Section 3

	Life cycle stage	LCI results	Impact category (IC)	Associated SOE indicator	Indicator result (assigned to IC)	Task	Location in report
6.6	Post Application	Herbicide production GHG emissions (per tonne of avoided product)	Global warming	GHG atmospheric concentrations – indicator 1.10 (SOE – atmosphere)	Kg of CO_2 equivalents	Modification of existing research/LCI data	Section 8.4
6.7		Fertilizer production (per tonne of avoided product)	Human/ Eco-toxicity	Not yet classified	Not yet classified	Modification of existing research/LCI data (consultation with NRA, TGA, NOHSC)	Section 8.3
6.8		Fertilizer production (per tonne of avoided product)	Global warming	GHG atmospheric concentrations – indicator 1.10 (SOE – atmosphere)	Kg of CO_2 equivalents	Modification of existing research/LCI data	Section 8.3
6.9		Gypsum production	Human toxicity/Eco-toxicity	Not yet classified	Not yet classified	Modification of research data (consultation with NRA, TGA, NOHSC)	Section 8.7
6.10		Gypsum production	Global warming	GHG atmospheric concentrations – indicator 1.10 (SOE – atmosphere)	Kg of CO_2 equivalents	Modification/use of existing research data	Section 8.7
6.11		Erosion (tonnes/ha)	Erosion	Surface soil loss index – Indicator 1.4 (SOE – land)	Tonnes/ha of soil loss	Modification of existing research data, consultation with technical advisors	Section 8.8

	Life cycle stage	LCI results	Impact category (IC)	Associated SOE indicator	Indicator result (assigned to IC)	Task	Location in report
6.12		Dryland salinity (area affected)	Salinity	% area of land effected by dryland salinity and acidity by catchment – indicator 3.3 (SOE – land)	% land potentially rehabilitated / tonne of compost?	Modification of existing research data, consultation with technical advisors	Section 8.7
6.13		Sodicity (area affected)	Sodicity	% area of land effected by dryland salinity and acidity by catchment	% land potentially rehabilitated / tonne of compost?	Modification of existing research data, consultation with technical advisors	Section 8.7
6.14		Water conservation	Resource depletion	Not yet classified	Not yet classified	Use of existing research data, consultation with technical advisors	Section 8.2

SOE = State of the Environment Reporting system developed by Environment Australia; NRA = National Registration Authority; TGA = Therapeutic Goods Administration; NOHSC = National Occupational Health and Safety Commission

Appendix 3: Glossary of LCA terms

All terms defined in this glossary are based upon definitions given in the AS/NZS ISO 14040 series of international standards.

Term	Definition
Category end-points	Environmental issue or area affected by an impact category – e.g. forests, oceans, soil, rivers, reservoirs etc.
Category indicator	Quantifiable representations of an impact category (e.g. global warming potential)
Characterisation	Characterization models reflect the environmental mechanism by describing the relationship between the LCI results, category indicators and in some cases category endpoints. The characterization model is used to derive the characterization factors used to calculate the category indicator results
Comparative assertion	Environmental claim regarding the superiority or equivalence of one product/system (e.g. composting) versus a competing product/system (e.g. waste-to-energy) which performs the same function (e.g. processing or compostable organics)
Composting system	In this LCA context, 'composting system' refers to the system model defined in Appendix 2, which includes processes associated with commercial windrow composting technology such as the collection and transport and receival of source separated compostable organic materials, and the manufacture, distribution and application of recycled organics products (i.e. compost).
Data quality	Characteristic of data that identifies its ability to satisfy stated requirements. Precision of data or compatibility of data with the goals and objectives of a study
Elementary flow	1. Material or energy entering the system being studied, which has been drawn from the environment without previous human transformation (e.g. coal, oil) 2. Material or energy leaving the system being studied, which is discarded into the environment without subsequent human transformation (e.g. waste materials)
Fugitive emission	Uncontrolled emission to air, water or land
Functional unit	The function or service that a system provides – for use as a reference point to make comparisons of environmental impacts. An appropriate functional unit for composting processes is the treatment of a specified amount of compostable organics (e.g. 1 tonne of compostable organics per year)
Impact category	Class representing environmental issues of concern to which LCI results may be assigned. Typical impact categories include: global warming, ozone, eutrophication, human toxicity, eco-toxicity and energy.
Input	Material or energy that enters a unit process (stage in a system) – may include coal, oil, electricity etc. required to operate equipment
Life cycle	Consecutive and interlinked stages of a product system, from raw material acquisition or generation of natural resources to the final disposal of the end-product
Life cycle assessment (LCA)	Compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle. LCA comprises three main phases, Life Cycle Inventory Analysis (LCI); Life Cycle Impact Assessment (LCIA), and Life Cycle Interpretation.

Term	Definition
Life cycle impact assessment (LCIA)	Phase of life cycle assessment aimed at understanding and evaluating the magnitude and significance of the potential environmental impacts of a product system
Life cycle inventory	Modified data used to determine the total environmental impact associated with processes in a system. Derived from a number of sources, including research data, consultation and experimentation. Mathematical transformations are usually required to adapt data for use in the context of an LCA and to adhere to the goals and scope of a study
Life cycle inventory analysis (LCI)	Phase of life cycle assessment involving the compilation and quantification of inputs and outputs (inventory data), for a given product or system throughout its life cycle
Life cycle interpretation	Phase of life cycle assessment involving the compilation and quantification of inputs and outputs (from LCI and LCIA phases), for a given product system throughout its life cycle
Output	Material or energy which leaves a unit process
Product system	Collection of materially and energetically connected unit processes which performs one or more defined functions (e.g. windrow composting system)
Raw material	Primary or secondary material that is used to produce a product (e.g. coal or oil used for electricity generation)
System boundary	Interface between a product system and the environment or other product systems
Sensitivity analysis	Systematic procedure for estimating the effects on the outcome of a study of the chosen methods and data – method for determining the significance or value of results in a life cycle assessment
Transparency	Open, comprehensive and understandable presentation of information
Uncertainty analysis	Systematic procedure to ascertain and quantify the uncertainty introduced into the results of a life cycle inventory analysis due to the cumulative effects of input uncertainty and data variability
Unit process	Smallest portion of a product system for which data are collected when performing a life cycle assessment (e.g. stockpiling compostable organics materials at a composting facility; size reducing compostable organics materials; etc)

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The impact of landfilling and composting on greenhouse gas emissions – A review

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ABSTRACT

Municipal solid waste is a significant contributor to greenhouse gas emissions through decomposition and life-cycle activities processes. The majority of these emissions are a result of landfilling, which remains the primary waste disposal strategy internationally. As a result, countries have been incorporating alternative forms of waste management strategies such as energy recovery from landfill gas capture, aerobic landfilling (aerox landfills), pre-composting of waste prior to landfilling, landfill capping and composting of the organic fraction of municipal solid waste. As the changing global climate has been one of the major environmental challenges facing the world today, there is an increasing need to understand the impact of waste management on greenhouse gas emissions. This review paper serves to provide an overview on the impact of landfilling (and its various alternatives) and composting on greenhouse gas emissions taking into account streamlined life cycle activities and the decomposition process. The review suggests greenhouse gas emissions from waste decomposition are considerably higher for landfills than composting. However, mixed results were found for greenhouse gas emissions for landfill and composting operational activities. Nonetheless, in general, net greenhouse gas emissions for landfills tend to be higher than that for composting facilities.

1. Introduction

Prior to the mid-1990s, waste management (WM) issues tended to revolve around public heath and safety issues (Marshall, 1972; Tchobanoglous and O'Leary, 1994), and the optimization of landfill gas (LFG) capture and utilization (Komilis et al., 1999a). The subject of its impact on greenhouse gases (GHG) has only been addressed recently, making it a relatively young study. This reflects an increasing acknowledgement of wastes contribution to GHG which is reflected in recent development in GHG mitigation policies internationally (SoE, 2007; Themelis and Ulloa, 2006; European Communities, 2001).

The waste sector is a significant contributor to greenhouse gas (GHG) emissions accountable for approximately 5% of the global greenhouse budget (IPCC, 2006). This 5% consist of methane (CH_4) emission from anaerobic decomposition of solid waste and carbon dioxide (CO_2) from wastewater decomposition (IPCC, 2006). Life cycle activities associated with these WM strategies are not included in this estimation. However, for a more holistic approach, streamline life cycle activities should also be accounted when quantifying a WM strategy impact on GHG emissions.

The most common WM strategy today is landfilling (Hogg et al., 2008; Pipatti and Wihersaari, 1998) and is expected to increase due to developing countries movement away from open dumping to landfilling (Meadows et al., 1997). However, as a result of increased environmental awareness, there has been an emergence of alternative and "add-on" features to landfilling such as energy recovery from LFG recovery, aerox landfills, pre-composting of waste prior to landfilling and compost capping; all of which have various impacts on GHG emissions. More importantly, there has been a movement to divert waste from landfills in order to reduce the negative environmental impact of landfills such as leachate contamination, GHG emissions and space limitation (Slater and Frederickson, 2001; Norbu et al., 2005). Composting has thus been widely acknowledged as an alternative to landfills.

In this review, the impact of landfilling and some of its more common mitigation applications, alongside with the impact of composting on GHG emissions are reviewed. This summary of information allows for the start and/or a further understanding of the consequences different WM strategies has on climate change; thus allowing decision makers to make more climate responsible decisions with regard to WM.

