LCA application to case studies and the use of dynamic models to improve food production modelling

Doctoral Dissertation
Doctoral Program in PRODUCTION SYSTEMS & INDUSTRIAL DESIGN

LCA APPLICATION TO CASE STUDIES AND THE USE OF DYNAMIC MODELS TO IMPROVE FOOD PRODUCTION MODELLING

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February 01, 2013

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Turin, February 01, 2013

CONTENTS

1. DESCRIPTION OF THE THESIS SUBJECT

Global food demand has constantly increased over the last 50 years and a 100% increase in demand is forecasted by 2050 (Tilman et al. 2011) propelled by the expected increase of global population, per capita incomes and consequent changes of dietary habits (Hochman et al. 2013). Biofuels and biomaterials are also expected to cause a further increase in demand for agricultural products, especially if current policies remain in place. For example, the Renewable Energy Directive (European Commission 2009), aiming at substituting 10% of fossil fuels used for transport with biofuels by 2020, is acting as a driver in bioenergy production growth but is also highlighting the need to adopt more sustainable farming practices especially due to the sustainability criteria it is adopting (35% GHG reduction compared to equivalent fossil fuels).

The required growth in agricultural production can be obtained by both clearing of new land and a more intensive use of existing croplands, but the potential to increase production by expanding farmed areas is limited due to the competition between forests, nature reserves and urban areas. Consequently, 90% of the growth in crop production will need to come from intensification, i.e. from higher yields per hectare and increased cropping intensity (Bruinsma 2009).

Agricultural and farming systems are already perceived as contributors to the degradation of the environment and as important drivers of both global (climate change) and regional (e.g. eutrophication) impacts. Their expected intensification is more and more attracting the attention of policymakers. In fact, in 2005, agriculture accounted for 10-12% of total global anthropogenic emissions of greenhouse gases, 60% of global anthropogenic N_2O and 50% of CH_4. Globally, agricultural CH_4 and N_2O emissions have increased by nearly 17% from 1990 to 2005 (Smith et al. 2007). It is also the most significant source of surface and ground water quality degradation (Moreau et al. 2012) as well as an important factor influencing soil degradation and freshwater reserve depletion. The European Union demonstrated attention especially towards water quality with the Water Framework Directive (European Commission 2000) committing all member states to achieve good qualitative and quantitative status of all water bodies by 2015.

Solutions trying to mitigate these impacts, but disregarding the complex dependencies between processes are likely to fail, thus Life Cycle Assessment (LCA), providing a holistic approach that considers potential impacts of all stages of production, has been recommended by the European Union as a valuable environmental assessment tool.

In the past years, LCA has been widely used to analyse agricultural systems at local (Blengini and Busto 2009), regional (Haas et al. 2001), national (Renouf et al. 2010) and supranational scales (Weiss and Leip 2012). Most studies generally aim at (i) estimating environmental impacts and resource consumptions, (ii) understanding where impacts are more concentrated (hot-spot analysis) and (iii) comparing different production alternatives (scenario analysis). LCA's global approach is also used to analyse the outcome of policies and to assess if improvement scenarios are successful in reducing overall impacts.

The development of more sustainable products and processes requires comprehensive and reliable decision support systems which are based on data. Life cycle assessment (LCA) is an established methodology for quantifying potential environmental impacts and can feed into decision support systems for improved environmental choices by manufacturers, policy makers and consumers.

Studying only one specific aspect of a problem at a time, instead, does not lead to an understanding of the full picture. The result could be that improvement scenarios just shift impacts (temporally or geographically) from one productive phase to another (burden shifting) or just move them from one impact to another (trade-offs).

LCA has demonstrated its strength as an environmental assessment and management tool, but its application to agricultural systems also revealed some of its inherent weak points and is driving current methodological developments.

For instance, the all too often used default assumption that emissions are proportional to inputs is often violated while dealing with direct emissions from agricultural fields (Bessou et al. 2012). For example, nitrous oxide (N_2O) emissions to atmosphere are strongly influenced by the variability of local soils, weather conditions and management factors and therefore highly variable at field scale. Hence the use of emission factor approaches (e.g. IPCC tier 1), correlating N inputs with emissions and disregarding other influencing parameters, are associated to a high level of uncertainty. To overcome this, LCA has been coupled with dynamic simulation models such as DNDC (Li et al. 1992) or CERES (Gabrielle et al. 2006) capable of grasping temporal and spatial variability of emissions (Bessou et al. 2012; Fukushima and Chen 2009).

Another interesting weak point of LCA application to agriculture is related to pesticide use. Pesticides are widely used in agriculture and usually spread in large quantities often in a short period of time. This leads to elevated concentration of different chemical substances being released at the same time in environmental matrices. Pesticides are likely to have an effect on a broad range of organisms, no matter whether these organisms are the intended target for the applied plant protection chemical or not (Dijkman et al. 2012). That said, accounting for the inherent toxicity of these products is extremely complicated as the fate of pesticide products after their spread on the field can be heavily influenced by soil, weather and crop management conditions. Moreover to accurately measure their impacts, knowledge over timing and spatial distribution of pesticide applications should be available.

Usually LCA of agricultural products either neglect the fate of pesticides after spreading or assume that the full dose of applied pesticide is emitted to one environmental compartment. For example, in Ecoinvent database (SwissCentre for LifeCycle Inventories 2011) it is assumed that the full pesticide dose is emitted to soil (Nemecek and Kägi 2007). To overcome this limitation specific models simulating pesticide fate have been recently developed. An example is PestLCI (Birkved and Hauschild 2006; Dijkman et al. 2012), able to account for spatially and temporal variability of pesticide fate according to local parameters.

The main objective of this PhD thesis is to contribute to LCA's methodology development by improving its accuracy when modeling production processes dealing with or based on agricultural products.

To proceed in this path, my PhD thesis developed in three main directions:

1. **LCA application to case studies**; A first, focusing on determining life cycle impacts of alternative agri-food chain management systems to produce rice in North West Italy (Piedmont). A second, assessing environmental performances of a recycled foam glass (RFG) to be used in high efficiency thermally insulating and lightweight concrete. A third, about two extensive applications of LCA to the integrated municipal solid waste management systems of Torino and Cuneo Districts in northern Italy.
2. **DNDC application case studies**; application and further development of DNDC, a dynamic model capable of estimating direct emissions of cultivated fields. As part of my job assignment at JRC, the model was setup to perform simulations at European-wide scale to predict emissions of GHG of EU crop productions. Model runs were performed in the framework of two FP projects (NitroEurope

and CCTAME) focusing on nitrogen based emissions (N_2O in particular) at EU scale. In a first paper, the possibility to replace the so-called Tier 1 IPCC approach to estimate soil N_2O emissions was investigated. Stratified emissions factors taking into account both N-input and the spatial variability of the environmental conditions within the countries of the European Union were calculated. In a second, bottom-up results from studies providing N_2O fluxes at a regional/country or continental scale were compared with estimates from the process-based model DNDC-EUROPE and from the TM5-4DVAR inverse modeling system.

3. **Coupling LCA with dynamic models**; coupling LCA with DNDC and other dynamic simulation models to account for temporal and spatial variability of emissions and to overcome the limitations of emission factor approaches. My work upon Italian rice was used as starting point. This time, instead of following an emission factor approach, dynamic models were used. Geo-referenced soil, weather and crop management data were gathered collaborating with agronomy faculty and through a literature review. Piedmont rice area was divided in 2877 geographical units characterized by homogeneous soil and crop management parameters and, for each unit, data from the nearest weather station was used. This amount of data was then fed to DNDC and PestLCI models estimating field GHG emissions and pesticides fate respectively.

2. LCA APPLICATION CASE STUDIES

2.1. THE LIFE CYCLE OF RICE: LCA OF ALTERNATIVE AGRI-FOOD CHAIN MANAGEMENT SYSTEMS IN VERCELLI (ITALY).

The Vercelli rice district in northern Italy plays a key role in the agri-food industry in a country which accounts for more than 50% of the EU rice production and exports roughly 70%. However, although wealth and jobs are created, the sector is said to be responsible for environmental impacts that are increasingly being perceived as topical. As a complex and comprehensive environmental evaluation is necessary to understand and manage the environmental impact of the agri-food chain, the Life Cycle Assessment (LCA) methodology has been applied to the rice production system: from the paddy field to the supermarket.

Impact indicator	Unit	White milled rice (50 ha)
GER	MJ	15.61
NRER	MJ	14.63
GWP100	kg CO_2eq	3.18
ODP	mg CFC11eq	0.11
AP	mol H+	0.26
EP	g O_2eq	329
POCP	g C_2H_4eq	0.71
WU	l	4 978

TABLE 1 CATEGORY INDICATOR FOR CONVENTIONAL RICE (50 HA FARM)

As it can be seen in Table 1, the production and delivery of 1 kg of exported white milled rice from the 50 ha rice farm requires 17.8 MJ of energy resources of which 16.6 MJ are non renewable. The GWP_{100} indicator shows a carbon dioxide equivalent emission of 2.9 kg, which seems to be in contrast with the value of 1.1 kg CO2eq reported in the Italian Greenhouse Gas Inventory (APAT, 2005). However, the difference can be explained in terms of life cycle phases and system boundaries. In fact, the greenhouse emission of rice, according to APAT (2005), corresponds to direct methane emissions from the paddy field: 48 g of CH4 multiplied by a characterization factor of 23. However, when adding up direct and indirect greenhouse emissions relevant to the subsequent life cycle steps and when considering the loss of weight after drying, as well as when allocating impacts between the refined rice and its by-products, the GWP indicator rises to almost 3 kg of CO2eq per kg of delivered white milled rice.

The direct use of water for irrigation appears to be particularly intense: almost 5 m3 per 1 kg of delivered rice. However, if the indirect use of fresh water is also considered, the WU_t indicator would rise to around 8 m^3/kg. The result is not far away from that reported in Oki et al. (2003) which have estimated the "irrigation water requirement" of rice in Japan.

Production phase	Subsystem	GER (%)	NRER (%)	GWP (%)	ODP (%)	AP (%)	EP (%)	POCP (%)
Agricultural	Mechanical field operations	10.1	10.4	3.2	11.6	11.0	2.2	0.4
	Fertilizers	34.5	35.8	7.9	44.1	20.1	13.8	1.2
	Pesticides	2.3	2.4	0.4	2.0	0.9	0.1	0.1
	Field emissions	0.0	0.0	73.5	0.0	43.1	77.3	93.3
	Seeds	3.2	3.2	3.3	3.3	3.4	3.3	3.3
	Capital goods	6.4	6.2	1.4	4.2	2.6	0.4	0.2
	Phase total	*56.4*	*58.0*	*89.5*	*65.1*	*81.2*	*97.1*	*98.5*
Post harvest processing	Rice drying and storing	9.5	9.9	2.9	10.5	2.6	0.3	0.5
	Rice processing and packaging	28.7	26.4	6.1	18.7	12.0	1.8	0.8
	Phase total	*38.2*	*36.3*	*8.9*	*29.2*	*14.6*	*2.1*	*1.2*
Transport	Field to farm	0.03	0.03	0.01	0.03	0.02	0.00	0.00
	Farm to processing plant	0.8	0.8	0.2	0.8	0.6	0.1	0.1
	Processing to local distribution	1.0	1.0	0.3	1.2	0.7	0.2	0.1
	Local to national distribution	3.6	3.8	1.0	3.7	2.9	0.6	0.2
	Phase total	*5.4*	*5.7*	*1.5*	*5.7*	*4.2*	*0.8*	*0.3*

TABLE 2 CONTRIBUTION ANALYSIS OF CONVENTIONAL RICE (50 HA)

In Table 6 contribution analysis for conventional rice is shown. Fertilizers production is the greatest contributor to the gross energy requirement (30%) and this is followed by refining and packing (25%) and transportation (17%). Global warming is mainly influenced by field emissions (68%) and then by fertilizers (9%) and transportation (6%). Paddy field emissions have the greatest impact on four indicators (GWP, AP, EP, POCP), thus emphasizing the need for further reliable and site specific data. As expected, direct water use is dominated by irrigation (97%), the remaining 3% being used for seed production. The total water use is also dominated by irrigation, but 18% is indirectly used for the production of packaging materials.

The agricultural phase has generally shown the most important contributions to the final impacts, thus representing an environmental hot spot. Nevertheless, the post-harvest processing showed remarkable contributions, therefore identifying further areas of potential improvement, mainly in terms of energy saving and reduction of the ozone depletion and acidification potentials.

As far as transportation is concerned, it should be noticed that there is a remarkable contribution for energy and ODP (17-18%), a contribution which is lower for GWP (6%) and negligible for the remaining indicators.

As the contribution of capital goods was considered a meaningful issue, it should be mentioned that they have a noticeable weight (6%) on energy requirement and WU_t, the contribution to ODP, AP and GWP being 3.9%, 2.4% and 1.6%, respectively. The contribution to EP and POCP is less than 1%.

Improvement scenarios have been analyzed considering alternative rice farming and food processing methods, such as organic and upland farming, as well as parboiling Figure 1 Comparison between alternative rice farming and processing (Figure 1). The research has shown that organic and upland farming have the potential to decrease the impact per unit of cultivated area. However, due to the lower grain yields, the environmental benefits per kg of the final products are greatly reduced in the case of upland rice production and almost cancelled for organic rice.

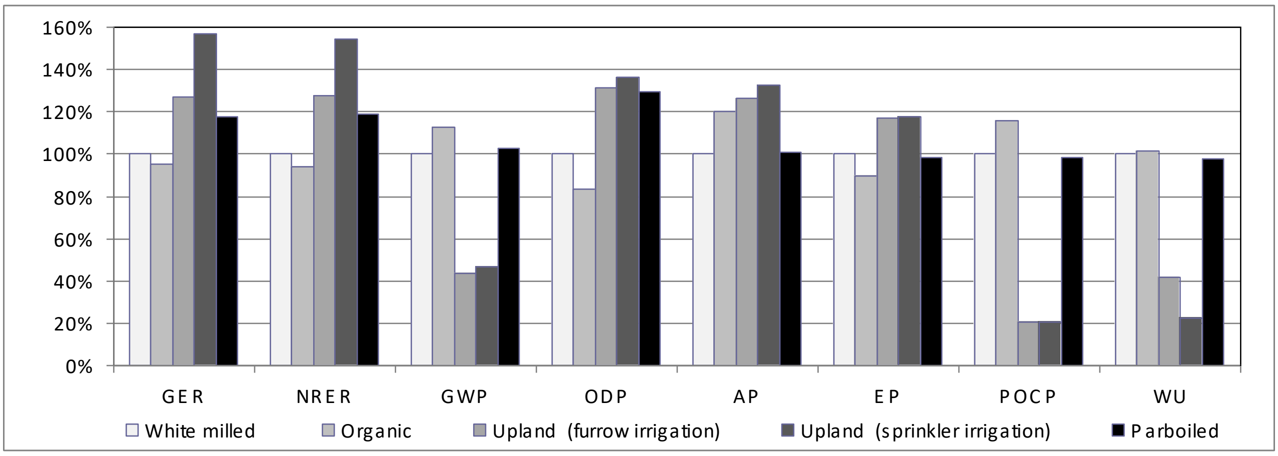


FIGURE 1 COMPARISON BETWEEN ALTERNATIVE RICE FARMING AND PROCESSING

2.2. *ECO-EFFICIENT WASTE GLASS RECYCLING: INTEGRATED WASTE MANAGEMENT AND GREEN PRODUCT DEVELOPMENT THROUGH LCA.*

As part of the EU Life + NOVEDI project, a new eco-efficient recycling route has been implemented to maximize resources and energy recovery from post-consumer waste glass, through integrated waste management and industrial production. Life cycle assessment (LCA) has been used to identify engineering solutions to sustainability during the development of green building products.

The green building product is a recycled foam glass (RFG) to be used in high efficiency thermally insulating and lightweight concrete.

The new process and the related LCA are framed within a meaningful case of industrial symbiosis, where multiple waste streams are utilized in a multi-output industrial process. The input is a mix of rejected waste glass from conventional container glass recycling and waste special glass such as monitor glass, bulbs and glass fibers. System boundaries of RFG production are shown in Figure 2.

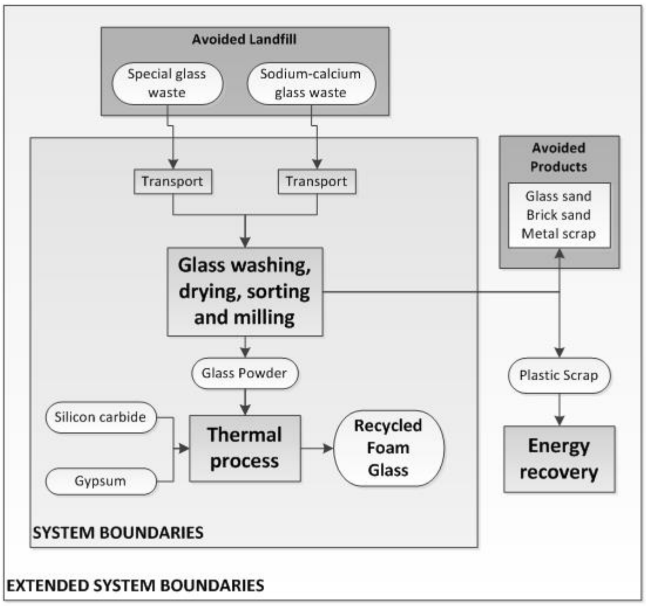


FIGURE 2 RFG SYSTEM BOUNDARIES

Glass waste used in RFG production has different origins that can be grouped into two main categories: Sodium-calcium glass and special glass. The sodium-calcium to special glass ratio can vary depending on technical and economic considerations.