2. Landfill

The impact of landfills on climate change lies in their emissions of CO_2 and CH_4, along with several other gaseous components. These gases are the by-product of anaerobic decomposition of organic waste, characteristic of conventional landfills, and tend to accumulate within the landfill (USEPA, 1998). Only CH_4 is accounted for in the estimation of GHG emissions from landfills

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and not CO_2 despite its global warming potential (GWP) upon release (IPCC, 2006; DCC, 2007). This is due to the general consensus that CO_2 from waste decomposition is of biogenic origin and hence does not add to the overall GHG emissions that contribute to global warming (IPCC, 2006).

There are two life stages in a landfill, its operating stage, where municipal solid waste (MSW) is being disposed of, and its closed stage, where storage capacity is reached. Operating landfills emits more CH_4 than closed landfills due to the majority of degradation occurring in the first few years following disposal with decreasing emission rates with time after closure (Fourie and Morris, 2004; Humer and Lechner, 1999a). Following closure, a landfill continues to emit GHG, possibly for several hundreds of years (Borjesson et al., 2004). The general trend of GHG emissions from landfills can be seen in Fig. 1. It can be observed from Fig. 1 that even if landfills were to be removed right now, the issue of CH_4 generating in current landfills will still remain for many years. Hence, landfill managers today are faced with the challenge of dealing with the GHG emissions from current and old landfills.

Various independent theoretical and experimental studies suggest a large variation of GHG generation from 1 ton of waste, ranging from 40 m^3 to 250 m^3 (Humer and Lechner, 1999a; Ayalon et al., 2000; Bogner et al., 1997; Themelis and Ulloa, 2006). This is understandable as LFG generation is highly dependent on a variety of factors which are covered in Komilis et al. (1999a). One of the most crucial factors determining GHG emissions is waste composition. The organic fraction of waste stream is responsible for the emission of GHG and different types of organic waste have their own associated degradable organic carbon (DOC) and rate constants which affect the total amount emitted, and the rate at which is generated, respectively (DCC, 2008). The rate of emissions can be controlled with strategies such as waste compaction, leachate recirculation (Cabaraban et al., 2008) and even aerobic landfilling (Read and Hudgins, 2000). Despite the rate of emissions, the potential emissions from a conventional landfill from MSW, from Australian default values, is approximately 1.287 ton CO_{2-e}/ton of waste. This value can be lowered by adopting different mitigation strategies that converts CH_4 to CO_2 such as waste-to-energy, and precomposting of waste prior to landfilling.

CH_4 accounting aside, GHG emissions from landfilling activities are also an important contribution to the overall impact of landfilling on GHG emissions. Emissions are a result of transportation, excavation, compaction and soil spreading. Baky and Eriksson (2003) estimated a fuel consumption of 1.036×10^{-3} kL/ton waste landfilled for a conventional landfill while WARM, designed by the USEPA (2006) estimated 0.72 ton CO_{2-e}/ton waste landfilled and McDougall et al. (2001) provided a fuel consumption estimate of 0.6 L/m^3 of void space. Different landfill operations will result in different operational emissions; hence, the choice of landfill operations will affect the net emissions. This raises the question of whether certain landfill mitigation strategies, despite their ability to reduce GHG emissions by CH_4 conversion, will be able to achieve net lowered GHG emissions, when operational activities are considered too.

2.1. Landfill gas capture

The most common mitigation strategy is the capture of LFG for flaring or combustion to recover energy as this present significant environmental, economic and energy benefits (El-Fadel and Sbayti, 2000). As of 2001, there were approximately 955 landfills collecting LFG worldwide (Themelis and Ulloa, 2006) as compared with an estimate of 400 sites in 1995 (Meadows et al., 1997). This shows that countries are taking a step in the right direction towards mitigating GHG from landfills. Despite an increase in LFG capture in recent years, the number of landfill sites recovering energy in Australia is still relatively low, ranging from 4% to 19% (WMAA, 2008). However, with legislative requirements such as the National Greenhouse and Energy Reporting Act 2007 that specifies an emission threshold of 25 kilo-ton of carbon dioxide equivalent units (expressed as CO_{2-e}) for a landfill facility (Wong, 2008), LFG capture and utilization will become a more promising option to fulfil that requirement.

There are two conventional fates that entail captured LFG: the biogas is flared or combusted to generate electricity (aka waste-to-energy i.e. WtE). The primary objectives of these practices are to recovery energy and protect the people and the environment from the volatile nature of LFG, respectively. GHG reduction, through the conversion of CH_4 to CO_2, is a secondary effect as a result of these actions. Recovery efficiency from landfill depends largely on the operating conditions and the age of the waste, though independent studies have yielded varying results. Pipatti and Wihersaari (1998) stated efficiencies to vary between 50% and 100% while Oonk and Boom (1995) reported 24% to 60% and Humer and Lechner (1999b) reported 40% to 60% efficiency. Laboratory experiments by Bogner and Spokas (1995) yielded values ranging 25–50%. This range of recovery efficiency is largely dependent on the waste composition and moisture content. Waste streams with a low organic fraction, e.g. food waste, will result in

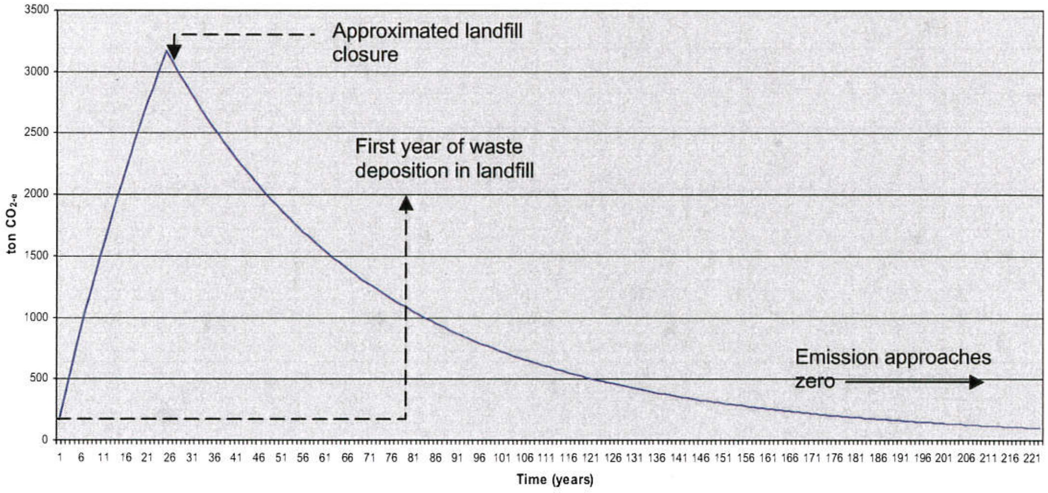

Fig. 1. General trend of CH_4 emission from landfills in their operating post closure years (calculated using the IPCC 1st order decay model).

lower capture efficiency, which others with a higher DOC such as paper or waste sludge are able to achieve higher capture efficiencies due to the amount of biogas produced (DCC, 2008).

Knowing the efficiency of biogas capture can have enormous implications for the GHG mitigation potential. This is illustrated by a study in Phuket, where a 50% recovery of LFG was recorded which led to a GWP reduction around 58% of the total landfill's GWP (Liamsanguan and Gheewala, 2008). This study suggests the possible large mitigation potential a landfill can bring if gas recovery efficiency was improved. Energy recovery in particular was found to have a net negative GWP (Themelis and Ulloa, 2006) and offered the most cost-effective method in reducing GHG emissions from landfills making it a promising WM option with regards to GHG mitigation (Pipatti and Wihersaari, 1998).

2.2. Aerobic landfilling (Aerox landfills)

As oppose to the "dry tomb" approach of traditional landfills where MSW decomposition is inhibited, some have opted to speed up the rate of decomposition in situ by introducing air into landfills (USEPA, 1998; Cossu, 2003). This effectively converts the anaerobic site to an aerobic state, acting like a composter. Aerobic respiring bacteria converts the organic fraction of MSW to mostly CO_2 and H_2O instead of CH_4, reducing the GWP of the landfill (Read et al., 2001a). Studies have shown a reduction of CH_4 concentration by at least 80% at the start of the aerobic process and remained consistently below 15% (Read et al., 2001b), with 90% reduction achieved in some cases (Cossu, 2003).

Aerobic landfilling was initiated with the principle objectives of: (1) leachate control (Read et al., 2001a), and (2) rapid waste stabilization (Stessel and Murphy, 1992; Read et al., 2001b). One application of aerobic landfills is bioreactors which incorporate leachate recirculation within the landfill to accelerate waste decomposition (Read and Hudgins, 2000). This can increased the LGF collection efficiency by 10% from LFG capture without leachate recirculation (Barlez et al., 2003). However, despite a greater rate of CH_4 production and energy recovery, prolong recirculation of leachate has been shown to trigger nitrous oxide (N_2O) production even when recirculation has ceased (Watzinger et al., 2005; Lee et al., 2002). With N_2O having a GWP 310 time of CO_2, this would be significant. The amount of leachate and the rate of its circulation will have to be monitored closely to ensure N_2O emission is minimized. Watzinger et al. (2005) has reported the threshold for leachate application to avoid increased GHG concentration is approximately 200 mm. The amount of leachate application will have to be balanced to ensure that maximum biogas recovery is obtained without causing extensive N_2O production.

Due to increased rate of waste stabilization and LFG gas capture efficiency, energy recovery from aerobic landfills is noted to be higher than anaerobic landfills (Cabaraban et al., 2008). Cabaraban et al. (2008) accounted and energy recovery of 9.6 MJ/kg of waste at a certain point of measurement, while Lou (2008) reported a potential average of 4.108×10^3 MJ/kg of waste recovered over the lifespan of the waste. Although aerobic landfills provide the opportunity for increase energy recovery, there comes the associated fuel consumption needed to introduce air and/or leachate into the landfill, which is absent in a sanitary landfill or a conventional WtE landfill. Energy consumption for aerobic landfill operations is quoted by White et al. (1995) to be 0.198 MJ/kg of waste, while that for conventional landfill was reported to be 3.99×10^{-3} MJ/kg of waste (Baky and Eriksson, 2003). This shows that the fuel consumption and hence GHG emissions from aerobic landfills is likely to be considerably higher than the operational requirement of conventional landfills. Whether this elevated GHG emissions from increased operational requirements can be offset by energy recovery should be quantified with further studies.

Depending on the landfill site, aeration of the landfill may be feasible at different stages of landfill operation. Early aeration means that WtE is forfeited, but may be suitable for landfills where WtE is unfeasible. Late aeration is more common as it allows for energy recovery and continues to mitigate CH_4 emissions when the production of CH_4 has plateau and is no longer cost-effective to continue operation. Should aerobic landfilling be carried out, consideration should be made to the type of energy, fossil fuel or alternatives, needed to operate the facility. As WtE is no longer an option once air is introduced, it may be that aerobic landfilling emits more GHG than mitigated in the grand scheme, should fossil fuel be used as the energy source. Hence, it would also be valuable to evaluate the net GHG mitigation potential of aerobic landfilling, taking into account external factors such as operational energy emissions, to determine if aerobic landfilling is truly sustainable.

In semi-arid climates, natural decomposition is slow, resulting in large C storage and little GHG emissions in the short-medium term (Bogner et al., 2007; Humer and Lechner, 1999b). Aerobic landfilling will thus cause more GHG emissions in the short-term as degradation is accelerated. Conversely, GWP will be reduced significantly in the tropics as degradation is rapid anyway (Kumar et al., 2004). Several European and Asian countries have deployed their own aerobic landfills studies (Hudgins, 2000). It would be useful to consolidate these findings to evaluate the effectiveness of aerobic landfilling on CH_4 emission reduction under different climates.

2.3. Landfill cover – compost application

Following closure, a landfill continues to emit LFG, which poses odour problems and increases the GWP of the landfill (Tchobanoglous and O'Leary, 1994). As a result, landfill covers are sought to resolve this. Conventional methods of degasifying closed landfills rely on non-organic materials, such as clay, which is designed to limit infiltration into the waste, thereby retarding decomposition in an attempt to stifle gaseous production and emissions into the atmosphere (Sadek et al., 2007; Chanton and Liptay, 2000). This has inherent problems, as these inert materials are bound to break down eventually, especially in semi-arid climates (Sadek et al., 2007), reducing the cover's efficiency. Hence, biological covers such as compost are turned to as an alternative landfill cover.

As oppose to clay covers, compost covers aim to improve water filtration into the landfill. This infiltration aids in the decomposition of waste and establishment of an optimal functional methanotrophic community in the compost that oxidises CH_4 into CO_2 (Stern et al., 2007). Although CH_4 is generated within the anaerobic regions, the majority of its uptake is performed by aerobes which are most densely located throughout the upper layer of the compost (Whalen et al., 1990). The first major field trial assessment of the CH_4 removal effectiveness of compost as a landfill cover is ongoing at an Austrian landfill (Humer and Lechner, 1999b, 2001). Results showed 100% CH_4 capture when compost is used in conjunction with gravel under bed, and 10% to 50% CH_4 capture when only compost was used (Humer and Lechner, 2001). Subsequent various studies have supported the concept of biological CH_4 removal via oxidation as an effective GHG reduction tool in landfill emissions (Stern et al., 2007; Hilger and Humer, 2003; Barlaz et al., 2004; Brown and Subler, 2007) and certainly an improvement from the clay cover with respect to CH_4 mitigation (Chanton and Liptay, 2000). Literature shows CH_4 removal rates ranges from 10% to 100% (Humer and Lechner, 2001; Stern et al., 2007; Zeiss, 2006; Chanton and Liptay, 2000; Whalen et al., 1990). This wide variation can be attributed to the various factors, such as temperature, nutrient level and moisture, controlling CH_4 oxidation (Borjesson et al., 2004). Hence, manipulation of these factors can contribute to increase in CH_4 oxidation, resulting in reduced GHG emissions.

In addition, negative CH_4 fluxes have also been reported in compost landfill cover indicating that the landfill surface is consuming, rather than emitting CH_4 (Bogner and Spokas, 1995; Bogner et al., 1997). Field verification of landfill cover soils functioning as CH_4 sink has profound implication for revision of landfill contribution to global CH_4 budget (Bogner and Spokas, 1995) and climate prediction models (Hilger and Humer, 2003).

Microbial oxidation of CH_4 in compost cover is a low-cost alternative to the conventional synthetic capping method in mitigating GHG from closed landfills (Humer and Lechner, 1999a). However, they are not suitable in all situations. For example, compost functionality may be inhibited in cold climates. Experiments by Zeiss (2006) observed a halt in microbial oxidation during the winter period, and did not resume oxidation until a month later. Zeiss (2006) suggested a method of filter bed heating, which resulted in double the emissions reduction in cold climate. However, considerations should be made as to whether operation of this heating device will offset the benefits achieved from the CH_4 reduction through the compost cover, with respect to GWP.

Further research of biotic cover is needed to address its durability and operational lifespan would directly influence the time span of which GHG is being mitigated in the landfill. It would also be of interest to investigate different strands of methanotrophs that may be inoculated into the compost to enhance CH_4 uptake. With the increased interest in phytoremediation, it would also be valuable to investigate the impact of compost cover on the ability of the vegetation in reducing landfill emissions. Would the compost cover enhance the growth of the vegetation? Or would the vegetation compete with the C consuming microbes for nutrients and water, so that the overall CH_4 consumption is decreased? Irregardless, compost cover remains as a competitive option in mitigating landfill CH_4 emissions under certain conditions.

2.4. Pre-composting prior to landfilling

There are two primary objectives for pre-treatment of waste before landfill disposal: (1) rapid stabilization of in order to reduce the landfilled waste volume, thereby prolonging landfill lifespan (Slater and Frederickson, 2001), and (2) to enhance initial methanogenic conditions in landfills (Komilis et al., 1999b; Leikam and Stegmann, 1999). In the course of attaining these objectives, overall CH_4 reduction is achieved (Leikam and Stegmann, 1999; Rieger and Bidlingmaier, 1995). This is achieved via 2 processes:

1. Pre-treatment induces aerobic environment which decomposes the organic fraction of MSW rapidly, releasing CO_2 gas and H_2O instead of CH_4. Binner (1995) found that pre-treating MSW halves the organic C of untreated waste. Thus, when this is later landfilled, CH_4 production is lessened.
2. As pre-treatment attains a thermophilic state, which favours rapid establishment of methanogenesis, moving this waste to an anaerobic condition (landfill) will temporary enhance CH_4 production (Brummeler and Koster, 1990; Brinkmann et al., 1995). This observation is supported by studies by Stegmann (1983), Stegmann and Spendlin (1987) and Norbu et al (2005), where results correlated on an increase in CH_4 production in landfills within the first few years with pre-treated waste as compared to one without. However, as composting has removed a portion of DOC, the total CH_4 production from pre-composted waste will be lessened as compared to a landfill that receives fresh waste.

With regards to overall GHG balance, it is generally agreed that a reduction of total gas production up to 90% can be achieved through pre-composting (Stegemann et al., 1995; Zach et al., 2000; Leikam and Stegmann, 1999). According to Tränkler and Ranaweera (2002), overall gaseous formation could be reduced by more than 35% and GWP would be abated by more than 63%, within 20 years of using pre-treatment technology in a landfill. However, further considerations have to be taken before implementation. With respect to process (1), although methanogenesis is enhanced in the landfill following pre-treatment, owing to the mineralization of some organic matter during composting, CH_4 yield are lower as compared to untreated MSW (Stegmann, 1983; Stegmann and Spendlin, 1987). Reduction in CH_4 production when MSW are later landfilled implies that gas recovery may not be economical if pre-treatment proceeds. Therefore, the duration of the pre-composting step is crucial to the overall GHG mitigation potential.

A short pre-treatment will allow some methanogenesis to occur in the landfill. Whether or not this remains economical is unknown. A longer pre-treatment will stabilize the waste to such a large extent that CH_4 potential is reduced significantly. A value of up to 90% reduction in CH_4 production was reported by Zach et al. (2000) and Stegemann et al. (1995). The time period to establish waste stabilization and hence reduced CH_4 production, varies widely between countries due to different climate influenced factors. For example, Komilis et al. (1999b) in Germany suggested a pre-treatment of one to two months to allow some generation of CH_4 while Norbu et al. (2005) in Thailand suggested a pre-treatment period of approximately one week, due to more rapid decomposition in tropical climate.

It can be seen that the suitability and duration of anaerobic pre-treatment of waste prior to landfilling are dependent on the LFG recovery objectives. This technology may be appropriate for landfills unsuitable for LFG recovery; in which MSW can be pre-treated till highly stabilized. This allows significant reduction in gaseous emissions and alternative strategies (such as landfill cover) can be sought to further mitigate any residual CH_4 emissions.

3. Composting

Composting has been concerned with the efficient and economic production of a product, while functioning within a larger process oriented approach to solid WM (Zeman et al., 2002; Murphy and Power, 2006). One of the conundrums which has emerged due to this product versus process, is that much of the composting research has been focused with the goal of improving production and the quality of the finished product, while the issue of composting process's contribution to GHG emissions is often neglected.