The environmental gains have been contrasted against induced impacts and improvements have been proposed. RFG produced from Mix 1 (50% of sodium-calcium glass and 50% of special glass) was chosen as a baseline scenario.

Table 3 displays indicator results relevant to both electric heating (EH) and natural gas heating (NGH). The differences between the environmental performances of the two alternatives are remarkable, accounting up to -51% in the case of Acidification (AP). This can be related to the high indirect environmental impacts related to the Italian energy mix, which is strongly dependent on fossil fuels.

Indicator	RFG-EH	RFG-NGH
EI-99	23 mPt/t	(-35%)
GER	7761 MJ/t	(-34%)
GWP	513 kg CO_{2eq}/t	(-32%)
AP	77 mol H^+/t	(-51%)
EP	7907 g O_{2eq}/t	(-45%)

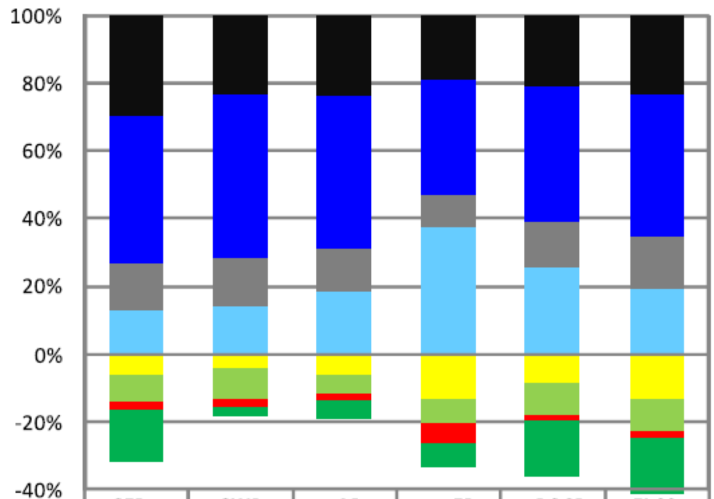
TABLE 3 ECO-PROFILE OF RFG FROM MIX USING EITHER ELECTRIC (EH) OR NATURAL GAS HEATING (NGH)

Impacts are due to transportation, processing and firing, while savings come from avoided landfill and recovery of co-products (Figure 3).

It can be observed that the environmental gains related to the avoided landfill are cancelled by the transport-related impacts. Thus, it is not sufficient to base environmental claims on the statement that RFG is sustainable because it avoids landfilling, as the related gains are lower than the induced impacts.

An important contribution to improve the environmental profile of RFG is represented by recovered plastic, metals and glass fragments/powders, whose environmental gains are higher than those corresponding to landfill avoidance. This suggests that, in order to improve the RFG eco-profile, the raw mix should preferably be made of soda-lime glass rather than special glass, which does not contain recoverable metals and plastics. This finding highlights that industrial symbiosis can play a key role in eco-efficient glass recycling and further supports the recommendation of Hurley (2003) according to which closed-loop container glass recycling remains a preferable option.

Production of SiC and RFG firing represent the highest induced impacts. In spite of the small amount used, SiC is an important contributor to the overall impacts. Consequently, although SiC proved to be an excellent foaming agent (Bernardo et al., 2007), a more environmentally friendly additive is preferable.



	GER (MJ)	GWP (kg CO_2)	AP (mol H+)	EP (g O_2)	POCP (g C_2H_4)	EI-99 (Pt)
Foaming agent	3382	144.6	22.5	2237	4.1	9.1
RFG firing	4928	304.2	42.8	4078	7.9	16.4
Processing	1595	92.7	12.2	1113	2.6	5.8
Transport	1491	86.1	17.8	4458	5.1	7.8
Avoided landfill	-773	-27.7	-6.4	-1605	-1.7	-5.3
Glass recovery	-876	-55.3	-5.0	-881	-1.9	-3.6
Metal recovery	-213	-18.1	-1.8	-663	-0.3	-0.9
Energy recovery	-1772	-13.1	-5.1	-827	-3.1	-6.2

FIGURE 3 INDUCED AND AVOIDED IMPACTS IN THE RFG WASTE-TO-PRODUCTION CHAIN (MIX 1, ELECTRIC HEATING EH)

In summary, recovered co-products, such as glass fragments/powders, plastics and metals, correspond to environmental gains that are higher than those related to landfill avoidance, whereas the latter is cancelled due to increased transportation distances. In accordance to an eco-efficiency principle, it has been highlighted that recourse to highly energy intensive recycling should be limited to waste that cannot be closed-loop recycled.

2.3. Participatory approach, acceptability and transparency of waste management LCAs: case studies of Torino and Cuneo

The paper summarizes the main results obtained from two extensive applications of Life Cycle Assessment (LCA) to the integrated municipal solid waste management systems of Torino and Cuneo Districts in northern Italy.

In both cases, the overall objective was identifying scenarios with best energy and environmental performance. A detailed energy and environmental analysis was carried out for the main components of the integrated waste management systems (I-WMS) and for the I-WMS as a whole in order to support public administrators towards sustainable waste management.

Separate collection (SC) and its downstream recycling chains were investigated in terms of environmental benefits and impacts, in order to quantify advantages and drawbacks that can be ascribed to the new objectives of SC introduced by the law presently in force in Italy (Dlgs.152/06). At the same time, the role and environmental implications of energy recovery from residual waste were analysed, paying attention to the consequences of possible pre-treatment options of the residual waste, and considering both incineration and co-incineration.

With reference to 1 ton of separately collected waste, Figure 4 shows the energy and carbon balances of SC and subsequent recycling/treatment in a life cycle perspective. The sequence of activities starts after collection (not included) and encompasses transportation, selection, recycling/treatment and substitution (avoided products/energy). Both the main waste flows and residues were included in the analysis, whereas residues are either landfilled of sent to energy recovery. Negative indicators mean that environmental gains are higher than induced impacts. Biowaste refers to a mix of composting and anaerobic digestion (AD), which reflects the current situation in Torino (33% AD and 67% composting), while metals refer to a mix of ferrous and non-ferrous metals (detailed data are reported as Supplementary content).

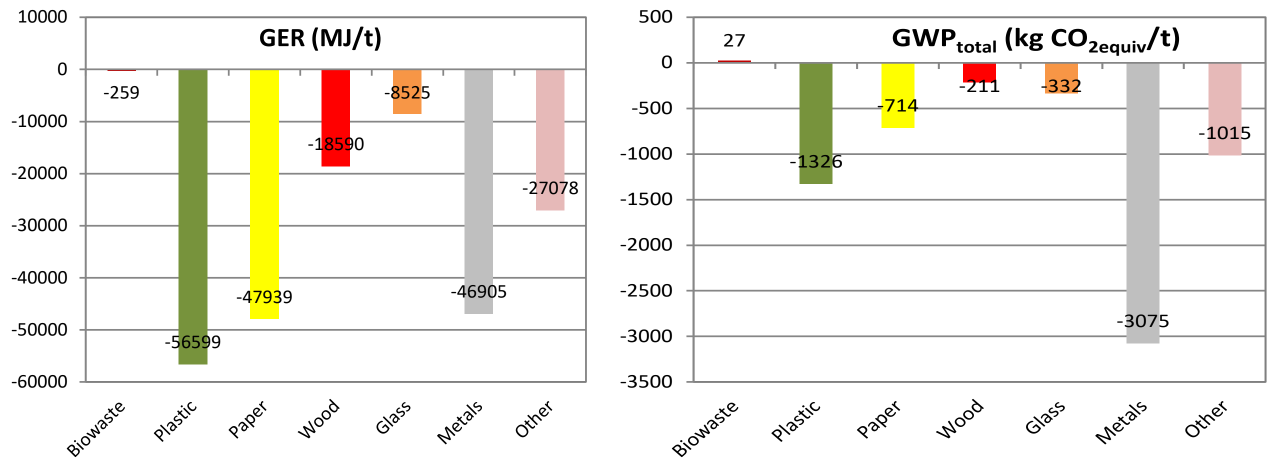


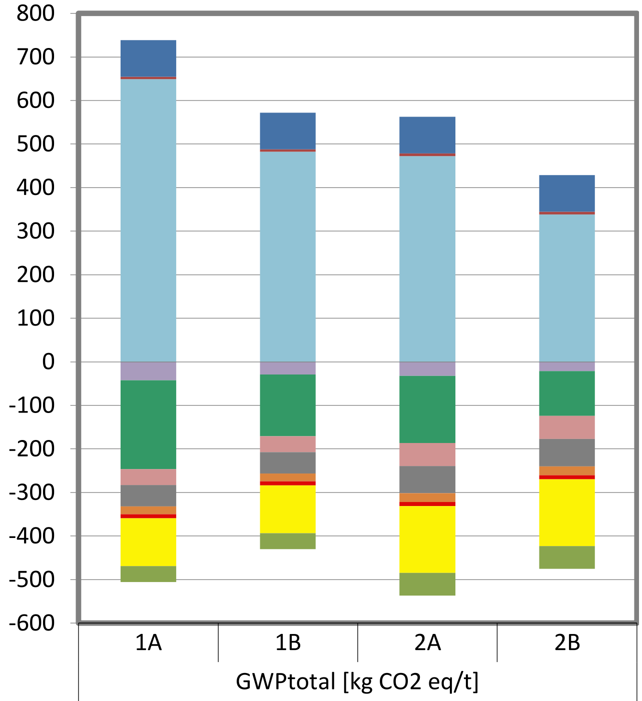
FIGURE 4 ENERGY AND CARBON BALANCE OF SEPARATELY COLLECTED WASTE MATERIALS

With reference to 1 ton of total waste, Table 2 shows the energy and carbon balances related to the four scenarios. According to both energy and climate change indicators, scenarios with 65.6% of separated collection appear to be more eco-efficient than those with 52.1%. Scenarios which include mechanical–biological treatment (MBT) (1B and 2B) show a more favourable carbon balance, but perform worse in terms of energy balance.

Impact Category	Unit	Scenario 1A	Scenario 1B	Scenario 2A	Scenario 2B
GER	MJ/t	-13,898	-12,858	-17,362	-16,497
NRE	MJ/t	-7,476	-6,499	-8,811	-8,001
GWP100total	kg CO_2eq/t	233	142	26	-46
GWP100fossil	kg	-156	-160	-230	-241

CO$_2$eq/t

TABLE 4 ENERGY AND CARBON BALANCE OF THE FOUR SCENARIOS UNDER COMPARISON (TORINO DISTRICT)



	1A	1B	2A	2B
Collection	84	84	84	84
Biowaste treat.	5	5	6	6
Plastic recycling	-37	-37	-52	-52
Paper recycling	-110	-110	-153	-153
Wood recycling	-9	-9	-9	-9
Glass recycling	-18	-18	-20	-20
Metals recycling	-49	-49	-63	-63
Others recycling	-36	-36	-53	-53
Electricity (WTE)	-204	-142	-154	-103
Heat (WTE)	-42	-29	-32	-21
Incineration	649	483	472	339

FIGURE 5 CONTRIBUTION OF SUBSYSTEMS TO THE CARBON BALANCE OF THE I-WMS (TORINO DISTRICT)

3. DNDC APPLICATION CASE STUDIES

3.1. DEVELOPING SPATIALLY STRATIFIED N2O EMISSION FACTORS FOR EUROPE

We investigate the possibility to replace the so-called Tier 1 IPCC approach to estimate soil N2O emissions with stratified emissions factors that take into account both N-input and the spatial variability of the environmental conditions within the countries of the European Union, using the DNDC-Europe model. Spatial variability in model simulations is high and corresponds to the variability reported in literature for field data.

Results indicate that, average simulated N2O fluxes for EU25 over all years and crop types are log-normally distributed (Figure 6) with a geometric mean of 1.8 kg N2O-N ha^{-1} yr^{-1} and a 68% confidence interval (CI; one standard deviation). The average N2O flux is 3.7 kg N2O-Nha^{-1} yr^{-1}.

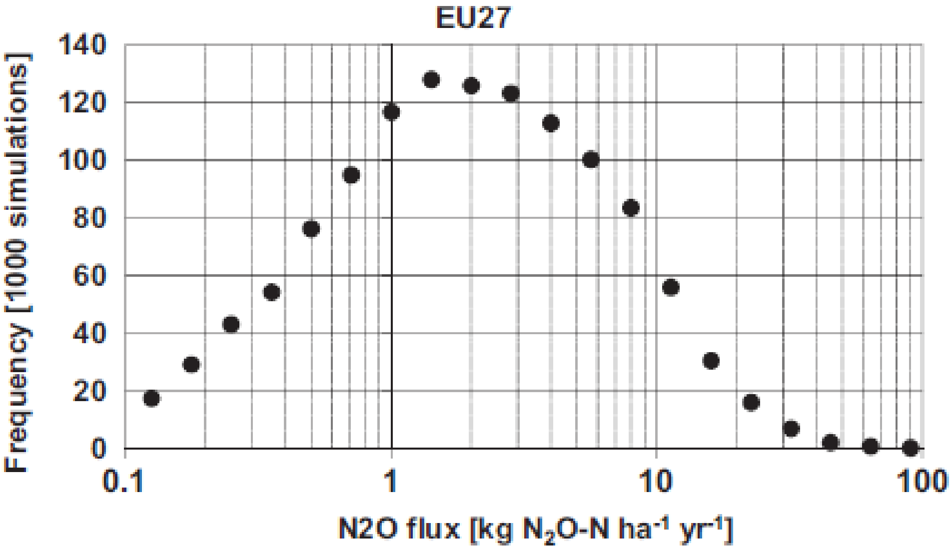


FIGURE 6 HISTOGRAM OF SIMULATED N2O FLUXES ON MINERAL SOILS IN THE REFERENCE SCENARIO. THE SIMULATED FLUXES WERE GROUPED INTO 30 CLASSES ON A LOGARITHMIC SCALE. NUMBERS ARE IN 1000 SIMULATIONS.

For simplicity reasons we used the linear regression model to estimate the rate of fertilizer-induced N2O emissions (FIE) the change of N2O emissions as a consequence of change of fertilizer nitrogen application for the application of mineral fertilizer (FIE_{min}) and manure (FIE_{man}) nitrogen.

Figure 7 shows the FIE_{min} and FIE_{man} for the 25 countries included in the simulation. According to the DNDC-EUROPE model, manure causes a slightly higher release of N2O than mineral fertilizer. At European level, the difference between FIE_{min} and FIE_{man} is about 10% with higher FIE for applied manure than mineral fertilizer. FIE_{man}/ FIE_{min} however increases with decreasing SOC content. For soils with low SOC content, FIE_{min} is only about 0.5%, but FIA_{man} is about 0.8%; on soils with high SOC content, FIE is around 1.8% regardless of the nature of the applied nitrogen. This effect can be explained by lack of anaerobic conditions in soils under dry meteorological situations with low SOC content, since the microbial activity necessary to deplete the oxygen is constrained by the lack of carbon substrate. As manure application adds carbon as well as nitrogen to the soil system, it enhances the carbon turnover processes with a higher rate of oxygen consumption and thus increased probability of the existence of anaerobic micro-sites which favour the production of N2O.

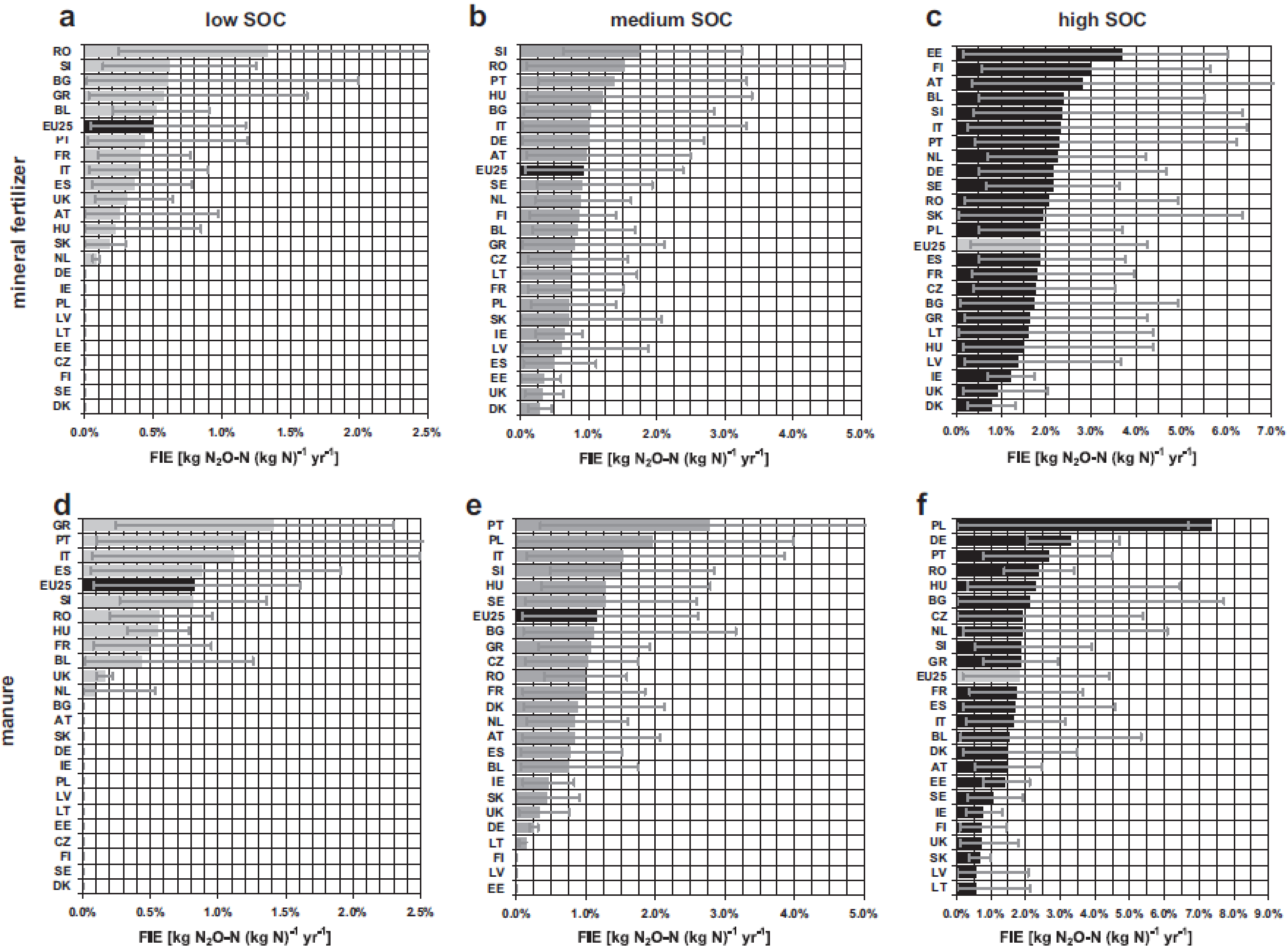


FIGURE 7 MEAN FIE OF N2O FOR 25 COUNTRIES IN THE EUROPEAN UNION FOR THE REFERENCE SCENARIO FOR THE APPLICATION OF MINERAL FERTILIZER (AEC) AND MANURE NITROGEN (DEF). THE PANELS SHOW THE FIE ON SOILS WITH (A,D) LOW SOC CONTENT; (B,F) MEDIUM SOC CONTENT AND (C,G) HIGH SOC CONTENT. THE DIAGRAMS ARE SORTED BY THE MEAN FIE SHOWN ON TOP OF THE FIGURE. THE ERROR BARS INDICATE THE 70%-CI.