Aerobic decomposition from well managed composting results in the emission of CO_2 and H_2O. Due to the heterogeneous nature of a compost pile, some CH_4 may form in anaerobic pockets within the pile (Bogner et al., 2007; Brown and Subler, 2007). However, studies have shown that the majority of this CH_4 emission oxidizes to CO_2 in aerobic pockets and near the surface of the compost pile, making CH_4 emission negligible (Zeman et al., 2002; Brown and Leonard, 2004). This claim is refuted by Edelmann and Josss (1999), Amlinger et al. (2008) and Rynk and Richards (2001), who reported considerable CH_4 emission even in well managed systems. This is probably due to various variables controlling the nature of the compost piles.

GHG emission from the composting process is highly dependent on the feedstock. Feedstock for composting ranges from greenwaste to MSW to manure. Waste stream with a higher DOC will result in higher GHG emissions from aerobic decomposition, while waste with lower rate decay constant will require a longer period of composting, which may cause a difference in the operational requirements to compost that waste stream. Theoretical calculations by White et al. (1995) estimated approximately 0.323 ton CO_{2-e}/ton of mixed waste, while that from Lou (2008) estimated a value of 0.284 ton CO_{2-e}/ton of mixed waste, taking into account the emissions from operational activities. Practical GHG measurements from alternatives studies have produced emissions from compositing operations ranging from 0.183 ton CO_{2-e}/ton of mixed waste to 193.2 ton CO_{2-e}/ton of mixed waste (Jackson and Line, 1997; Jakobsen, 1994) which are lower than those predicted from theoretical calculations. This suggests theoretical estimates may tend to overestimate what is actually emitted. These emission estimates suggests the importance of operational activities on the impact of composting on GHG emissions. Composting tends to occur via windrows composting or aerobic in-vessel composting, the former tending to have lower associated operating emissions (White et al., 1995; Lou, 2008). For example, aerobic in-vessel composting has been reported to consume 0.198 MJ/kg of waste, while windrows composting consumed a reasonable reduced 0.071 MJ/kg of waste (White et al., 1995). Emissions from operational activities vary with reports due to the type of composting operations, the amount of waste treated and other variations, an average emission from operation of composting if thus difficult to quantify. As a brief indication of the emissions from windrow operations, White et al. (1995) reported 0.071 MJ/kg of waste, Baky and Eriksson (2003) estimated 0.00151 MJ/kg of waste, and Franklin Associates (1994) estimated 0.134 MJ/kg of waste. In order to obtain an accurate estimation of a composting facility GHG emission, it would be best to undertake a site specific GHG inventory instead of taking an average from the literature.

Recently, there has been a renewed interest in the emission of N_2O from composting (Beck-Friis et al., 2000; He et al., 2000). N_2O can be produced by either incomplete ammonium oxidation or via incomplete denitrification (Beck-Friis et al., 2000). Results of N_2O generation have been mixed. He et al., (2000) reported high emissions at the beginning of composting, and negligible amount after the initial 2 days, while Beck-Friis et al., (2000) reported higher emissions with time. Little is known regarding N_2O emission, however the fact that it has a GWP 310 times that of CO_2, makes its study justifiable (IPCC, 2006).

Despite potential CH_4 and N_2O emissions, these are usually not taken into account in C budgeting. In fact in the IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 2006) and the Australian Methodology for the Estimation of Greenhouse Gas Emissions and Sinks (DCC 2007), only CO_2 is assumed to be emitted during composting. Even so, although this CO_2 is climate active when released, it is generally not considered when it comes to GHG accounting. This is due to the general consensus that CO_2 from composting is of biogenic origin and hence does not add to the overall GHG emissions that contribute to global warming (IPCC, 2006). Therefore, when C accounting is performed by composting facilities, their GHG contribution is a result of operational activities, as opposed to the actual decomposition process. Nevertheless, there are some that advocate the inclusion of all GHG emissions derived from non-fossil C (Hogg et al., 2008). The rationale behind this inclusion is that the climate responds the same whether the source is no differently to fossil or non-fossil C, and thus all emissions should be accounted for on a like-for-like basis.

Due to the exclusion of CO_2 from composting as a GHG contribution, there have been relatively fewer studies devoted to composting contribution to GHG emissions than for landfills; and the need for this is often questioned by the composting waste sectors. This is because, unlike landfilling, where there is increasing pressure to quantify their GHG contribution, there is little legislative pressure for composting facilities to quantify their GHG contribution (Wong, 2008). However, from an environmental perspective, the negligence of composting contribution to GHG emissions is questionable, especially from a life cycle analysis approach. This is especially relevant to centralized composting systems where operational activities are energy consuming (ROU, 2007). The question is how can composting contribution to GHG emissions be objectively quantified using a life cycle approach that will be acceptable to the scientific and managerial community? Through logical reasoning, most waste managers and literature agree that with regards to landfilling, the majority of their GHG impact lies with the degradation of waste so much that, emissions from operational activities are negligible (Chen and Lin, 2007; Wilson, 2002). The converse may be said for composting. However, there has been few comparative studies between landfilling and composting, utilizing a life cycle approach to justify this statement.

Some centralized composting facilities are driven to produce high quality compost that can be used for land application (Wei et al., 2000; Murphy and Power, 2006). The production of compost, especially high quality compost, requires energy. Source separation of MSW received at the composting facility is needed prior to landfilling. This activity is carried out using heavy machineries which makes emissions of GHG inevitable. Machines are essential to grind and mix the waste material, set up compost piles, aeration *inter alia*. All these processes require energy which will have an associated equivalent GHG cost (Brown and Subler, 2007). An alternative would be to initial source separation at household level, therefore reducing the need for heavy machinery centralized separation and the associated GHG emissions (ROU, 2007). It should also be noted that, no matter how much waste is composted and recycled, a portion of waste will still remain that requires landfilling.

Nonetheless, the production of compost can help mitigate GHG by:

1. Decreasing the need of chemical fertilisers and pesticides; thereby reducing GHG emissions from the use of fossil fuel associated with their production and application (Cogger, 2005; Favoino and Hogg, 2008).
2. Allowing for more rapid growth in plants, thereby increasing C uptake and storage within the plant. Studies have shown the increase biomass of plants growth with compost application (Gonzalez and Cooperband, 2002; Lynch et al., 2005; Wei et al., 2000). This is a form of C sequestration which removes CO_2 from the atmosphere.
3. Sequestering C in soil that has received compost application (Favoino and Hogg, 2008; Mondini et al., 2007). The USEPA (2002) estimated approximately 50 kg C (183 kg CO_2) sequestered per ton of wet compost. On a large scale, this may be significant.
4. Improving tillage and workability of soil (thereby reducing emissions from fossil fuel that would otherwise be used to work the soil) (Favoino and Hogg 2008).

With the exception of C sequestration (point 3), these offsets have been recognized but actual quantification of them remains incomplete. The USEPA has derived an estimated figure of approximately 50 kg C (183 kg CO_2) sequestered per ton of wet compost (USEPA, 2002). This figure is however specific to the US, and to a particular soil type.

4. Carbon sequestration

Although C sequestration is not within the scope of this review, mention should be made as its impact on GHG mitigation is relevant to the landfilling strategies and land application of compost mentioned above. Landfills can be great C sinks owing to slow degradation of waste (Bogner et al., 2007; Pipatti and Wihersaari, 1998). The amount of C stored in a landfill is dependent on the deposited waste stream's DOC and their rate constant, both of which can be obtain from the IPCC (2006). Waste material that tend to degrade rapidly such as food waste will not remain stable as stored C, while other materials such as wood finds difficulty in decomposing under anaerobic conditions in landfill but is easily compostable under aerobic composting (US Composting Council, 2008). This suggests the need to direct certain waste to appropriate waste management facilities in order to degrade waste efficiently. Compost application to land also allows for C sequestration. Although this provides a short-medium term solution to climate change mitigation, it may be perceived as simply a manner of buying time (Kirschbaum, 2006). The importance of C sequestration is represented by Weitz et al. (2002) calculations which demonstrated that when C storage is considered in determining the net GHG emitted, it dramatically offsets the emissions from landfill operations. This is supported by many who suggest the probability of landfilling as a more competitive option with respect to climate change when C storage is included; especially when coupled with WtE (Pipatti and Wihersaari, 1998; Bogner et al., 2007). This would greatly influence how landfilling strategies are perceived with regards to climate change mitigation.

5. Conclusion

Despite the various waste management strategies ability to reduce GHG emissions from conventional landfills, a certain amount of GHG will inevitably be produced. This literature review suggests GHG emissions from waste decomposition are greatly higher for landfills than for composting system. This is because anaerobic decomposition facilitates the production of CH_4 which has a GWP 25 times that of CO_2, Energy recovery and appropriate landfill capping strategies can help to reduce this impact, but composting still remains a simpler and effective mean of reducing GHG emissions. This brings about the fundamental difference between landfilling and composting – landfilling, together with its mitigation strategies are often reactive measures in environmental protection, seeking to remediate harmful effects it has caused. Conversely, composting adopts a more proactive approached, with an objective to prevent or minimise such negative impacts in the first place.

With regards to emissions from operational activities, results are mixed. Depending on the landfill operations and composting operations, composting may emit more GHG emissions than landfill activities. For example, aerobic in-vessel composting has approximately doubled the operational fuel consumption than aerobic landfills. In general, when the sum of GHG emissions is considered, factoring decomposition and operational emissions, landfills appears to have a heavier impact on GHG emissions than composting. However, the existence and continual usage of landfills now and in the future cannot be denied. Not all waste can be composted or recycled, and a certain portion of waste will inevitable be landfilled. Nonetheless, countries should always aim to reduce the amount of waste produced and divert waste to appropriate waste management facilities so as to ensure overall operational and environmental sustainability.