It is interesting to note the differences between countries in Southern Europe vs. those of Northern Europe, with soils in Northern Europe generally exposed to more water, carbon content and lower temperature. Portugal, Slovenia, Romania and Italy have consistently high FIE on all soils and irrespective of the type of nitrogen. Countries like Denmark, Lithuania, Latvia and Ireland have low FIE throughout the considered cases. Notwithstanding, Denmark and Lithuania have N_2O fluxes that are with 2.9 and 5.1 kg N2O-N (kg N ha^{-1}) yr^{-1}, respectively, close or even above the European average indicating high back-ground fluxes.

The differences observed in simulated FIEs point to a complex interaction of weather conditions, their impact on the soil-vegetation continuum and hence their impact on N2O emissions. As noted, for all countries considered, FIE_{man} is on average about 10% higher than FIE_{min}, but for individual years the difference can be as large as 30%. This is the case of 1994, a year with the highest average mean temperature during the simulated years (11.4 °C) and average annual precipitation of 690mm. Yet in 1996, which experienced about the same mean annual precipitation as 1994 but had an average mean temperature of only 9.9 °C (the lowest in our dataset), FIE_{min} is larger than FIE_{man} by 23% (see Table 5). Microbial activities are reduced in cold temperatures, reducing also the occurrence of anaerobic micro- sites in dry soils because of the lower mineralization rate of manure. Thus, the relative rate of

N_2O formation from manure in dry soils with respect to N_2O formation from mineral fertilizer is lower in cold than in warm years, as mineral fertilizers are assumed to be less affected.

	FIE_{man}	FIE_{man}
1990	1.01%	1.09%
1991	1.02%	1.13%
1992	1.12%	1.38%
1993	1.20%	1.26%
1994	1.14%	1.50%
1995	1.03%	1.25%
1996	1.25%	1.01%
1997	1.21%	1.36%
1998	1.13%	1.24%
1999	1.31%	1.46%
2000	1.23%	1.15%
all years	1.15%	1.26%

TABLE 5 AVERAGE FIEMIN AND FIEMAN FOR 25 COUNTRIES IN EUROPE FOR 11 DIFFERENT METEOROLOGICAL YEARS.

Moreover, our results indicate that (a) much of the observed variability in N2O fluxes reflects the response of soils to external conditions, (b) it is likely that national inventories tend to overestimate the uncertainties in their estimated direct N_2O emissions from arable soils; (c) on average over Europe, the fertilizer-induced emissions (FIE) coincide with the IPCC factors, but they display large spatial variations. Therefore, at scales of individual countries or smaller, a stratified approach considering fertilizer type, soil characteristics and climatic parameters is preferable.

3.2. ESTIMATION OF N2O FLUXES AT THE REGIONAL SCALE: DATA, MODELS, CHALLENGES

Empirical and process-based models simulating N_2O fluxes from agricultural soils have the advantage that they can be applied at the scale at which mitigation measures can be designed and implemented. We compared bottom-up results from studies providing N_2O fluxes at a regional/country or continental scale with estimates from the process-based model DNDC-EUROPE and from the TM5-4DVAR inverse modeling system (Figure 8). While the agreement between different bottom-up models is generally satisfying, only in a few cases a thorough validation of the result was done. Complex empirical or process-based models do not appear to have a better agreement with inverse model results in estimating N_2O emissions from agricultural soils for countries or country-groups than simple ones. Both bottom-up and inverse models are limited by the density and quality of observations. Research needs to focus on developing tools that inherit the advantages of both methods.

The plot shows data for three individual countries (Germany, France, and Poland) and three country groups, that is, Belgium, Netherlands and Luxembourg (BENELUX), United Kingdom and Ireland (UKIRE), and Czech Republic, Slovakia and Hungary (CSH). The bottom- up models are IPCC [UNFCCC and EDGARv4.0, IPCC- approach using a factor for fertilizer-induced emissions from the Stehfest and Bouwman model (Sub-FIE-JRC), Stehfest and Bouwman as implemented by JRC (SuB- JRC), INTEGRATOR, FISE, DNDC- EUROPE [DNDC-EU], and IDEAg.

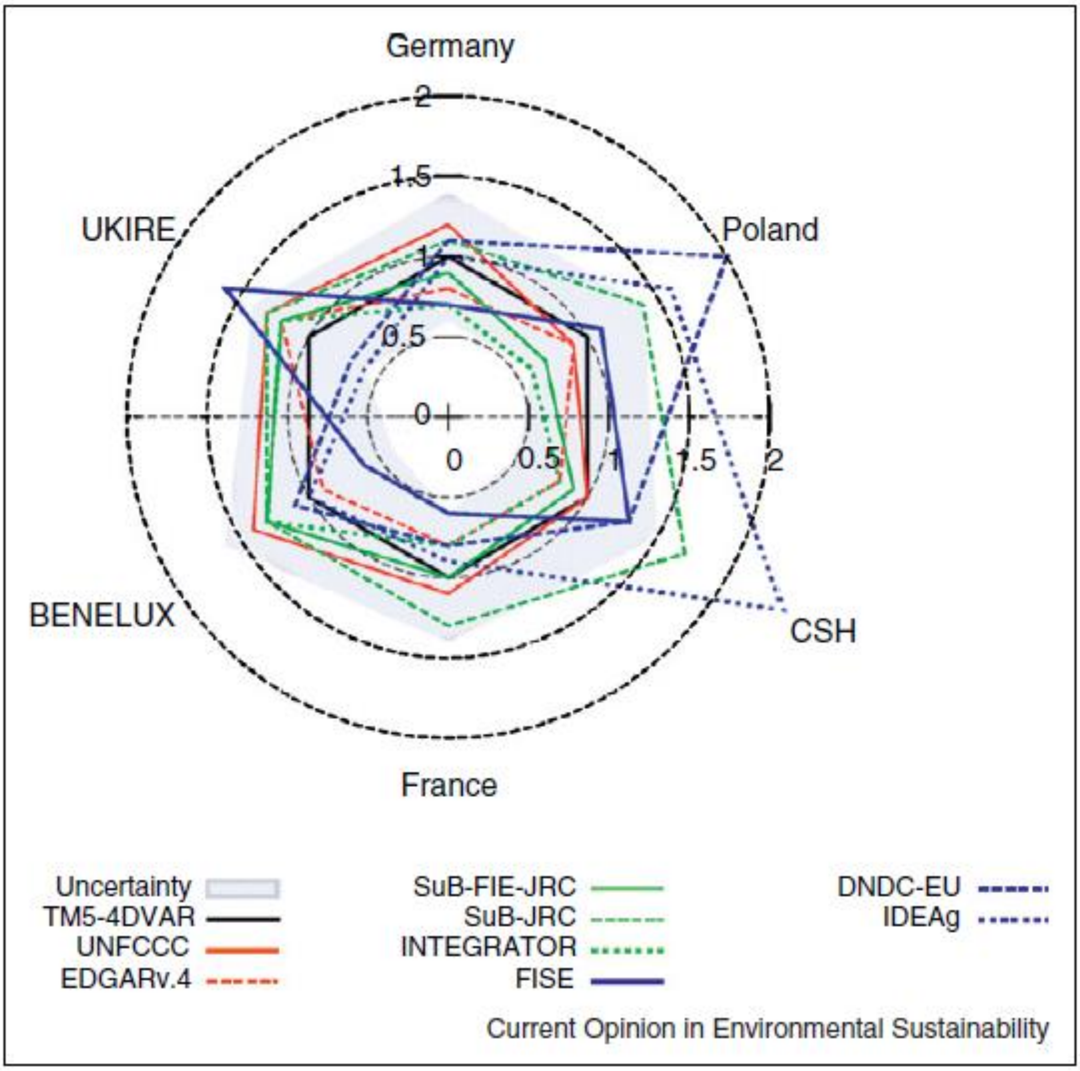


FIGURE 8 PLOT OF RELATIVE ESTIMATES OF DIRECT N2O FLUXES [GG N2O-N YEAR^{-1}] FROM AGRICULTURAL SOILS FROM EIGHT BOTTOM-UP MODELS AS COMPARED WITH DATA FROM THE TM5-MODEL IN INVERSE MODE

The paper presents also a review of available regional estimates of N_2O fluxes in Europe using models or IPCC methodologies. A comparison of these literature data with results from the DNDC-EUROPE model is shown in Figure 9. The plot shows data as flux rates [kg N2O-N ha^{-1} year^{-1}]. The shape of the point indicates whether default IPCC methodology (diamond) or another model (circle) has been used. Dark grey dots indicate that the model has been calibrated on regional data, while light grey dots indicate that no specific calibration for the study was undertaken. A black border around the dot indicates that some dedicated validation has been done.

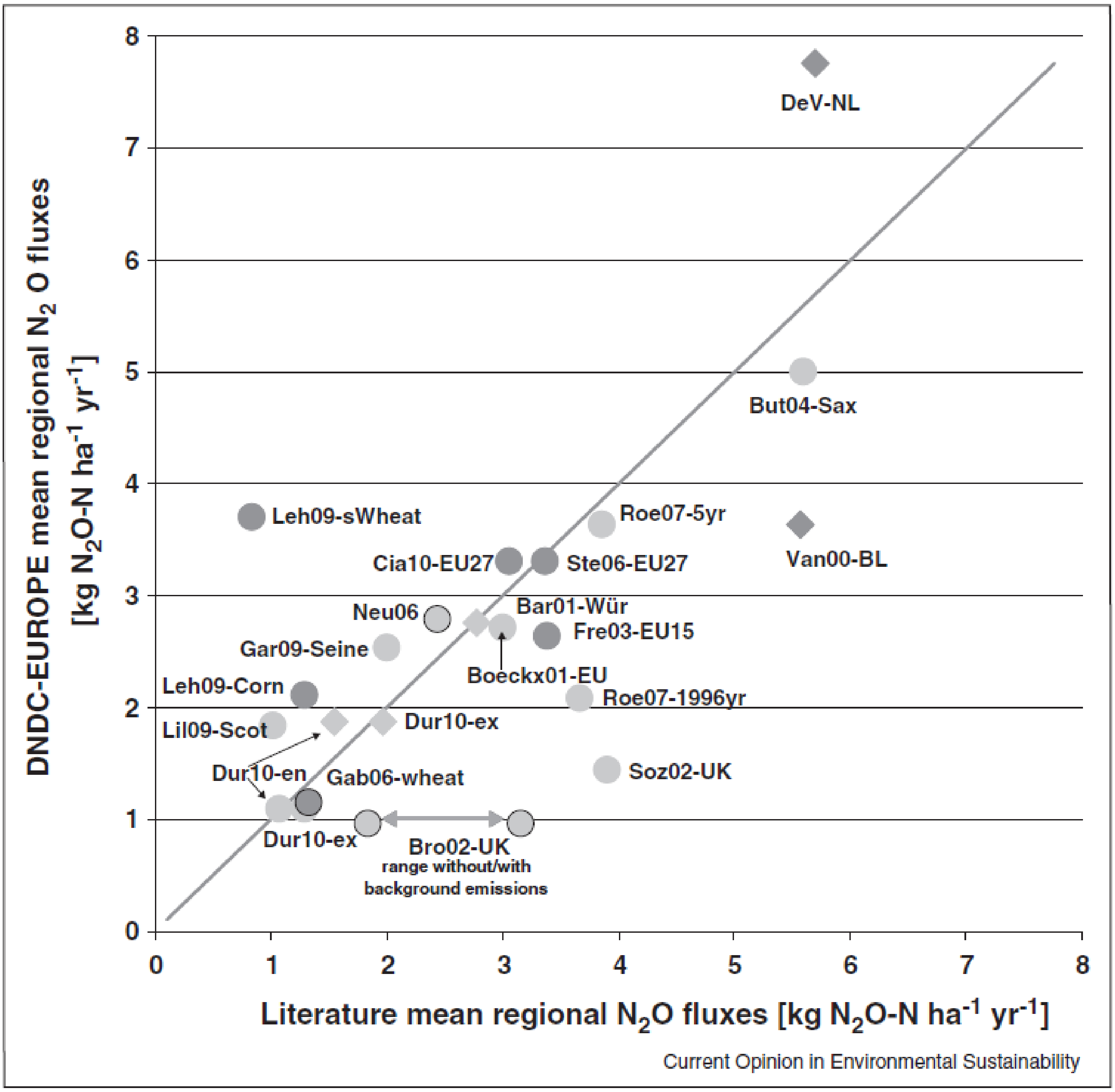


FIGURE 9 COMPARISON OF REGIONAL ESTIMATES OF N2O FLUXES VERSUS THE AVERAGE N2O FLUX RATES FOR MINERAL SOILS SIMULATED WITH THE DNDC-EUROPE MODEL

Despite the scatter in the data, there is a trend that DNDC-EUROPE tends to estimate lower N2O fluxes than the more specific studies, particularly for the United Kingdom and Belgium. Most of the studies use more detailed input data than are available for a EU-wide study. For example, Roelandt et al. provide data for the year 1996 and as average over five years. While simulations of DNDC-EUROPE match very well for the average value, it estimated much lower emissions using the meteo-data for 1996.

4. Coupling LCA with dynamic models to calculate overall life cycle emissions of rice production in Piedmont

4.1. INTRODUCTION

Rice is grown on more than 140 million hectares worldwide and is the most heavily consumed staple food on earth. Among the rice producing countries, Italy only ranks 29th, with a national production rate for paddy rice (Rough Rice: threshed unmilled rice) of 1.52 million tons, an average grain yield of 6.12 t/ha and a total harvested area of 247700 ha in the year 2010 (FAOSTAT, 2012).

Italian rice represents the most important European Union producer both in mass and cultivated area terms. In fact, rice cultivation is occurring in five EU-15 countries: France, Greece, Italy, Portugal, and Spain, but Italy accounts for 49% of total production and 51% of rice-harvested area (2010).

About one quarter of this total European surface is located in an area known as the "paddy area" of Piedmont (PPA), a wide, continuous and unconfined territory located in NW Italy, between 45°00' and 45°30' N and 8°10' and 8°30' E. (Zavattaro et al. 2008). This important agricultural district comprises 7 provinces (see Table 6) and occupies 174569 ha excluding the urban areas. The area is characterized by monocultures such as rice, maize and sorghum, but 68% of the area (118677 ha) is completely dedicated to rice cultivation (ISTAT, 2011).

Piemonte	Surface	%
Torino	211	0.18%
Vercelli	74490	62.77%
Novara	39788	33.53%
Cuneo	216	0.18%
Asti	0	0%
Alessandria	0	0%
Biella	3972	3.35%
Verbano-Cusio-Ossola	0	0%
Total in PPA	**118677**	

TABLE 6 RICE PRODUCTION IN PPA PROVINCES

Rice production generates wealth and jobs, but also creates high environmental impacts that some believe to be unacceptably high (Tilman et al. 2001; Zhang et al. 2006). Apart from soil and water pollution and consumption of energy and raw materials, paddy fields (irrigated or flooded land used for growing rice) are in fact claimed to heavily contribute to global warming phenomenon.

Rice systems are typically grown in flooded soils, with a usual depth of 15-25 cm and in a timeframe from April-May to August. In this particular soil condition, methane (CH_4) is the dominant greenhouse gas (GHG) emitted due to anaerobic decomposition of organic material. The annual amount emitted is largely controlled by water and residue management practices, soil type, temperature and the amount of fertilizers and other organic and inorganic amendments (Yagi et al. 1997; Wassmann et al. 2000; Dan et al. 2001; Harada et al. 2007). However, rice systems also emit N_2O fluxes, which are characterized by a very large spatial and temporal variability due to their strong dependence on environmental factors. The intensity of emissions is related to nitrogen additions, concentration of organic material in the soil, temperature and precipitation (Zou et al. 2007, 2009; Butterbach-Bahl et al. 2011). N_20 and CH_4 emissions follow opposite trends and are likely to be in a tradeoff: increasing submersion time triggers larger CH_4 emissions while interrupting it through extended drainage periods generates more N_2O while lowering CH_4 production (Cai 2003).

The Intergovernmental Panel on Climate Change (IPCC 1996) estimated CH_4 global emission rate from paddy fields at 60 Tg/yr, with a range of 20 to 100 Tg/yr. This is about 5-20 per cent of the total emission from all anthropogenic sources. At EU 27 level, rice accounts for just 0.6% of total emitted anthropogenic CH_4 (including land use, land use change and forestry), while, at Italian scale, it accounts for 4.2% of total (European Environment Agency 2012). On the contrary, N_2O emissions from rice are not specifically reported in EU GHG inventory, probably due to low N_2O emission rates (the majority of EU and Italian rice fields are currently flooded) and high uncertainties correlated. However, N_2O is a more potent GHG than CH_4 and CO2 with a radiative forcing potential that is approximately 12 and 296 times larger respectively (IPCC, 2001). Moreover, in the near future water resource conservation practices might potentially reduce or abandon submersion, thus contributing to an N_2O emission increase.

Various kinds of models have been developed to estimate the fluxes of pollutants caused by agriculture (e.g. CH_4 and N_2O), including empirical models of different complexity (IPCC 1996; Mosier et al. 1998; Bouwman et al. 2002), process-based models (Li et al. 1992, 2006) and meta models based on applications of detailed process-based models.

Empirical models are usually based on an emission factor approach, correlating GHG emissions to few controlling parameters. Model rationale is based on extrapolation of the results of plot scale experiments (measures of fluxes). They can usually consider differences in water regime and of types and amount of amendment applied only; they are hence not able to fully account for all determining parameters influencing emissions. The result of this simplification is that spatial and temporal differentiation of emissions according to specific local parameters is almost impossible, especially at small scales. Areas receiving the same amount and typology of fertilizer, in fact, can yield very different emissions according to soil, weather and other crop management parameters (e.g. fertilization timing, tillage typology and timing, etc.). More complicated empirical models can include more parameters and can partially reach some differentiation. However, their results will always be based on a set of physical measures. This is the strength (because they are referring to real measures), but also the weakness of this approaches. Fluxes measures are costly and require long timeframes to be significant; they can hardly be representative of all complex combinations between soils, climates and crop management techniques. Moreover this kind of models cannot be used when trying to assess the outcome of new scenarios that bring new, unprecedently tested, crop management practices.

Process based models use biological, chemical and physical equations to describe the complex set of phenomena that are happening in cultivated soils. This capability allows a full spatial and temporal differentiation of emitted fluxes that is limited only by the elevated data requirement.

Still, process based models cannot account for the whole impacts associable to rice production. Environmental sustainability, in fact, concerns not only climate change effect but also other environmental problems. For example, nitrates emissions are a considerable source of surface and ground water quality degradation (Moreau et al. 2012) greatly contributing to eutrophication effect.

Moreover the abovementioned pollutant fluxes represent what is commonly regarded as direct emissions from the field. In a life cycle perspective, to direct GHG emissions from cultivated soils we have to complement direct and indirect emissions and the depletion of natural resources associable to each process related to agricultural activity. For example, the chemical pollution derived from toxic compounds associated to pesticide use or from nitrate and phosphorus leaching losses, or indirect emissions from fertilizer production. In fact, previous studies demonstrated that fertilizer production emissions (indirect) contribute for 9.2% of total GHG emissions associated to rice production (Blengini and Busto 2009).

Adopting solution to mitigate impacts concentrating on GHG emissions only or disregarding the complex dependencies between the single processes in the agri-food chain and between agriculture and other sectors, brings the risk of just shifting impacts (temporally or geographically) from one productive phase to another (burden shifting) or just move them from one impact category to another (trade-offs) (Breiling et al. 2005; Laurent et al. 2012).

Life Cycle Assessment (LCA) methodology, aims at a comprehensive quantification of the environmental performances of products through a holistic approach. It considers potential impacts of all stages of productions and covers a broad range of impact categories, typically including climate change, stratospheric ozone depletion, acidification, photochemical ozone formation, aquatic and terrestrial eutrophication, impacts of toxic substances, land use impacts, water use, and depletion of both renewable and non-renewable resources.

The great potential of LCA, is the ability to integrate knowledge e from different sources/fields (e.g., nitrogen chemistry, pollutant fate modelling) in order to understand the complex interaction that characterize agricultural processes and provide outputs that can help decision-making processes.

4.2. Methodology

Two environmental models were coupled and their outputs fed to a LCA model in order to calculate overall environmental impacts of rice production in Piedmont "paddy area" in northwest Italy. Crop management, climate and soil data were gathered from literature sources concerning the area under study, referenced to homogeneous soil and crop management geographic units and used as input for the dynamic simulation models. The hence obtained results were then fed as probability distribution to the LCA model (see Figure 10).

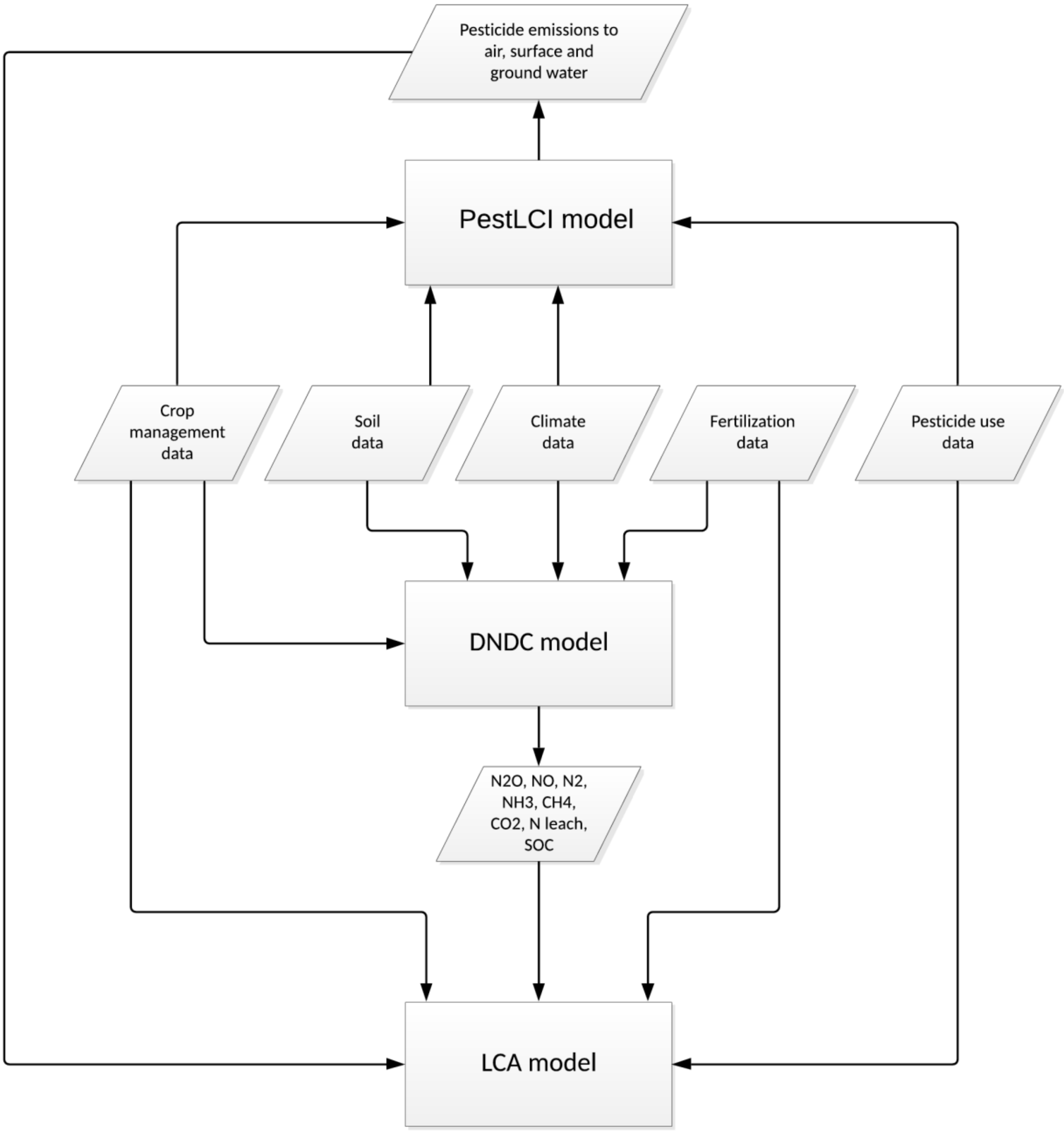


FIGURE 10 MODEL COUPLING SCHEME

To predicts direct emissions from the field two models were used:

- Denitrification–decomposition model (DNDC), a process-based model of carbon and nitrogen biogeochemistry in agro-ecosystems. First described in 1992 (Li et al. 1992), it can simultaneously model nitrate leaching, crop yield and agricultural trace gas emissions (N_2O, NO, N_2, NH_3, CH_4 and CO_2).
- PestLCI, a pesticide fate model capable of estimating how spread pesticides are distributed to environment compartments. It calculates the percentage of chemical active principle (contained in pesticide products) that is released to air, surface and groundwater waters as well as plant uptake and degradation. It is specifically designed to be used in life cycle inventories (LCI) of field applications.

4.3. DATA SOURCES

Crop management data covers information about fertilization management, residues management, pesticide use, tillage, seeding, harvesting, irrigation and other field operations conduced to prepare the field.

Fertilization and residue management data in the area was retrieved by agronomic literature (Zavattaro et al. 2008) that subdivided PPA into 67 spatial units with variable extension, characterized by homogeneous soil type, irrigation water source, agricultural system and farm type. For each unit a survey was performed encompassing a variable (according to unit size) number of farms.

The quantity and typology of pesticide used were not surveyed in the study by Zavattaro et al.. In fact, data representative for the whole rice cultivation area are very difficult to obtain. Moreover the extreme variability of pesticides market (due to product obsolescence, legislative restrictions and market costs) makes time representativeness a very complex issue. As a proof for this assumption, literature data found, regarding the area under study, was very scarce: Ferrero and Tabacchi (Ferrero and Tabacchi 2000) report average pesticide use for year 2000 between 0.4 and 13.2 kg of active principle per hectare. Since specific data on pesticide use for each spatial unit were not retrieved, an expert estimate (personal communication from prof. Aldo Ferrero of agronomy faculty of Turin) was used in order to obtain a representative picture of the amount and typology of pesticides used in the area.

Irrigation management and field operations data are also required by models to simulate the complex interaction between natural processes and human activities. Unfortunately no detailed regional database containing irrigation is available. In this study, data coming from direct measures on a particular study site were used. The field in object is situated in the Po Valley, in Northern Italy, in the municipality of Torre Beretti and Castellaro, (Meijide et al. 2011a).

Climate data was retrieved from ARPA (Italian acronym standing for Regional Agency for Environment Protection - Agenzia Regionale per la Protezione dell'Ambiente) Piedmont website (ARPA Piemonte 2012). Piedmont meteorological stations were geo-referenced and the nearest station to each spatial unit was calculated. ESRI ArcMap 10 software was used for the calculation.

Soil data used as model input comes from two main sources: Piedmont regional database and data recorded in the previously cited site (Meijide et al. 2011b).

In this study uncertainty was assessed according to the methodology proposed in Huijbregts et al. (Huijbregts et al. 2003). Parameter, scenario, and model uncertainty were estimated and the proposed iterative approach was followed: i.e. performing consecutive Monte Carlo simulations and Spearman Rank correlations to assess and improve the uncertainty of those parameters most influencing output uncertainty (Figure 11).

Parameter uncertainty is introduced by measurement errors, expert estimations and assumptions and reflects our incomplete knowledge about the true value of parameters.

Scenario uncertainty reflects results dependence over normative choices in the modeling procedure, for example about allocation procedure for multi-output and multi-waste processes, system boundaries or impact assessment.

4.4. UNCERTAINTY PROCEDURE

Model uncertainty is introduced by simplification of real processes introduced by LCA modeling structure, by models used in our study to provide input for LCA and by models used to estimate impact assessment characterization factors.

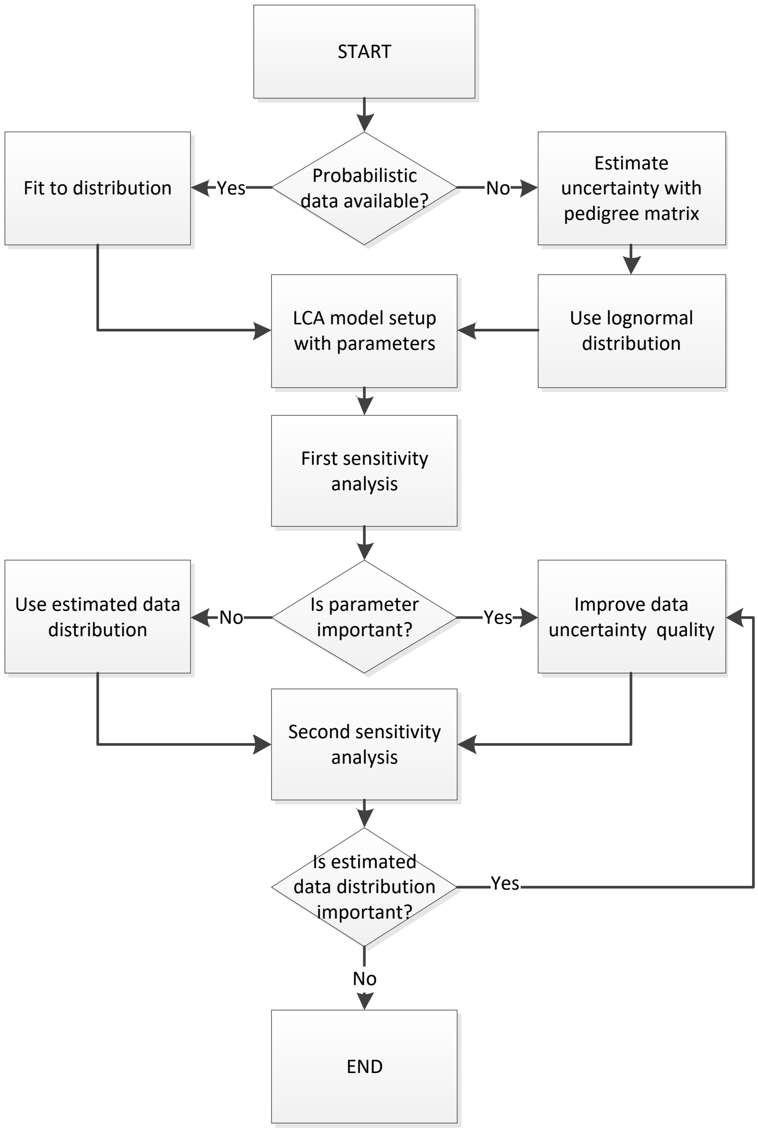


FIGURE 11 UNCERTAINTY PROCEDURE

Specific probability distribution for individual parameters is generally not known in LCA, because parameter values are mostly based on few measurements or on estimates. Choosing the same probability distribution for all parameters is therefore reasonable to avoid bias among parameters (Geisler 2003). In order to estimate uncertainty of parameters having deterministic values the Pedigree Matrix approach (Bo P. Weidema 1998; Ciroth 2009; Weidema and Beaufort 2001) was followed. Figure 11 shows a set of questions regarding data source and representativeness. To each question regarding reliability, completeness, temporal correlation, geographic correlation, further technological correlation and sample size, a quality level is associated from 1 to 5. To each level a score value is attributed, expressed as a contribution to the square of the geometric standard deviation (Table 8). Equation 1 calculates the geometric standard deviation (SD95) where U1 to U6 are six characteristic's contributions while U_b is basic uncertainty factor.

EQUATION 1 $$SD95 = exp^{\sqrt{[\ln(U1)]^2+[\ln(U2)]^2+[\ln(U3)]^2+[\ln(U4)]^2+[\ln(U5)]^2+[\ln(U6)]^2+[\ln(Ub)]^2}}$$

By answering to questions this approach allows estimating a probability function for the input value. The methodology proposes to use a lognormal distribution because it remains always positive and fits skewed and highly variable experimental data well (Mattila et al. 2011).

We hence assumed a lognormal using literature mean value and estimated geometric standard deviation (with pedigree matrix) to insert probability distributions for those parameters that had deterministic values only.