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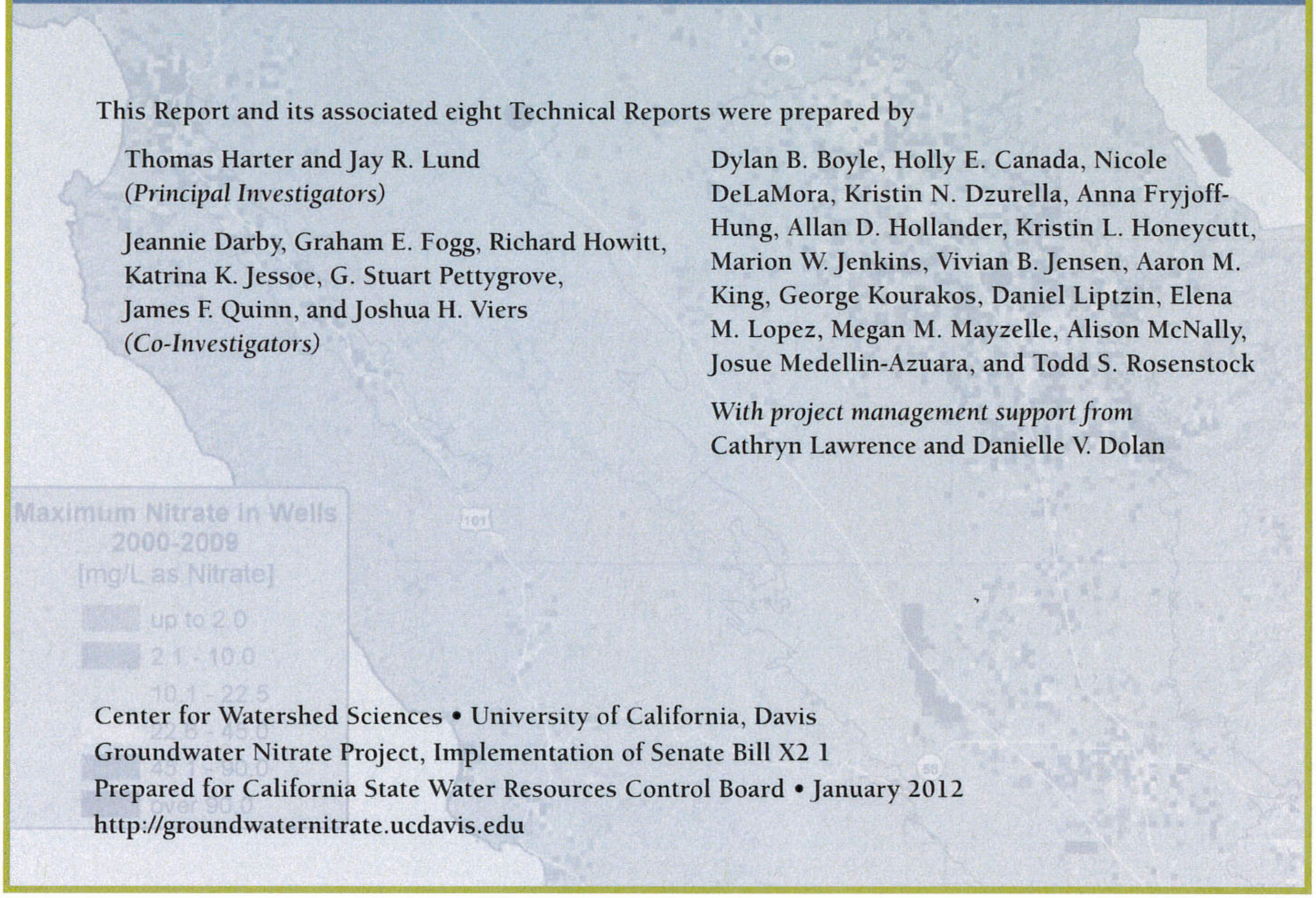
Addressing Nitrate in California's Drinking Water

With a Focus on Tulare Lake Basin and Salinas Valley Groundwater

Report for the State Water Resources Control Board Report to the Legislature

EXECUTIVE SUMMARY



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Groundwater Nitrate Project, Implementation of Senate Bill X2 1
Prepared for California State Water Resources Control Board • January 2012
http://groundwaternitrate.ucdavis.edu

Executive Summary

In 2008, Senate Bill SBX2 1 (Perata) was signed into law (Water Code Section 83002.5), requiring the State Water Resources Control Board (State Water Board), in consultation with other agencies, to prepare a Report to the Legislature to "improve understanding of the causes of [nitrate] groundwater contamination, identify potential remediation solutions and funding sources to recover costs expended by the State…to clean up or treat groundwater, and ensure the provision of safe drinking water to all communities." The University of California prepared this Report under contract with the State Water Board as it prepares its Report to the Legislature.

This executive summary focuses on major findings and promising actions. Details can be found in the Main Report and eight accompanying Technical Reports.

Key Issues

Groundwater is essential to California, and nitrate is one of the state's most widespread groundwater contaminants. Nitrate in groundwater is principally a by-product of nitrogen use, a key input to agricultural production. However, too much intake of nitrate through drinking water can harm human health.

California's governments, communities, and agricultural industry have struggled over nitrate contamination for decades. **The California Department of Public Health (CDPH) has set the maximum contaminant level (MCL)** for nitrate in drinking water at 45 milligrams per liter (as nitrate). Nitrate concentrations in public drinking water supplies exceeding the MCL require water system actions to provide safe drinking water.

For this study, the four-county **Tulare Lake Basin and the Monterey County portion of the Salinas Valley are examined**. About 2.6 million people in these regions rely on groundwater for drinking water. The study area includes four of the nation's five counties with the largest agricultural production. It represents about 40% of California's irrigated cropland (including 80 different crops) and over half of California's dairy herd. Many communities in the area are among the poorest in California and have limited economic means or technical capacity to maintain safe drinking water given threats from nitrate and other contaminants.

Summary of Key Findings

1. **Nitrate problems will likely worsen for several decades. For more than half a century, nitrate from fertilizer and animal waste have infiltrated into Tulare Lake Basin and Salinas Valley aquifers. Most nitrate in drinking water wells today was applied to the surface decades ago.**
2. **Agricultural fertilizers and animal wastes applied to cropland are by far the largest regional sources of nitrate in groundwater. Other sources can be locally relevant.**
3. **Nitrate loading reductions are possible, some at modest cost. Large reductions of nitrate loads to groundwater can have substantial economic cost.**
4. **Direct remediation to remove nitrate from large groundwater basins is extremely costly and not technically feasible. Instead, "pump-and-fertilize" and improved groundwater recharge management are less costly long-term alternatives.**
5. **Drinking water supply actions such as blending, treatment, and alternative water supplies are most cost-effective. Blending will become less available in many cases as nitrate pollution continues to spread.**
6. **Many small communities cannot afford safe drinking water treatment and supply actions. High fixed costs affect small systems disproportionately.**
7. **The most promising revenue source is a fee on nitrogen fertilizer use in these basins. A nitrogen fertilizer use fee could compensate affected small communities for mitigation expenses and effects of nitrate pollution.**
8. **Inconsistency and inaccessibility of data prevent effective and continuous assessment. A statewide effort is needed to integrate diverse water-related data collection activities by many state and local agencies.**

Nitrate in groundwater poses two major problems and risks:

- **Public health concerns** for those exposed to nitrate contamination in drinking water; in California's Tulare Lake Basin and Salinas Valley, roughly 254,000 people are currently at risk for nitrate contamination of their drinking water. Of these, 220,000 are connected to community public (>14 connections) or state small water systems (5–14 connections), and 34,000 are served by private domestic wells or other systems smaller than the threshold for state or county regulation and which are largely unmonitored.
- **Financial costs of nitrate contamination** include additional drinking water treatment, new wells, monitoring, or other safe drinking water actions; over 1.3 million people are financially susceptible because nitrate in raw source water exceeds the MCL, requiring actions by drinking water systems. Nitrate contamination of drinking water sources will continue to increase as nitrogen from fertilizer, manure, and other sources applied in the last half century continues to percolate downward and flow toward drinking water wells.

Findings: Sources of Nitrate Pollution

Within the study area, human-generated nitrate sources to groundwater include (Figure ES-1):

- cropland (96% of total), where nitrogen applied to crops, but not removed by harvest, air emission, or runoff, is leached from the root zone to groundwater. Nitrogen intentionally or incidentally applied to cropland includes synthetic fertilizer (54%), animal manure (33%), irrigation source water (8%), atmospheric deposition (3%), and wastewater treatment and food processing facility effluent and associated solids (2%) (Figure ES-2);
- percolation of wastewater treatment plant (WWTP) and food processing (FP) wastes (1.5% of total);
- leachate from septic system drainfields (1% of total);
- urban parks, lawns, golf courses, and leaky sewer systems (less than 1% of total); and
- recharge from animal corrals and manure storage lagoons (less than 1% of total);
- downward migration of nitrate-contaminated water via wells (less than 1% of total).

Findings: Reducing Nitrate Pollution

Options for reducing nitrate pollution were identified for all sources. For cropland, where less than 40% of applied nitrogen is removed by crop harvest, 10 management measures (and 50 practices and technologies to achieve these management objectives) were reviewed that can reduce—but not eliminate—nitrate leaching to groundwater. These fall into four categories:

1. Design and operate irrigation and drainage systems to reduce deep percolation.
2. Manage crop plants to capture more nitrogen and decrease deep percolation.
3. Manage nitrogen fertilizer and manure to increase crop nitrogen use efficiency.
4. Improve storage and handling of fertilizers and manure to decrease off-target discharge.