Indicator score	**1**	**2**	**3**	**4**	**5**
Reliability of source	Verified data based on measurements	Verified data partly based on assumptions or non-verified data based on measurements	Non-verified data partly based on assumptions	Qualified estimate (e.g. by industrial expert)	Non-qualified estimate or unknown origin
Completeness	Representative data from a sufficient sample of sites over an adequate period to even out normal fluctuations	Representative data from a smaller number of sites but for adequate periods	Representative data from an adequate number of sites but from shorter periods	Representative data but from a smaller number of sites and shorter periods or incomplete data from an adequate number of sites and periods	Representativeness unknown or incomplete data from a smaller number of sites and/or from shorter periods
Temporal differences	Less than 0.5 years of difference to year of study	Less than 2 years difference	Less than 4 years difference	Less than 8 years difference	Age of data unknown or more than 8 years of difference
Geographical differences	Data from area under study, same currency	Average data from larger area in which the area under study is included, same currency	Data from area with slightly similar cost conditions, same currency, or with similar cost conditions, and similar currency	Data from area with slightly similar cost conditions, different currency	Data from unknown area or area with very different cost conditions
Further technological differences	Data from enterprises, processes, and materials under study	Data from processes and materials under study from different enterprises, similar accounting systems	Data from processes and materials under study but from different technology, and/or different accounting systems	Data on related processes or materials but same technology	Data on related processes or materials but different technology

TABLE 7 PEDIGREE MATRIX

Indicator score		**1**	**2**	**3**	**4**	**5**
Reliability	U1	1	1.05	1.1	1.2	1.5
Completeness	U2	1	1.02	1.05	1.1	1.2
Temporal correlation	U3	1	1.03	1.1	1.2	1.5

Geographical correlation	U4	1	1.01	1.02	–	1.1
Further technological correlation	U5	1	–	1.2	1.5	2
Sample size	U6	1	1.02	1.05	1.1	1.2

TABLE 8 DEFAULT UNCERTAINTY FACTORS APPLIED IN CONJUNCTION WITH THE PEDIGREE MATRIX.

4.5. GOAL AND SCOPE DEFINITION

4.5.1. GOAL OF THE STUDY

The overall goal of the study is determining total life cycle environmental impacts of rice cultivated in Piedmont “paddy area” in Italy.

4.5.2. SCOPE OF THE STUDY

LCA reports impacts through the concept of Functional Unit (FU). The purpose of FU is to provide a reference unit to which the inventory data are normalized, results are reported and to provide a base for scenario comparisons. In agricultural products, commonly used functional units are area, mass of final products and energy or protein content in food products.

Reporting impacts per area is common in national emissions inventories which usually report direct field emissions only using simplified methodologies (IPCC tier 1 and 2). Reporting impacts in this way shows an immediate comparison of direct with overall (direct + indirect) impacts. However, this approach shows its limitations when assessing the impacts of alternative cultural systems. Scenarios with alternative crop management techniques, for example, present very different per area impacts but potentially also grain yields. In this way they provide different services (grain production quantity), making their comparison impossible.

A possible solution is reporting impacts as a function of mass of product (i.e. per kg of rice). Each cultivated hectare of land yields a certain amount of product to which a certain environmental impact can be associated. This approach can be useful when comparing different crop management techniques trying to identify which scenario presents lowest yield-scaled impacts (Linquist et al. 2012). However, it could be said that different rice qualities present different nutrient or energy contents.

Energy content based FU’s are often used, for example, when dealing with agricultural products used for energy production. Nutrient content based FU’s are more appropriate in this specific case given the fact that rice is cultivated as food. However two are the main reasons against this choice: first there are a very high number of rice qualities (cultivars) each with specific yield, nutrient content as well as crop management requirements (water, fertilization and weed management requirements). No separated data for the cultivation of different cultivar are available in PPA. Second, if it’s true that rice is produced as food, it is also true that different cultivars (with different nutrient contents) are chosen by consumers mainly for their organoleptic properties, whose preference is also highly subjective, and not for their nutrient value.

This said, the selected FU is 1 kg of produced rice regardless of rice cultivar distinction and of any quality parameters. The study adopts a cradle to retailer approach;

Nitrogen (N) Fertilizer input is a major driver of N2O emissions is often the limiting nutrient for crop production and hence a major driver of crop yield increases. Higher yields can often be obtained with greater N inputs, the question is whether the yield increase is large enough to offset the corresponding increase in N2O emissions and result in an overall lower yield-scaled impact. In rice systems, the relationship between fertilizer rate and impacts is potentially more complex, as CH_4 emissions are not as closely linked to N fertilizer inputs as N_2O emissions.

4.6. Model Data

This section describes input data used by the dynamic models. It is categorized in crop management, soil and climate properties.

4.6.1. Fertilization Data

Zavattaro et al. presented a well-documented analysis of fertilization management techniques practiced by farmers in PPA. The study represents a picture of the status in year 2000.

The agricultural land of PPA was divided into homogeneous units through an examination of soil type, irrigation water, agricultural system, and farm type. Sixty-seven land units were outlined, representative farms were selected and interviewed on their adopted fertilization management technique, the supply of nitrogen, phosphorus, and potassium (NPK), the fate of straw (buried, burned or removed), and the average yield.

Nitrogen fertilizers were divided into two main categories: Mineral and organic.

- Mineral fertilizers are inorganic substances, primarily salts, containing nutrients required by plants. Commercial products contain at least one of three primary nutrients: N, P or K.
- Organic fertilizers are derived from animal or vegetable matter. They can be provided in an unprocessed form (e.g. manure, slurry, peat, guano etc.) or processed (e.g. compost, bloodmeal, bone meal, etc.)

On average, a total of 123 kg ha^{-1} of nitrogen (N) was spread on the rice crop. However, the variability of this value was remarkable as shown in Figure 12.

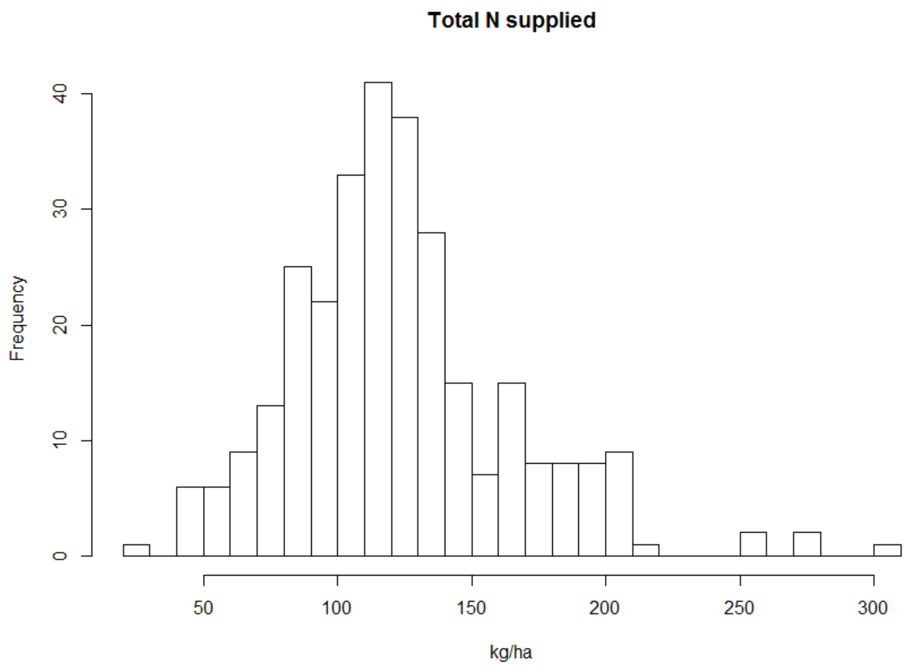


FIGURE 12 HISTOGRAM OF TOTAL N SUPPLIED IN PPA

Further analyzing N supply, it can be noted that mineral N is, by far, more applied than organic N in PPA. Mineral N average supplied value is108 kg ha^{-1}, while organic N is 15 kg ha^{-1} only (Figure 13).

On average, a total amount of 67 kg ha^{-1} of P_2O_5 was supplied to rice, with a wide variability over the surface. Potassium fertilizers, on average, supplied 161 kg ha^{-1} of K_2O.

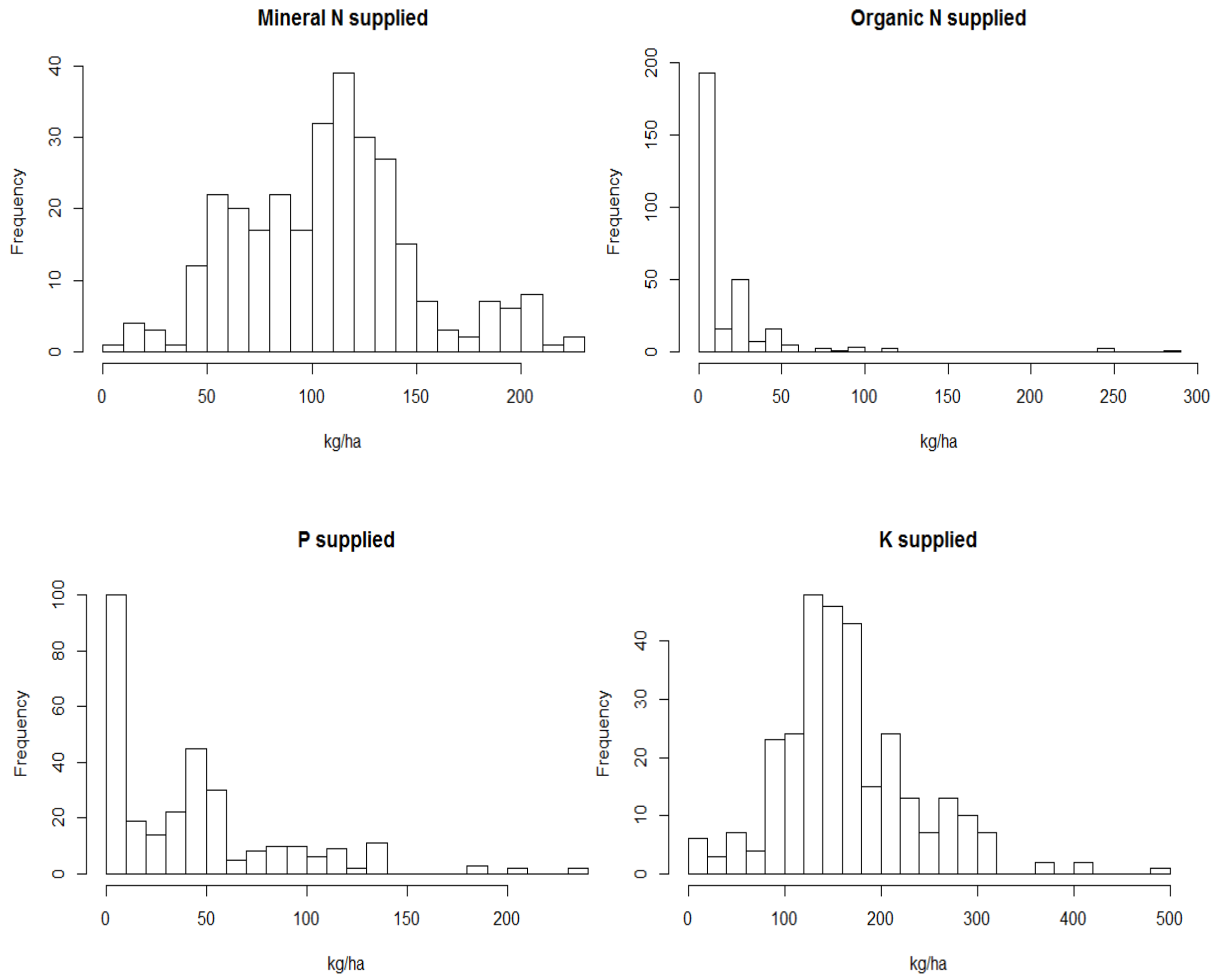


FIGURE 13 HISTOGRAM OF APPLIED N (MINERAL AND ORGANIC), P (AS P2O5) AND K (AS K2O).

Figure 14 and Figure 15 show a map of mineral and organic fertilization in the PPA.

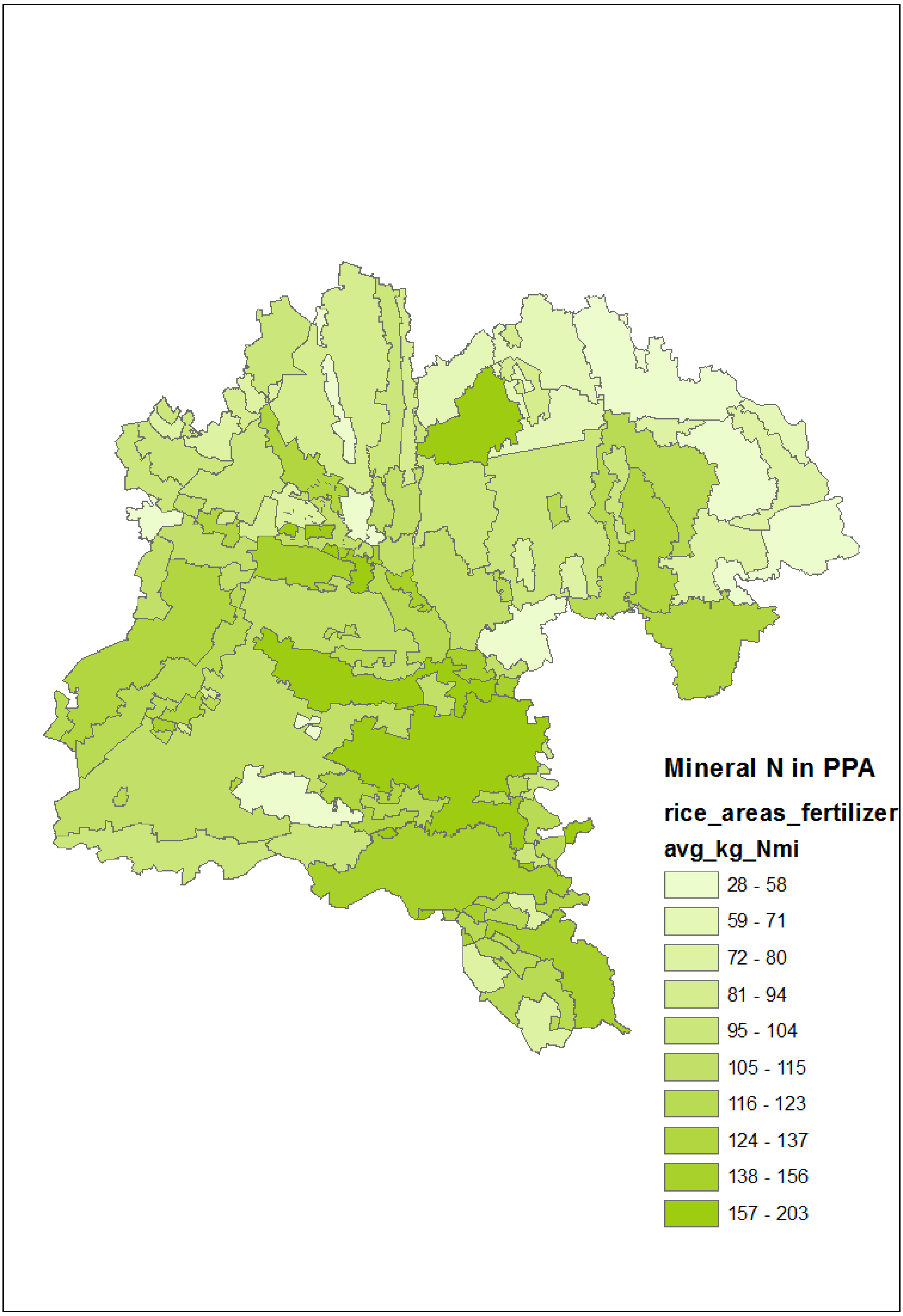


FIGURE 14 MAP OF MINERAL N USED IN PPA

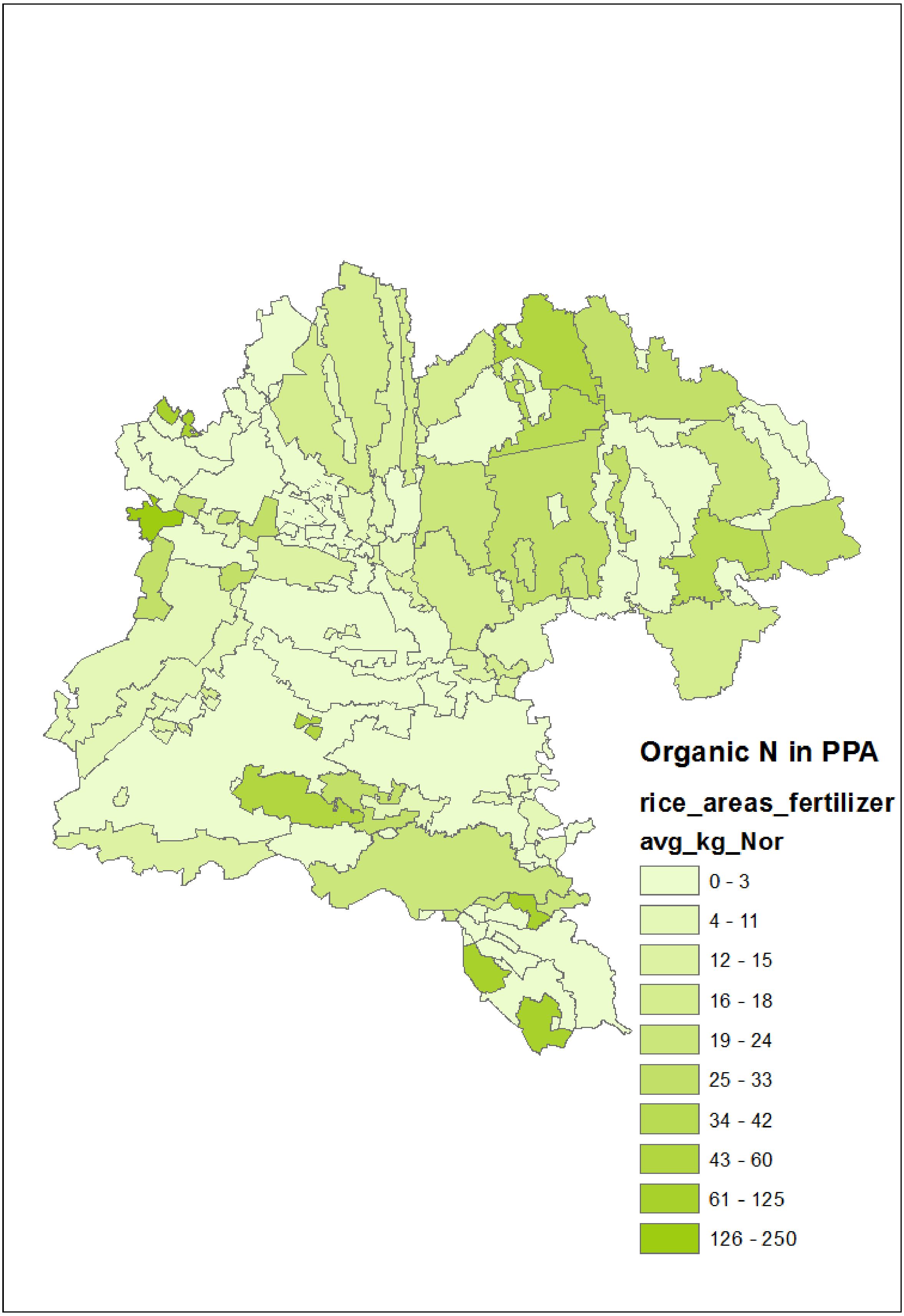


FIGURE 15 MAP OF ORGANIC N USED IN PPA

4.6.2. *Residues management*

Burning of crop residues on the field gives rise to emissions of various compounds, including aerosols and trace gases. Field burning of crop residues is forbidden in Europe. Most countries therefore do not report CH4 and N2O emissions from this source category. (European Environment Agency 2012).

In PPA, however, Italian government still allows residue biomass burning, even if it is more and more encouraging the adoption of more sustainable practice. Rice straw is either removed from the field, burned in situ or incorporated in the soil. Each of these measures has a different effect on overall nutrient balance, long-term soil fertility and environmental impacts.

Removal of straw removes most of the nutrients contained in straw and it's not a widespread practice in PPA, accounting for just 7% of the total farm surveyed in Zavattaro *et al.* (data for year 2000). Few commercial viable markets exist for rice straw, hence their low market demand and consequent economic value.

Incorporation of the remaining stubble and straw into the soil returns most of the nutrients and can help to conserve soil nutrient reserves in the long-term. For this reason and also to prevent emissions related to straw burning, this practice is currently strongly encouraged together with removal. However, farmers are concerned that rice straw incorporation is related to 1) crop effects that potentially reduce yield, 2) physical inability to consistently incorporate large acreage in a timely manner, and 3) added cost for the incorporation which does not generate additional income (yield). The result of this is that only 39% of farms adopt this practice.

The cheapest, fastest, and most widespread (54%) method of straw disposal is burning. Burning causes atmospheric pollution and results in nutrient loss, but it is a cost-effective method of straw disposal and helps reduce pest and disease populations that may occur due to reinfection from inoculum in the straw biomass.

Rice straw fate	% of PPA area	% of PPA farms
Incorporated	30%	39%
Removed	7%	7%
Burnt	63%	54%

Table 9 Rice straw management in PPA (year 2000)

4.6.3. *Pesticide use*

In the past, the European Union has mainly concentrated on pesticides' start and end-of-life phases, for example on the authorisation for placing pesticides on the market and control of their residues in food and foodstuffs. Eurostat has produced statistics on pesticides that are based on sales data for the main types of pesticides. This high level of aggregation makes it very difficult to estimate in which areas and over which crops pesticides are used.

Pesticide use in rice cultivation of PPA is not an exception to this. Detailed georeferenced data cannot be found in any literature source, published database or statistics.

Pesticide use data for PPA was then obtained by interviews with pesticide resellers conducted by prof. Aldo Ferrero of agronomy university of Turin. In Table 10, the list of commercial products is presented together with the contained active principles, dosage range and estimated applied surface.

Product name	Active principle	Max dose	Min dose	Dose	% Surface
Gulliver	Azimsulfuron	0.05	0.04	kg/ha	8%
Londax 60 DF: Koron WDG	Bensulfuron methyl	0.1	0.1	kg/ha	3%
Pull 52 DF: Sigma 52 DF	Bensulfuron+Metsulfuron	0.1	0.1	kg/ha	3%
Sunrice	Ethoxysulfuron	0.1	0.1	kg/ha	2%
Permit	Halosulfuron methyl	0.05	0.03	kg/ha	5%
Kocis	Imazosulfuron	0.8	0.7	l/ha	3%
Kelion WG	Orthosulfamuron	0.12	0.1	kg/ha	5%
Cadou WG	Flufenacet	0.7	0.7	kg/ha	8%
Beam	Tricyclazole	0.6	0.3	kg/ha	45%
various (Roundup, Hopper blue)	Glyphosate	12	0.8	l/ha	20%
Nominee	Bispyribac-Na	0.075	0.06	l/ha	8%
Command 36 CS	Clomazone	1	0.5	l/ha	5%
Stratos Ultra	Cycloxydim	4	4	l/ha	8%
Clincher	Cyhalofop butyl	1.5	1	l/ha	20%
Beyond	Imazamox	1.75	1.75	l/ha	20%
Ronstar FL	Oxadiazon	1.3	0.65	l/ha	70%
Aura	Profoxydim	0.8	0.4	l/ha	20%
Stomp Aqua: Most Micro	Pendimethalin	2.5	2	l/ha	20%
Tripion Ee: U46M class	MCPA	2	0.75	l/ha	30%
Agil	Propaquizafop	1	0.8	l/ha	2%
Viper	Penoxsulam	2	2	l/ha	25%
Garlon	Triclopyr	1.5	1	l/ha	7%
Karate Zeon	Lambda-cialotrina	0.125	0.175	l/ha	2%

TABLE 10 DOSES AND RATES OF DISTRIBUTION OF PESTICIDE IN PPA IN 2012

4.6.4. Field operations

Field operations include the following :

1. Maintenance: cleaning and maintenance of watering canals with mechanical tools, remaking of land embankments;
2. Ploughing: preparation of the earth breaking and turning over earth with a plow;
3. Spreading of organic fertilizers;
4. Land leveling: creating a slight surface gradient to facilitate the uniform distribution of irrigation water and to help water retention;
5. Fertilizer application (first): mainly with KCl;
6. Harrowing: to break and puddle clods of soil and incorporate organic materials into the soil;
7. Flooding of rice chambers;
8. Levelling of the clods outside chamber rice;
9. Pesticide spraying (Pre-planting);
10. Seeding: rice seed is sown and sprouted directly into the field;
11. Fertilizer application (second): mainly using N and P fertilizers;
12. Pesticide spraying (Post-emergence);
13. Harvesting; is the process of collecting the mature rice crop from the field. Harvesting consists of cutting, threshing, cleaning, hauling and bagging.

Field operation timing and typology can potentially influence the timing and amount of direct field emissions calculated by DNDC model as well as the fate of pesticides calculated with PestLCI. However differentiated input data concerning these agricultural aspects was not found in any literature source. Data used by models was hence retrieved from a single case study concerned with an agricultural field located in the Po Valley, in Northern Italy, in the municipality of Torre Beretti and Castellaro (Pavia) (Meijide et al. 2011b).

In Table 11 input data required by models is listed.

Operation	Date	
Harvest date	22/09/2010	
Planting date	30/05/2010	
Flooding date	14/04/2010 – 16/07/2010	
Flooding type	Continuous flooding	
Tillage dates	15/4/2010	28/5/2010
Tillage types	30 cm deep	5 cm deep

TABLE 11 FIELD OPERATIONS

4.6.5. *SOIL DATA*

Soil properties are required by all models adopted. Fortunately, soil maps of the whole Piedmont region are obtainable online. Maps are available at two scales: 1:250000 (low-res) and 1:50000 (hi-res).

Information provided in low-res map encompass soil destination category (describing for which use soil typology is more suitable), soil organic matter content and other soil parameters. The whole piedmont area is divided in 1938 polygons. Hi-res map contains information about soil texture classification (according to USDA), pH, rock content, etc. It contains 5223 polygons but does not cover the whole Piedmont (Figure 16).

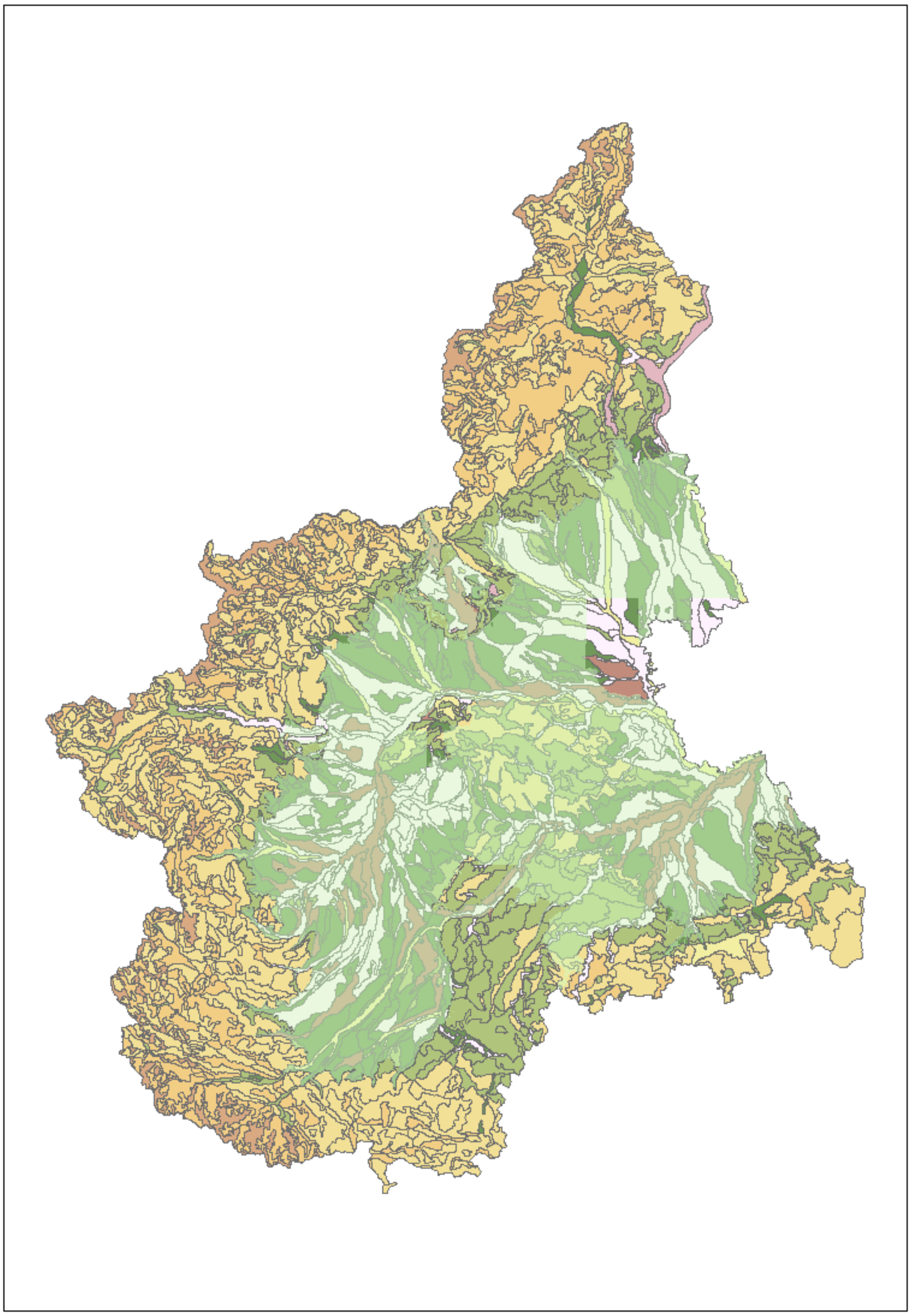


FIGURE 16 PIEDMONT SOIL MAP

Both maps were intersected with the map of Piedmont Paddy Area (PPA) as can be seen in Figure 17Figure 16. Since the hi-res map is currently covering only 80% of total PPA (159425 over 199193 ha) and since dynamic models require soil information input provided in hi-res map only, the remaining 20% area has been left out of calculations.

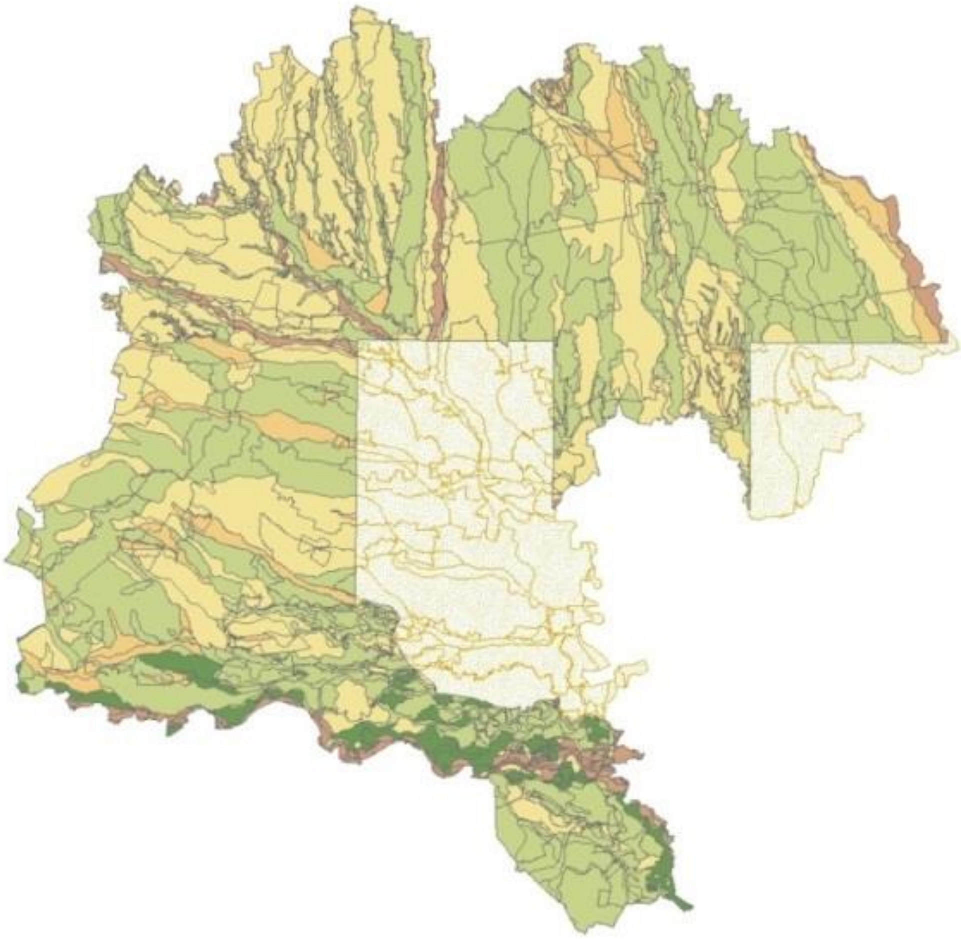


FIGURE 17 PIEDMONT PADDY AREA SOIL INFORMATION MAP

DNDC model requires the following soil information:

- Longitude
- Latitude
- N-deposition
- Soil organic carbon max and min value (SOCmax, SOCmin)
- Clay content max and min (Claymax, Claymin)
- pH (pHmax, pHmin)
- Bulk density (Densmax, Densmin)
- Slope

Bulk density was not given in soil maps and was hence calculated using the equations of Saxton (Saxton et al. 1986) using USDA Texture Triangle values as input.

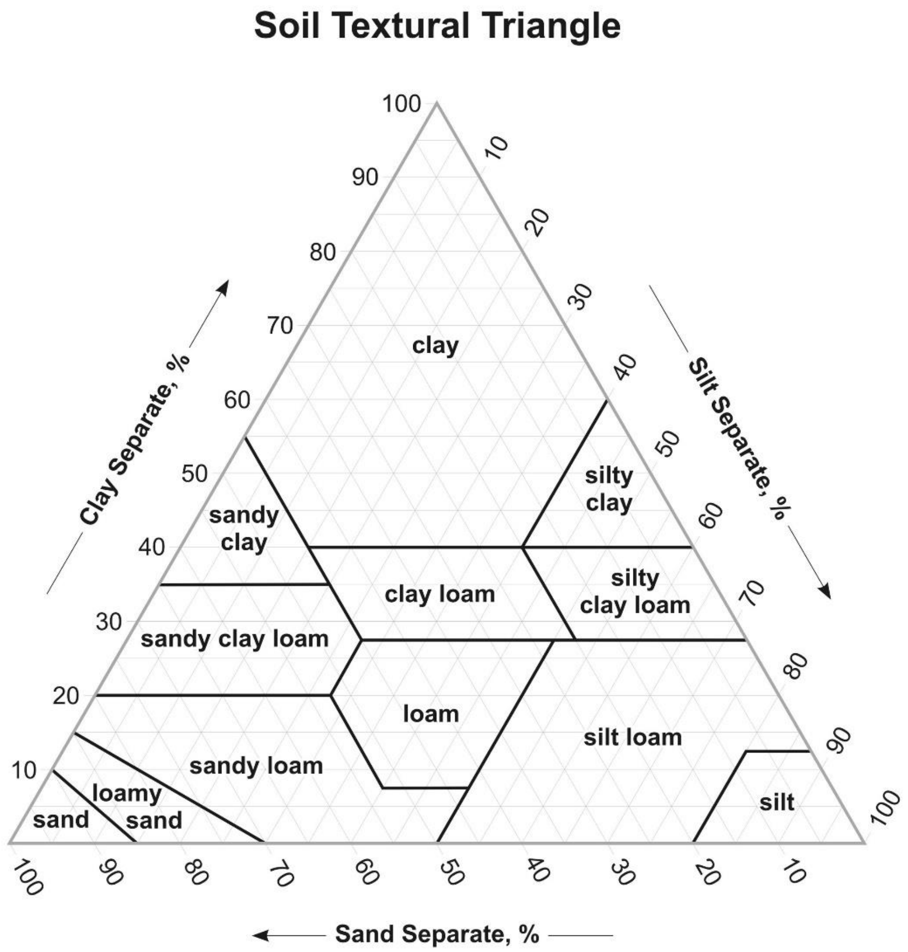


FIGURE 18 SOIL TEXTURAL TRIANGLE (USDA)

SOC values where given in low-res map but not in hi-res. The values where hence downscaled.