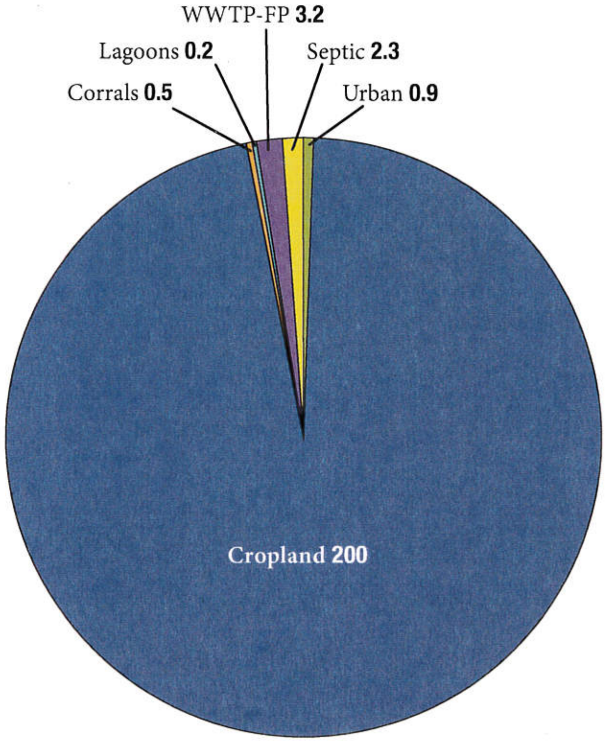
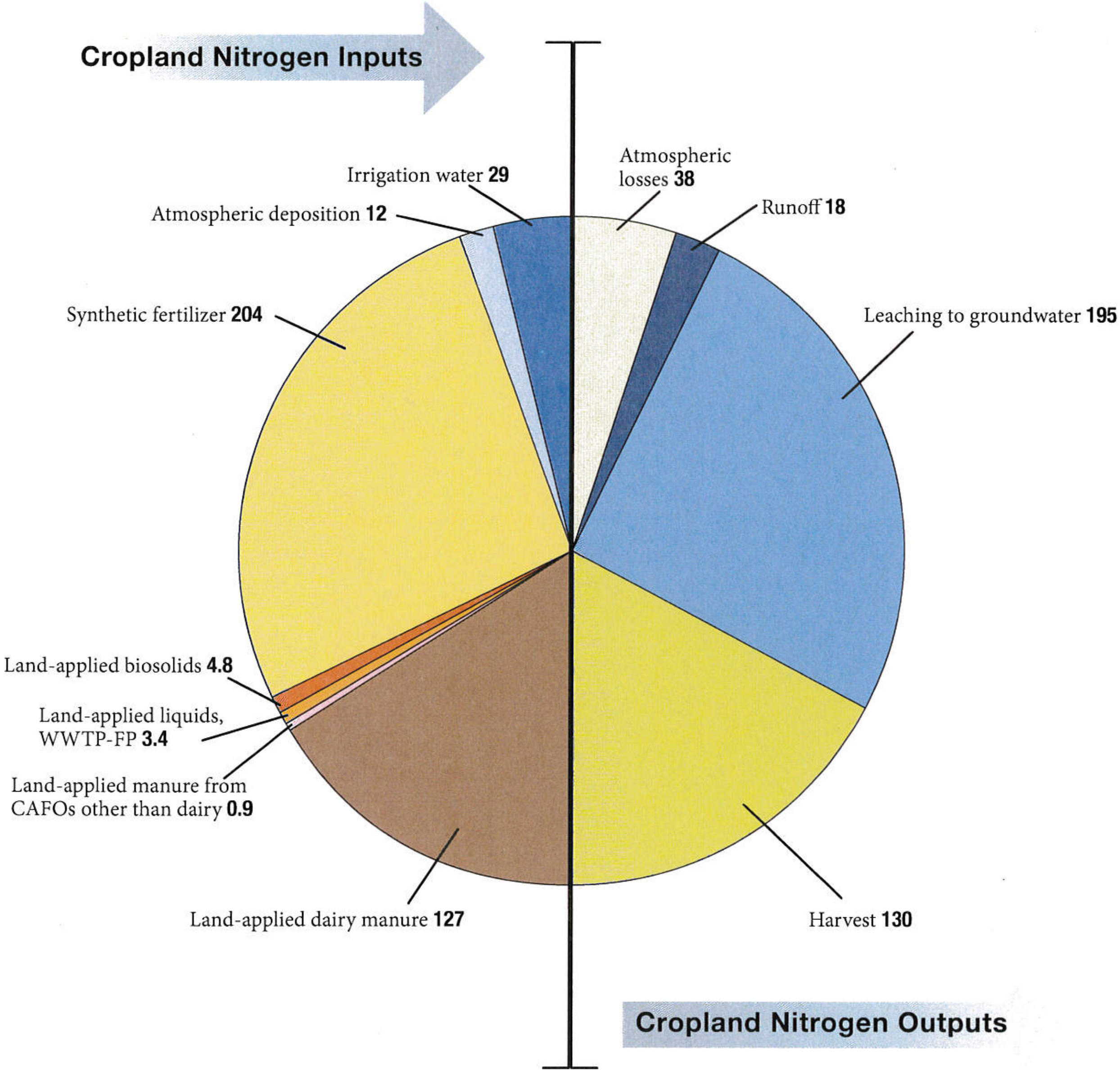


Figure ES-1. Estimated groundwater nitrate loading from major sources within the Tulare Lake Basin and Salinas Valley, in Gg nitrogen per year (1 Gg = 1,100 t).



Note: No mass balance was performed on 0.17 million ha (0.4 million ac) of nitrogen-fixing alfalfa, which is estimated to contribute an additional 5 Gg N/yr to groundwater. Groundwater nitrate loading from all non-cropland sources is about 8 Gg N/yr.

Figure ES-2. Overview of cropland input and output (Gg N/yr) in the study area (Tulare Lake Basin and Salinas Valley) in 2005. The left half of the pie chart represents total nitrogen inputs to 1.27 million ha (3.12 million ac) of cropland, not including alfalfa. The right half of the pie chart represents total nitrogen outputs with leaching to groundwater estimated by difference between the known inputs and the known outputs. Source: Viers et al. 2012.

Some of the needed improvements in nitrogen use efficiency by crops will require increased operating costs, capital improvements, and education. For some cropland, the high economic costs of nitrate source reduction sufficient to prevent groundwater degradation will likely hinder strict compliance with the state's current anti-degradation policy for groundwater (State Water Board Resolution 68-16).

Findings: Groundwater Nitrate Pollution

Groundwater nitrate data were assembled from nearly two dozen agencies and other sources (100,000 samples from nearly 20,000 wells). Of the 20,000 wells, 2,500 are frequently sampled public water supply wells (over 60,000 samples). In these public supply wells, about 1 in 10 raw water samples exceed the nitrate MCL. Apart from the recently established Central Valley dairy regulatory program in the Tulare Lake Basin, there are no existing regular well sampling programs for domestic and other private wells.

The largest percentages of groundwater nitrate MCL exceedances are in the eastern Tulare Lake Basin and in the northern, eastern, and central Salinas Valley, where about one-third of tested domestic and irrigation wells exceed the MCL. These same areas have seen a significant increase in nitrate concentrations over the past half century, although local conditions and short-term trends vary widely.

Travel times of nitrate from source to wells range from a few years to decades in domestic wells, and from years to many decades and even centuries in deeper production wells. This means that nitrate source reduction actions made today may not affect sources of drinking water for years to many decades.

Findings: Groundwater Remediation

Groundwater remediation is the cleanup of contaminated groundwater to within regulatory limits. Traditional pump-and-treat and in-place approaches to remediation, common for localized industrial contamination plumes, would cost billions of dollars over many decades to remove nitrate from groundwater in the Tulare Lake Basin and Salinas Valley. Timely cleanup of basin-scale nitrate contamination is not technically feasible.

Instead, long-term remediation by "pump-and-fertilize" would use existing agricultural wells to gradually remove nitrate-contaminated groundwater and treat the water by ensuring nitrate uptake by crops through appropriate nutrient and irrigation water management. Improved groundwater recharge management would provide clean groundwater recharge to mix with irrigation water recharge and partially mitigate nitrate levels in groundwater regionally.

Removal or reduction of contamination sources must accompany any successful remediation effort. Combining "pump-and-fertilize" with improved groundwater recharge management is more technically feasible and cost-effective.

Findings: Safe Drinking Water Supply

Nitrate contamination is widespread and increasing. Groundwater data show that 57% of the current population in the study area use a community public water system with recorded raw (untreated) nitrate concentrations that have exceeded the MCL at least once between 2006 and 2010. Continued basin-wide trends in nitrate groundwater concentration may raise the affected population to nearly 80% by 2050. Most of this population is protected by water system treatment, or alternative wells, at additional cost. But about 10% of the current population is at risk of nitrate contamination in their delivered drinking water, primarily in small systems and self-supplied households.

No single solution will fit every community affected by nitrate in groundwater. Each affected water system requires individual engineering and financial analyses.

Communities served by small systems vulnerable to nitrate contamination can (a) consolidate with a larger system that can provide safe drinking water to more customers; (b) consolidate with nearby small systems into a new single larger system that has a larger ratepayer base and economies of scale; (c) treat the contaminated water source; (d) switch to surface water; (e) use interim bottled water or point-of-use treatment until an approved long-term solution can be implemented; (f) drill a new well; or (g) blend contaminated wells with cleaner sources, at least temporarily.