Clay content was obtained from texture class as maximum and minimum values allowed in the respective class.

pH value was obtained by hi-res map.

Slope was set as 0 (Meijide et al. 2011b).

For N-deposition, a value obtained by a particular field was used (Meijide et al. 2011b).

Parameters	Value	Source
Longitude, latitude	-	Piedmont soil map
N-deposition	2 mg N/l	Meijide et al. 2011
SOC_at_Surface	2.61÷0.63 kgC/kg soil	Piedmont soil map
Clay_fraction	55÷0.1%	Calculated (Saxton et al. 1986)
Soil_pH	8.5÷0.1	Calculated (Saxton et al. 1986)
Density	1.65÷1.4 g/cm3	Calculated (Saxton et al. 1986)
Slope	0	Meijide et al.

TABLE 12 SOIL PARAMETERS AND THEIR SOURCES

4.6.6. CLIMATE DATA

Piedmont is covered by a network of meteorological stations managed by ARPA (Figure 19). Meteorological data concerning rainfall, maximum and minimum temperature, wind speed, solar radiation and snow covering is recorded on a daily basis. Data is accessible in an online database (www. webgis.arpa.piemonte.it).

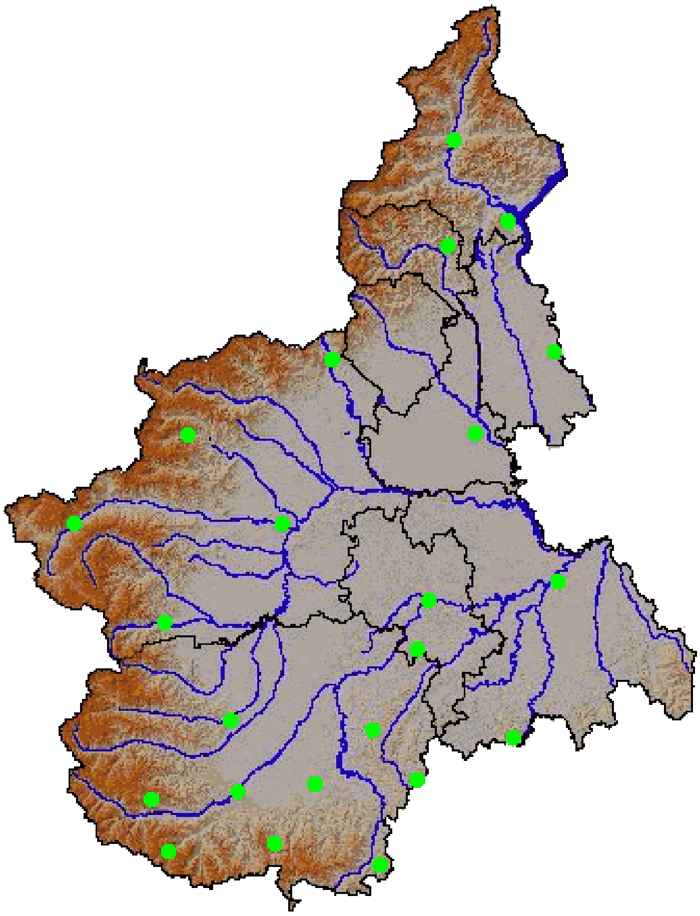


FIGURE 19 ACTIVE METEREOLOGICAL STATIONS IN PIEDMONT (SOURCE: WWW. WEBGIS.ARPA.PIEMONTE.IT)

In order to associate climate data to each map polygon and hence calculation unit, centroids (geometric center of the object's shape) of every polygon were first calculated. Then distances from centroids to each of the meteo station were calculated and the shortest ones selected.

Table 13 shows the meteorological stations that are located in the PPA region or in the immediate surroundings, their geographical coordinates and the available recorded years. Since some years were incomplete, probably due to instrument failures or maintenance, a gap filling procedure was adopted. When major percentages (>15%) of daily data was missing, the full year was skipped. Otherwise, for temperature, missing daily values were covered by the last available data before the gap and, for rainfall, gaps were covered by 0 values (no rainfall).

id	long	lat	Station_name	Province	Start year	last year
3	8.2322	45.262	ALBANO VERCELLESE	VC	1988	2012
19	8.2326	45.1932	VERCELLI	VC	1993	2012
21	8.4141	45.3256	CAMERI	NO	1988	2012
22	8.4809	45.2455	CERANO	NO	2002	2012

23	8.3301	45.3409	MOMO AGOGNA	NO	2005	2012
26	8.3803	45.2632	NOVARA	NO	2005	2012
34	8.1015	45.2826	MASSAZZA	BI	1993	2012
41	8.3019	45.0759	CASALE MONFERRATO	AL	1988	2012

TABLE 13 NEAREST METEREOLOGICAL STATION FROM PPA

4.7. LIFE CYCLE INVENTORY ANALYSIS (LCI)

4.7.1.1. DIRECT FIELD EMISSIONS

To calculate direct field emissions, 2877 DNDC model runs were setup covering 159248 ha, 75% of PPA total extension. Soil properties, weather conditions and crop management practices were derived from data presented in paragraph 4.6. DNDC For each homogeneous unit model calculated CO2, CH4, N2O, NO, N2, NH3 and the amount of leached Nitrogen (Leached N). Table 14 presents a summary of DNDC results.

	CO2	**CH4**	**N2O**	**NO**	**N2**	**NH3**	**LeachN**
	[kg C/ha]	**[kg C/ha]**	**[kg N/ha]**	**[kg N/ha]**	**[kg N/ha]**	**[kg N/ha]**	**[kg N/ha]**
Minimum	377	23.62	0.0040	0.000000	0.089	1.138	0.440
1st Quartile	650	72.90	0.1670	0.001000	0.616	2.647	1.250
Median	891	112.51	0.4140	0.001000	0.762	4.217	3.140
Mean	887.4	109.67	0.4594	0.001557	1.004	4.268	3.105
3rd Quartile	1066	140.23	0.7490	0.002000	1.584	5.279	4.230
Maximum	1882	240.76	1.8520	0.006000	1.812	11.553	9.890

TABLE 14 DNDC SUMMARY OF DNDC RESULTS

In order to represent an average plot situated in PPA calculated values were inserted into the LCA model as a probability distribution. Before fitting a distribution to a data set, it is generally necessary to choose good candidates among a predefined set of distributions.

The CO2 emissions values were taken as an example of the fitting procedure. First a visual test was performed plotting the empirical distribution function and the histogram (or density plot). Figure 20 shows at the left side the histogram on a density scale and at the right side the empirical cumulative distribution function (CDF).

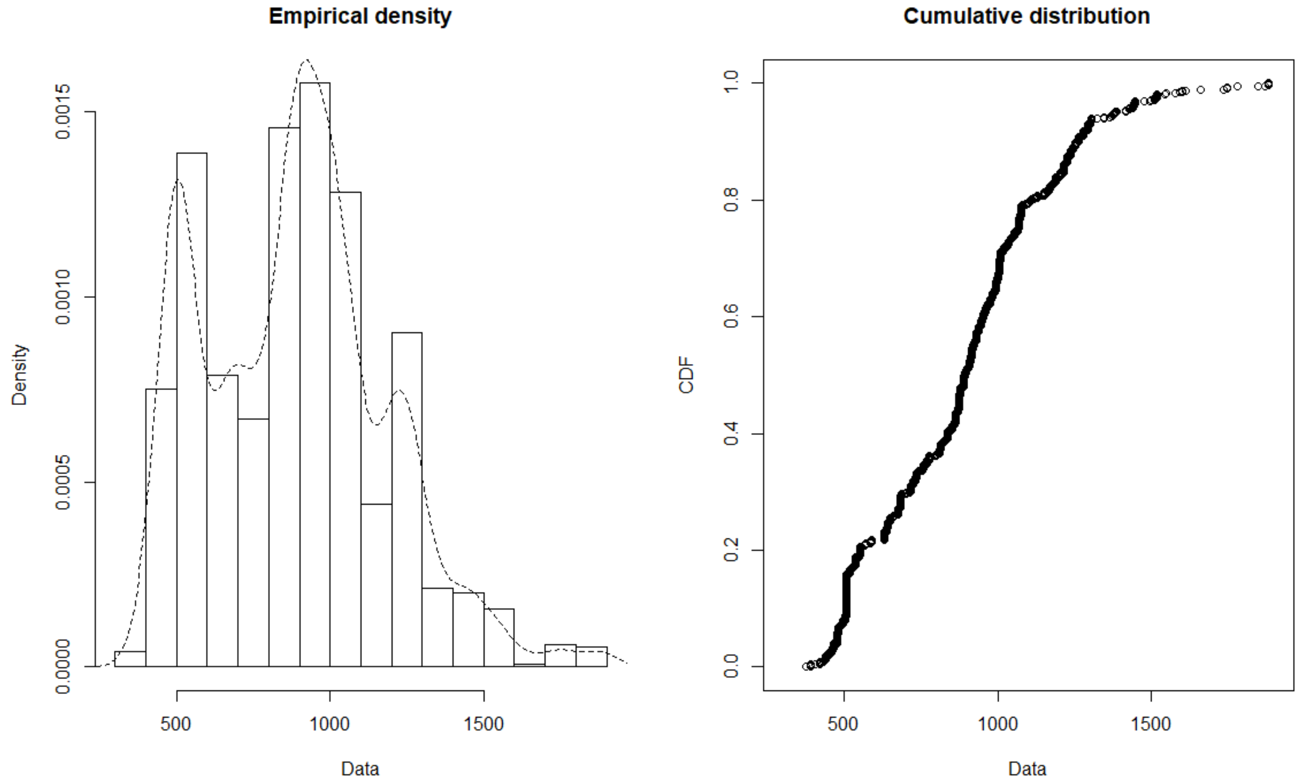


FIGURE 20 HISTOGRAM AND CDF PLOTS OF CO2 EMISSIONS

In addition to empirical plots, descriptive statistics such as skewness and kurtosis may help to choose candidates to describe a distribution among a set of parametric distributions. A non-zero skewness describes a lack of symmetry of the empirical distribution, while the kurtosis quantifies the weight of tails in comparison to the normal distribution. For CO2 emissions, summary statistics estimated standard deviation of 291.837, skewness of 0.4443286 and kurtosis of 3.033699. Skewness and Kurtosis were plotted in Figure 21 for a visual selection of most likely distributions.

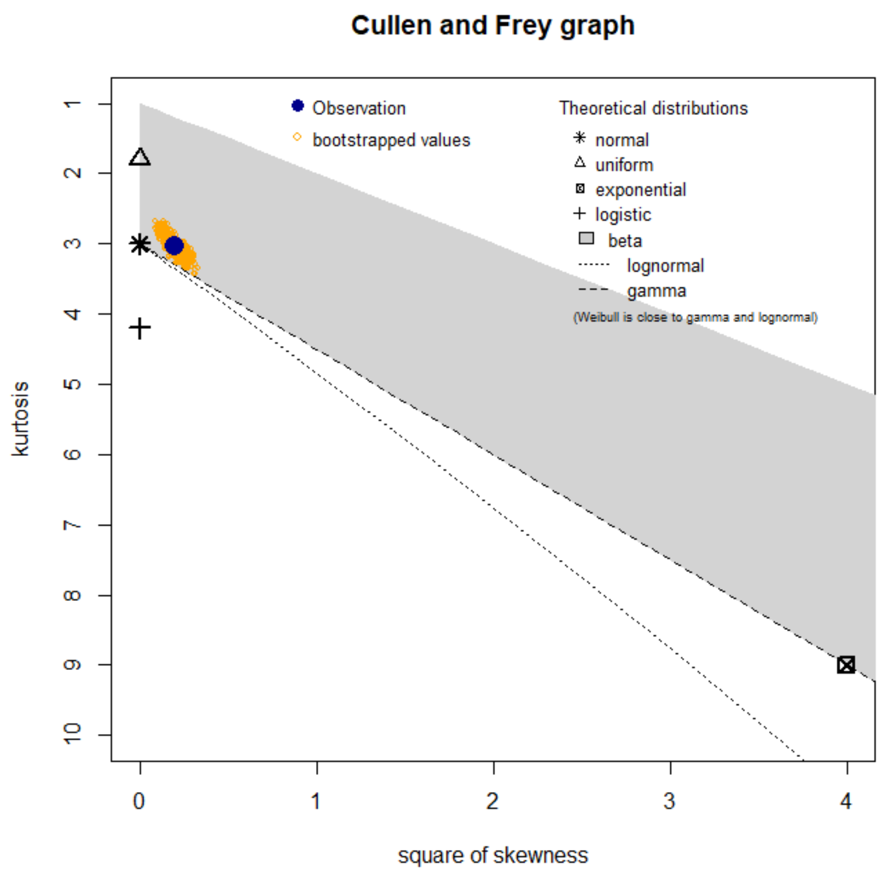


FIGURE 21 SKEWNESS-KURTOSIS PLOT FOR CO2 EMISSIONS

Gamma, lognormal and normal distributions were fitted with CO2 emissions dataset and four classical goodness-of-fit plots were presented in Figure 22:

- a density plot
- a CDF plot
- a Q-Q plot representing the empirical quantiles (y-axis) against the theoretical quantiles (x-axis)
- a P-P plot representing the empirical distribution function evaluated at each data point (y-axis) against the fitted distribution function (x-axis).

In order to further compare fitted distributions, three goodness-of fit statistics were considered: Cramer-von Mises, Kolmogorov-Smirnov and Anderson-Darling statistics.

Goodness-of-fit statistics	**norm**	**gamma**	**lnorm**
Kolmogorov-Smirnov	0.08052628	0.09194735	0.1121684
Cramer-von Mises	2.12697575	3.77952585	5.8389716
Anderson-Darling	18.41025474	24.95736632	35.4700223
Goodness-of-fit criteria			
Akaike's Information Criterion	33279.18	33154.90	33197.33
Bayesian Information Criterion	33290.70	33166.42	33208.85

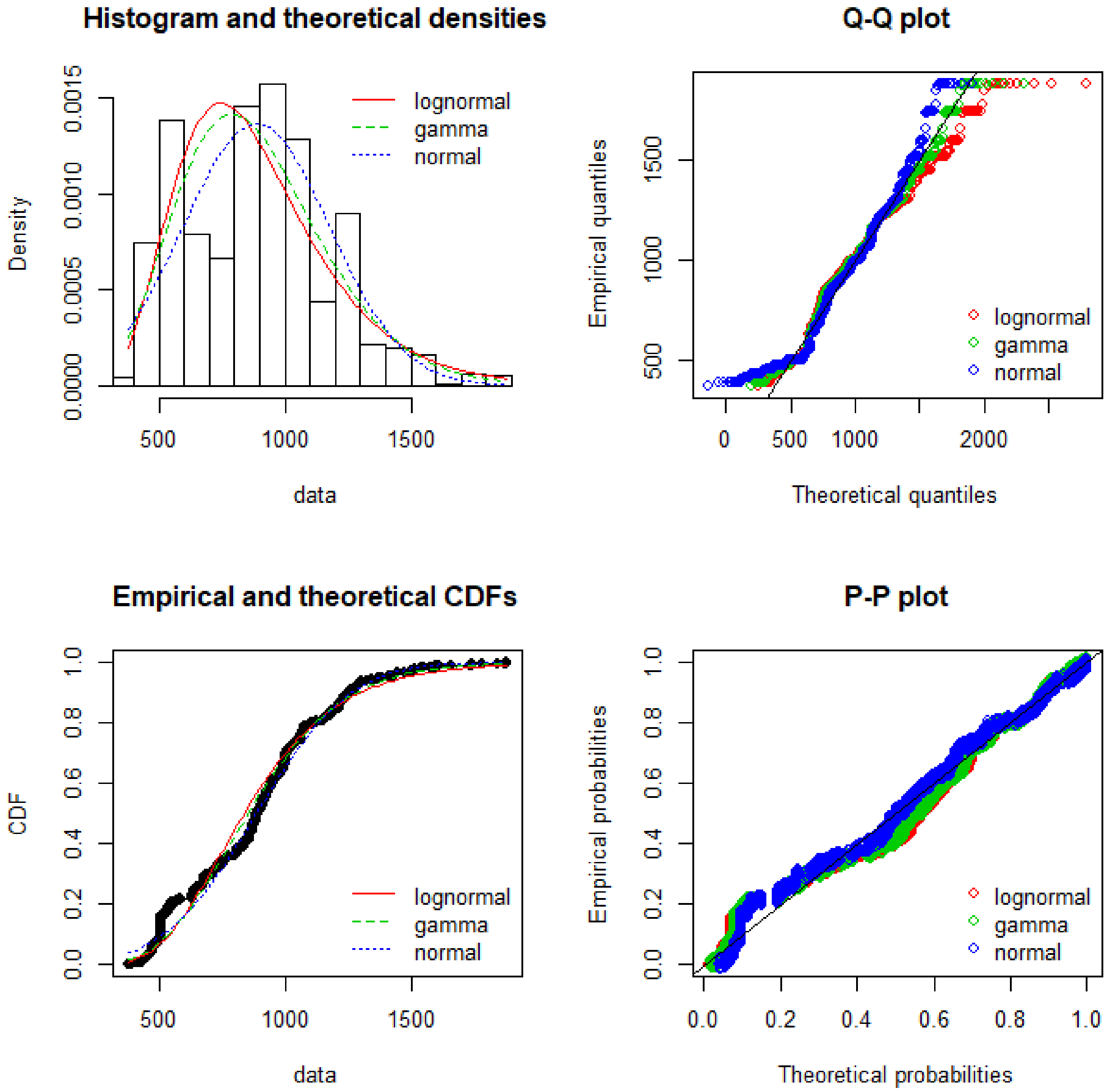


FIGURE 22 GOODNESS-OF-FIT PLOTS FOR VARIOUS DISTRIBUTIONS FITTED TO CO2 EMISSIONS (GAMMA, NORMAL AND LOGNORMAL)

Even if normal distribution appears to be the best fit, the lognormal was chosen because it has the advantage of not being defined in the negative domain, so credits do not accidentally happen during Monte Carlo simulation.