There is significant engineering and economic potential for consolidating some systems. Consolidation can often permanently address nitrate problems, as well as many other problems faced by small water systems.

Solutions for self-supplied households (domestic well) or local small water systems (2–4 connections) affected by nitrate contamination are point-of-use (POU) or point-of-entry (POE) treatment and drilling a new or deeper well, albeit with no guarantee for safe drinking water.

Additional costs for safe drinking water solutions to nitrate contamination in the Tulare Lake Basin and Salinas Valley are roughly $20 and $36 million per year for the short- and long-term solutions, respectively. About $17 to $34 million per year will be needed to provide safe drinking water for 85 identified community public and state small water systems in the study area that exceed the nitrate drinking water MCL (serving an estimated 220,000 people). The annualized cost of providing nitrate-compliant drinking water to an estimated 10,000 affected rural households (34,000 people) using private domestic wells or local small water systems is estimated to be at least $2.5 million for point-of-use treatment for drinking use only. The total cost for alternative solutions translates to $80 to $142 per affected person per year, $5 to $9 per irrigated acre per year, or $100 to $180 per ton of fertilizer nitrogen applied in these groundwater basins.

Findings: Regulatory, Funding, and Policy Options

To date, regulatory actions have been insufficient to control nitrate contamination of groundwater. Many options exist to regulate nitrate loading to groundwater, with no ideal solution. Nitrate source reductions will improve drinking water quality only after years to decades. Fertilizer regulations have lower monitoring and enforcement costs and information requirements than do nitrate leachate regulations, but they achieve nitrate reduction targets less directly. Costs to farmers can be lower with fertilizer fees or market-based regulations than with technology mandates or prescriptive standards. Market-based approaches may also encourage the development and adoption of new technologies to reduce fertilizer use.

Current funding programs cannot ensure safe drinking water in the Salinas Valley and Tulare Lake Basin. Small water system costs are high, and some of these systems already face chronic financial problems. Most current state funding for nitrate contamination problems is short term. Little funding is provided for regionalization and consolidation of drinking water systems. Policy options exist for long-term funding of safe drinking water, but all existing and potential options will require someone to bear the costs.

Promising Actions

Addressing groundwater nitrate contamination requires actions in four areas: (a) safe drinking water actions for affected areas, (b) reducing sources of nitrate contamination to groundwater, (c) monitoring and assessment of groundwater and drinking water, and (d) revenues to help fund solutions. Promising actions for legislative and state agency consideration in these areas appear below (see also Table ES-1). Starred (*) actions do not appear to require legislative action, but might benefit from it.

Safe Drinking Water Actions (D)

Safe drinking water actions are the most effective and economical short- and long-term approach to address nitrate contamination problems in the Tulare Lake Basin and Salinas Valley. These actions apply especially to small and self-supplied household water systems, which face the greatest financial and public health problems from nitrate groundwater contamination.

D1: Point-of-Use (POU) Treatment Option. CDPH reports on how to make economical household and point-of-use treatment for nitrate contamination an available and permanent solution for small water systems.*

D2: Small Water System Task Force. CalEPA and CDPH convene an independently led Task Force on Small Water Systems that would report on problems and solutions of small water and wastewater systems statewide as well as the efficacy of various state, county, and federal programs to aid small water and wastewater systems. Many nitrate contamination problems are symptomatic of the broad problems of small water and wastewater systems.*

D3: Regional Consolidation. CDPH and counties provide more legal, technical, and funding support for preparing consolidation of small water systems with nearby larger systems and creating new, regional safe drinking water solutions for groups of small water systems, where cost-effective.*

D4: Domestic Well Testing. In areas identified as being at risk for nitrate contamination by the California Water Boards, as a public health requirement, CDPH (a) mandates periodic nitrate testing for private domestic wells and local and state small systems and (b) requires disclosure of recent well tests for nitrate contamination on sales of residential property. County health departments also might impose such requirements.

D5: Stable Small System Funds. CDPH receives more stable funding to help support capital and operation and maintenance costs for new, cost-effective and sustainable safe drinking water solutions, particularly for disadvantaged communities (DACs).

Source Reduction Actions (S)

Reducing nitrate loading to groundwater is possible, sometimes at a modest expense. But nitrate source reduction works slowly and cannot effectively restore all affected aquifers to drinking water quality. Within the framework of Porter-Cologne, unless groundwater were to be de-designated as a drinking water source, reduction of nitrate loading to groundwater is required to improve long-term water quality. The following options seem most promising to reduce nitrate loading.

S1: Education and Research. California Department of Food and Agriculture (CDFA), in cooperation with the University of California and other organizations, develops and delivers a comprehensive educational and technical program to help farmers improve efficiency in nitrogen use (including manure) and reduce nitrate loading to groundwater. This could include a groundwater nitrate–focused element for the existing CDFA Fertilizer Research and Education Program, including "pump-and-fertilize" remediation and improved recharge options for groundwater cleanup.*

S2: Nitrogen Mass Accounting Task Force. CalEPA establishes a Task Force, including CDFA, to explore nitrogen mass balance accounting methods for regulating agricultural land uses in areas at risk for nitrate contamination, and to compare three long-term nitrogen source control approaches: (a) a cap and trade system; (b) farm-level nutrient management plans, standards, and penalties; and (c) nitrogen fertilizer fees.*

S3: Fertilizer Excise Fee. Significantly raising the cost of commercial fertilizer through a fee or excise tax would fund safe drinking water actions and monitoring and give further incentive to farmers for reducing nitrate contamination. An equivalent fee or excise tax could be considered for organic fertilizer sources (manure, green waste, wastewater effluent, biosolids, etc.).

S4: Higher Fertilizer Fee in Areas at Risk. Areas declared to be at risk for nitrate contamination might be authorized to maintain a higher set of excise fees on nitrogen fertilizer applications (including synthetic fertilizer, manure, waste effluent, biosolids, and organic amendments), perhaps as part of a local safe drinking water compensation agreement.

Monitoring and Assessment (M)

Monitoring and assessment is needed to better assess the evolving nitrate pollution problem and the effectiveness of safe drinking water and nitrate source loading reduction actions. Such activities should be integrated with other state agricultural, environmental, and land use management; groundwater data; and assessment programs (source loading reduction actions)—along with other drinking water, treatment, and wastewater management programs (safe drinking water actions).

M1: Define Areas at Risk. Regional Water Boards designate areas where groundwater sources of drinking water are at risk of being contaminated by nitrate.*

M2: Monitor at-Risk Population. CDPH and the State Water Board, in coordination with DWR and CDFA, issue a report every 5 years to identify populations at risk of contaminated drinking water and to monitor long-term trends of the state's success in providing safe drinking water as a supplement to the California Water Plan Update.*

M3: Learn from Department of Pesticide Regulation Programs. CalEPA and CDFA examine successful DPR data collection, analysis, education, and enforcement programs for lessons in managing nitrogen and other agricultural contaminants, and consider expanding or building upon the existing DPR program to include comprehensive nitrogen use reporting to support nitrate discharge management.*

M4: Groundwater Data Task Force. CalEPA , in coordination with CalNRA and CDPH, convenes an independently led State Groundwater Data Task Force to examine the efficacy of current state and local efforts to collect, maintain, report, and use groundwater data for California's groundwater quality and quantity problems.

M5: Groundwater Task Force. CalEPA, CalNRA, and CDPH maintain a joint, permanent, and independently led State Groundwater Task Force to periodically assess and coordinate state technical and regulatory groundwater programs in terms of effectiveness at addressing California's groundwater quality and quantity problems. These reports would be incorporated into each California Water Plan Update.*

Funding (F)

Little effective action can occur without funding. Four funding options seem most promising, individually or in combination. State funding from fees on nitrogen or water use, which directly affect nitrate groundwater contamination, seem particularly promising and appropriate.

F1: Mill Fee. Increase the mill assessment rate on nitrogen fertilizer to the full authorized amount (CAL. FAC Code Section 14611). This would raise roughly $1 million/year statewide and is authorized for fertilizer use research and education.*

F2: Local Compensation Agreements. Regional Water Boards can require and arrange for local compensation of affected drinking water users under Porter-Cologne Act Water Code Section 13304. Strengthening existing authority, the Legislature could require that a Regional Water Board finding that an area is at risk of groundwater nitrate contamination for drinking water be accompanied by a cleanup and abatement order requiring overlying, current sources of nitrate to financially support safe drinking water actions acceptable to the local County Health Department. This might take the form of a local "liability district."*

F3: Fertilizer Excise Fee. Introduce a substantial fee on nitrogen fertilizer sales or use, statewide or regionally, to fund safe drinking water actions, nitrate source load reduction efforts, and nitrate monitoring and assessment programs.

F4: Water Use Fee. A more comprehensive statewide fee on water use could support many beneficial activities. Some of such revenues could fund management and safe drinking water actions in areas affected by nitrate contamination, including short-term emergency drinking water measures for disadvantaged communities.

Table ES-1. Likely performance of promising state and agency actions for nitrate groundwater contamination.

Action	Safe Drinking Water	Groundwater Degradation	Economic Cost
No Legislation Required			
Safe Drinking Water Actions			
D1: Point-of-Use Treatment Option for Small Systems +	◆◆		low
D2: Small Water Systems Task Force +	◆		low
D3: Regionalization and Consolidation of Small Systems +	◆◆		low
Source Reduction Actions			
S1: Nitrogen/Nitrate Education and Research +		◆◆◆	low–moderate
S2: Nitrogen Accounting Task Force +		◆◆	low
Monitoring and Assessment			
M1: Regional Boards Define Areas at Risk +	◆◆◆	◆◆◆	low
M2: CDPH Monitors At-Risk Population +	◆	◆	low
M3: Implement Nitrogen Use Reporting +		◆◆	low
M4: Groundwater Data Task Force +	◆	◆	low
M5: Groundwater Task Force +	◆	◆	low
Funding			
F1: Nitrogen Fertilizer Mill Fee		◆◆◆	low
F2: Local Compensation Agreements for Water +	◆◆	◆	moderate
New Legislation Required			
D4: Domestic Well Testing *	◆◆		low
D5: Stable Small System Funds	◆		moderate
Non-tax legislation could also strengthen and augment existing authority.			
Fiscal Legislation Required			
Source Reduction			
S3: Fertilizer Excise Fee	◆◆	◆	moderate
S4: Higher Fertilizer Fee in Areas at Risk	◆	◆	moderate
Funding Options			
F3: Fertilizer Excise Fee	◆◆	◆◆	moderate
F4: Water Use Fee	◆◆	◆◆	moderate

◆ Helpful

◆◆ Effective

◆◆◆ Essential

+ Legislation would strengthen.

* County health departments may have authority; CDPH requires legislation.

COMPOST FUNDAMENTALS

Compost Benefits & Uses

Compost Benefits

Using compost as mulch, in the soil or as potting media is beneficial in many ways.

Compost contains a full spectrum of essential plant nutrients. You can test the nutrient levels in your compost and soil to find out what other supplements it may need for specific plants.

- Compost contains macro and micronutrients often absent in synthetic fertilizers.
- Compost releases nutrients slowly—over months or years, unlike synthetic fertilizers
- Compost enriched soil retains fertilizers better. Less fertilizer runs off to pollute waterways.
- Compost buffers the soil, neutralizing both acid & alkaline soils, bringing pH levels to the optimum range for nutrient availability to plants.

Compost helps bind clusters of soil particles, called aggregates, which provide good soil structure. Such soil is full of tiny air channels & pores that hold air, moisture and nutrients.

- Compost helps sandy soil retain water and nutrients.
- Compost loosens tightly bound particles in clay or silt soil so roots can spread, water drain & air penetrate.
- Compost alters soil structure, making it less likely to erode, and prevents soil spattering on plants—spreading disease.
- Compost can hold nutrients tight enough to prevent them from washing out, but loosely enough so plants can take them up as needed.
- Compost makes any soil easier to work.

Compost brings and feeds diverse life in the soil. These bacteria, fungi, insects, worms and more support healthy plant growth.

Benefits & Uses

economic aspects

testing and judging condition of compost

quality of composts

benefits of compost

use of compost

- Compost bacteria break down organics into plant available nutrients. Some bacteria convert nitrogen from the air into a plant available nutrient.
- Compost enriched soil have lots of beneficial insects, worms and other organisms that burrow through soil keeping it well aerated.
- Compost may suppress diseases and harmful pests that could overrun poor, lifeless soil.

Healthy soil is an important factor in protecting our waters. Compost increases soil's ability to retain water & decreases runoff. Runoff pollutes water by carrying soil, fertilizers and pesticides to nearby streams.

- Compost encourages healthy root systems, which decrease runoff
- Compost can reduce or eliminate use of synthetic fertilizers
- Compost can reduce chemical pesticides since it contains beneficial microorganisms that may protect plants from diseases and pests.
- Only a 5% increase in organic material quadruples soils water holding capacity.

When that first batch of finished compost is ready to spread, congratulate yourself for your efforts because you are ecological minded, and know that organic materials should be recycled into the soil instead of being put in a garbage can. By recycling the organic materials, valuable nutrients and organic matter are recycled. You have helped alleviate the solid waste problem!

Why Compost | Biology & Chemistry | Compost Needs
Composter's Needs | Benefits & Uses | Conclusion

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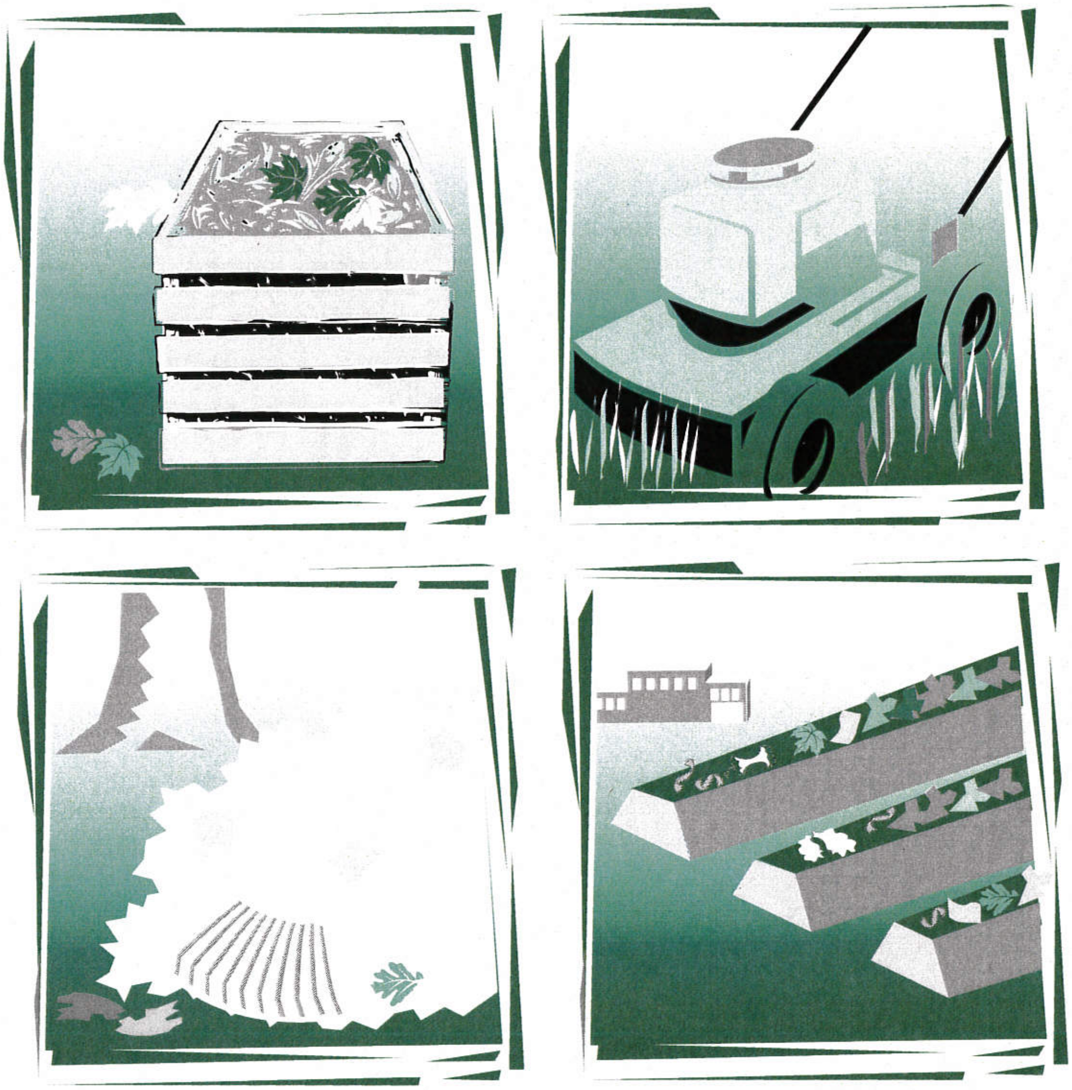

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4. COMPOST MARKETS AND PRODUCT VALUE

4.1 Review of Benefits Associated With Compost End-Uses

The demand for finished compost helps divert an increasing amount of organic materials from landfills. In addition, the use and application of finished compost result in a multitude of benefits, such as enhancing the physical, chemical, and biological properties of soils, which in turn results in various environmental and economic benefits. A summary of some of the major benefits of composting is provided below.[54]

4.1.1 Direct Benefits to Soil

- **Improves the Physical Properties of Soils**. Compost enhances water holding, soil aeration, structural stability, resistance to water and wind erosion, root penetration, and soil temperature stabilization.
- **Enhances the Chemical Properties of Soils**. Compost increases macro- and micronutrient content, increases availability of mineral substances, ensures pH stability, and provides a long-term source of nutrient input by acting as a nutrient reservoir.
- **Improves the Biological Properties of Soils**. Compost promotes the activity of beneficial micro-organisms, reduces attack by parasites, promotes faster root development, and promotes higher yields of agricultural crops.

4.1.2 Indirect Environmental and Economic Benefits

- Since compost has the ability to improve soil water holding capacity and fix nitrogen into a form that can be used by plants, its use mitigates (at least partially) nonpoint sources of pollution such as commercial fertilizers.
- By improving soil water holding capacity and reducing water loss as a result of percolation, evaporation, and runoff, compost application results in water conservation benefits.
- Compost reduces reliance on pesticides, herbicides, and fungicides by providing an environment rich in organic matter. Beneficial micro-organisms thrive in this environment and can outcompete and suppress detrimental pathogens found in soils where organic matter is low.
- Consistent application of compost reduces soil erosion resulting from wind and water by improving soil stability.

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[54] Based on Pratt, W., and W. Shireman. 1994. *Agricultural Markets for Compost and Mulch: Cost, Benefits, and Policy Recommendations.* California Futures, Sacramento, California. Also based on Rhode Island Solid Waste Management Corporation. 1991. *End Use of Leaf and Yard Waste Compost.* Prepared by Tellus Institute. For more information on characteristics and benefits of compost, see *Markets for Compost*, EPA. 1993.