Mean value and squared geometric standard deviation (GSD^2) of the chosen distribution were calculated. Such values were then inserted into the LCA model.

	CO2	CH4	N2O	NO	N2	NH3	Leached N
Mean	683.96	302.41	1.82	0.05	7.77	0.54	7.92
GSD^2	2.68	4.82	5.68	4.00	1.71	42.26	6.15

TABLE 15 DNDC MODEL RESULTS

4.7.2. PESTICIDES

Pesticides production were modeled trying to find a match between the products used in PPA (see Table 10) and pesticide production processes in the Ecoinvent database. In some cases (Glyphosate, MCPA and Pendimethalin) an **exact match** was found. In terms of share of the full amount of pesticides used in PPA, exact matches represent 49%. When an exact match with the pesticide name was not found the substance group of the active principle was used as proxy and a general substance-group specific Ecoinvent process representative of the chemical **compound** was used. This second choice covers 20% of pesticides used in PPA. In case none of the previous choices was possible, a **general** pesticide production process was chosen covering 32% of pesticides used.

Active principle	Substance group	Pesticide type	Ecoinvent record	Note	in PestLCI database
Azimsulfuron	Sulfonylurea	Herbicide	[sulfonyl]urea-compounds	Compound	NO
Bensulfuron methyl	Sulfonylurea	Herbicide	[sulfonyl]urea-compounds	Compound	NO
Bensulfuron + Metsulfuron	Sulfonylurea	Herbicide	[sulfonyl]urea-compounds	Compound	NO
Ethoxysulfuron	Sulfonylurea	Herbicide	[sulfonyl]urea-compounds	Compound	NO
Halosulfuron methyl	Pyrazole	Herbicide	[sulfonyl]urea-compounds	Compound	NO
Imazosulfuron	Sulfonylurea	Herbicide	[sulfonyl]urea-compounds	Compound	NO
Orthosulfamuron	Pyrimidinylsulfonylurea	Herbicide	[sulfonyl]urea-compounds	Compound	NO
Flufenacet	Oxyacetamide	Herbicide	acetamide-anillide-compounds	Proxy	NO
Tricyclazole	Triazolobenzothiazole	Fungicide	diazole-compounds	Proxy	NO
Glyphosate	Phosphonoglycine	Herbicide	glyphosate	Exact match	YES
Bispyribac-Na	Pyrimidinyl carboxy compound	Herbicide	herbicides	General	NO
Clomazone	Isoxazolidinone	Herbicide	herbicides	General	YES
Cycloxydim	Cyclohexanedione	Herbicide	herbicides	General	NO
Cyhalofop butyl	Cyclohexanedione	Herbicide	herbicides	General	NO
Imazamox	Imidazolinone	Herbicide	herbicides	General	NO
Oxadiazon	Oxadiazole	Herbicide	herbicides	General	NO
Profoxydim	Cyclohexanedione	Herbicide	herbicides	General	NO
Pendimethalin	Dinitroaniline	Herbicide	pendimethalin	Exact match	YES
MCPA	Aryloxyalkanoic acid	Herbicide-metabolite	pesticides MCPA	Exact match	YES
Propaquizafop	Aryloxyphenoxypropionate	Herbicide	phenoxy-compounds	Proxy	YES
Penoxsulam	Triazopyrimidine	Herbicide	pyridine-compounds	Proxy	NO
Triclopyr	Pyridine	Herbicide	pyridine-compounds	Compound	NO
Lambda-cialotrina	Pyrethroid	Insecticide	pyridine-compounds	Proxy	NO

TABLE 16 PESTICIDE USED IN PPA AND THE ECOINVENT PROCESS CHOSED TO MODEL THEIR PRODUCTION

Ecoinvent process entry	Average use [g ha-1]
[Sulfonyl]urea-compounds, at regional storehouse	53,53
Acetamide-anillide-compounds, at regional storehouse	56,00
Diazole-compounds, at regional storehouse	202,50
Glyphosate, at regional storehouse	2188,80
Herbicides, at regional storehouse	2016,46
Pesticides MCPA, at regional storehouse	412,50
Phenoxy-compounds, at regional storehouse	17,28
Pyridine-compounds, at regional storehouse	922,74
Pendimethalin, at regional storage	526,50

TABLE 17 PESTICIDE PRODUCTION IN THE LCA MODEL

In order to model pesticide fate the same data was fed into PestLCI model. Given the lack of site-specific pesticide use data, the full PPA was modelled as a single plot where the full amount of pesticide was released. However, it was not possible to model the fate of all the active principles due to limitations in the model database; only 5 active principles (Glyphpsate, clomazone, Pendimethalin, MCPA and Propaquizafop) were available in PestLCI database accounting for 50% of the amount of pesticide used in PPA. For each product in database, a model run was set to calculate pesticide fate. The model calculated the degradation and uptake share (%) and dose (in g/ha) as well as emissions to air, surface water and groundwater. Results are shown in Table 18.

Soil and climate data were inserted as average PPA values.

	Degradation and uptake		Emissions to air		Emissions to surface water		Emissions to ground water	
	[%]	[g ha-1]	[%]	[g ha-1]	[%]	[g ha-1]	[%]	[g ha-1]
Glyphosate	88.08	1927.90	0.36	7.80	8.58	187.72	2.99	65.38
Clomazone	69.89	1409.25	10.71	215.93	14.83	298.97	4.58	92.30
MCPA	97.72	403.09	0.82	3.36	0.72	2.98	0.74	3.07
Propaquizafop	99.86	17.26	0.14	0.02	0.00	0.00	0.00	0.00
Pendimethalin	88.43	465.59	0.57	3.00	8.70	45.83	2.29	12.08

TABLE 18 PESTICIDE FATE CALCULATED WITH PESTLCI MODEL.

4.7.3. Fertilization and yield

Area -specific mineral and organic fertilization data (see paragraph 4.6.1) were fed to DNDC model together with crop management and climate data to setup 2345 model runs. For each run yield. mineral nitrogen application (Nmin). organic nitrogen application (Norg). phosphate application (P2O5) and Potassium chloride (K2O) were calculated. Mean value and squared geometric standard deviation were then calculated with R and a match with fertilizer production datasets in Ecoinvent database was found. Table 19 shows the results that were fed to the LCA model.

	Yield	Nmin	Norg	P2O5	K2O
Unit	kg ha^{-1}	kg ha^{-1}	kg ha^{-1}	kg ha^{-1}	kg ha^{-1}
Mean	6511.64	110.22	40.42	68.82	170.04
GSD^2	1.26	2.29	4.15	3.85	2.27
Ecoinvent dataset		Urea. as N	Compost	Triple superphosphate. as P2O5	Potassium chloride. as K2O

TABLE 19 FERTILIZERS AND YIELD PROVIDED AS INPUT IN LCA MODEL

4.7.4. Seeds

Agronomic literature indicates an average use of 180-240 kg of rice seed per hectare. Rice seeds can be produced in other Italian regions outside of PPA. Although rice seed is produced in a slightly different way. in the present case study the same dried rice produced in PPA in has been re-entered in the LCA model in a closed loop fashion. Differences between rice seed and edible rice have been considered negligible. Transport of seed. preservative products and use of storage facilities (including capital goods) have been considered as in the Ecoinvent dataset “rice seed. at regional storehouse”. Dataset includes seed transport to the processing centre. treatment (pre-cleaning. cleaning. eventually drying. chemical dressing and bag filling). storage and afterwards transport to the regional storage centre.

4.7.5. Farm size

In Table 20 the distribution of farms in four size categories is shown: from 0 to more than 300 ha. Farm size probability distribution has been fitted over a lognormal distribution curve and mean and geometric standard deviation (GSD) were calculated and inserted in the LCA model (Figure 23). The mean rice farms' size in PPA was calculated as 49.57 ha. while GSD as 3.01.

Province	Number 0-50	Area [ha] 0-50	Number 50-150	Area [ha] 50-150	Number 150-300	Area [ha] 150-300	Number 300+	Area [ha] 300+	Total Number	Total area
AL	137	2681	47	3839	3	657	0	0	187	7177
BI	63	1325	24	2186	2	318	0	0	89	3829
CN	14	177	1	51	0	0	0	0	15	228
NO	444	9942	220	17530	23	4758	2	798	689	33027
PV	2	50	2	137	1	173	0	0	5	360
TO	7	65	2	143	0	0	0	0	9	208
VC	929	20499	433	36297	48	8939	7	3335	1417	69070
Total	1596	34738	729	60181	77	14845	9	4133	2411	113897

TABLE 20 SIZE OF RICE FARMS IN PIEDMONT

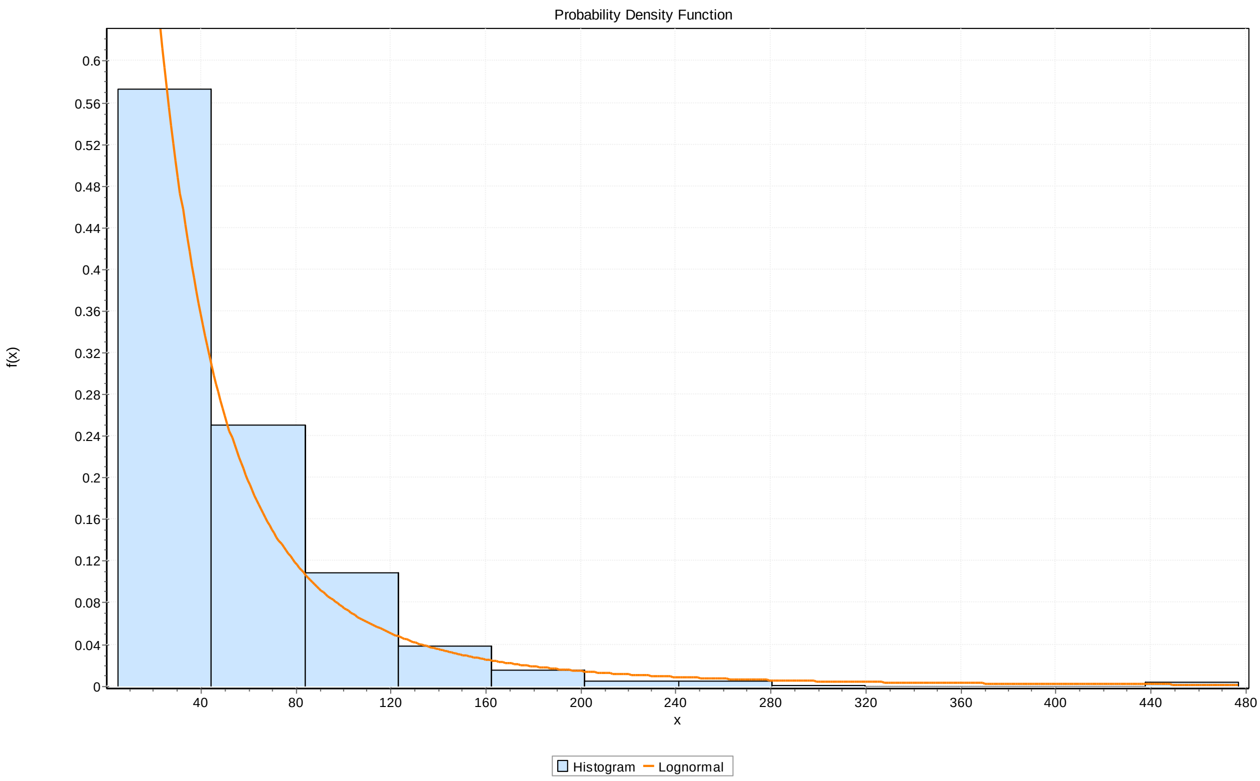


FIGURE 23 PROBABILITY DENSITY FUNCTION OF FARM SIZE AND LOGNORMAL FIT

4.7.6. *CAPITAL GOODS*

Many life cycle assessment case studies neglect the production of capital goods that are necessary to manufacture a good or to provide a service (Frischknecht et al. 2007). However in particular cases. such impacts cannot be neglected. Blengini & Busto (Blengini and Busto 2009) demonstrated that the contribution of capital goods has a noticeable weight (6%) on energy requirement and water consumptions. while the contribution to ozone depletion (ODP). acidification (AP) and global warming (GWP) is 3.9%. 2.4% and 1.6%. respectively. For this reason in this study capital goods were considered as much as data availability allowed it. In particular. only data upon rice cultivation machinery was retrieved (Berruto and Busato 2007). Tractors. harvesters and tools were associated to three farm size categories to account for differences in mechanization intensity. The Functional unit is 1 kg of agricultural machinery. As farm size is not given as a deterministic value but as a distribution. an internal parameter of the LCA model is set to automatically attribute the corresponding machinery to farm size.

Ecoinvent process	Machinery (kg per 50 ha farm)	Machinery (kg per 150 ha farm)	Machinery (kg per 300 ha farm)
tractor	10600	27650	40550
harvester	9000	12000	24000
trailer	3900	5400	6900
agricultural machinery. general	600	800	1300
agricultural machinery. tillage	3850	6150	9950

TABLE 21 MECHANIZATION INTENSITY FOR THREE DIFFERENT FARM SIZE CATEGORIES

4.7.7. FIELD BURNING OF RESIDUES

In order to determine the amount of air pollutants generated as a result of rice straw burning. an emission factors (EF) approach is used. EF's are expressed in terms of mass of pollutant emitted per unit mass of dry fuel consumed.

According to IPCC (Paustian et al. 2006) EF's can be calculated using Equation 2 is used to quantify air pollutant emissions from rice straw open field burning:

EQUATION 2

$$E_a = Q_{SSFB} \times EF_a \times f_{Co}$$

where a = Pollutant species; E_a = Emission of a in Mg/yr; EF_a = Emission factor of a in g/kg of dry straw; f_{Co} = Combustion factor. fraction of the mass combusted during the course of a fire. = 0.80 (default value as per IPCC 2006 guidelines); Q_{SSFB} = Quantity of rice straw subject to open field burning in Gg/yr (Paustian et al. 2006; Gadde et al. 2009).

However this methodology can only account for GHG emissions derived from straw burning and does not consider other important fluxes of pollutant. For example. the open burning of rice straw produces hazardous air pollutants. volatile organic compounds. polycyclic aromatic hydrocarbons. particulate matter (PM). organic carbon particles. elemental carbon particles and other air pollutants (Andreae and Merlet 2001).

Emission factors reported in Yu et al. (Yu et al. 2012) were hence used. The study calculated EF's for a specific emission episode in Taiwan in 2002. EF's are reported in Table 22 and address emissions of CO. NMHCs (non-methane hydrocarbon). NO_x. PM_{10} and SO_2.

Pollutant	Mean emission [g kg-1]	SD
CO	30.3	1.45
NMHCs	5.14	0.13
VOC	2.13	0
NO_X	3.44	0.09
PM_{10}	6.28	0.34
SO_2	0.058	0.003

TABLE 22 EMISSION FACTORS FOR RICE STRAW BURNING

A total production of 344787 tons of rice straw was calculated adopting an average yield of 5 tons per hectare (A.P.E.V.V. 2002) and calculating an overall burnt area of 68957 ha (63% of total calculated area).

4.7.8. *POST-HARVEST OPERATIONS*

Harvested paddy rice must be dried to reduce moisture in the grains in order to prevent mould formation during storage (Blengini and Busto 2009). Before drying. the typical water content of rice ranges between 20 and 30%. After drying. rice water content must be kept under 14%. This is also a normative requirement.

To model rice drying process. “Grain drying. low temperature” of Ecoinvent database was modified. The process takes into account the energy demand (supplied by burning light fuel oil and consumption of electricity) for evaporating 1 kg of water. when drying grain at low temperature (80-90°C). Also included in the inventory is the infrastructure (building and machinery). Inventoried outputs are heat waste and air emissions from combustion. Not included are waste and other air emissions (like dust).

The process was modified account for the lower temperatures used in rice drying (32-50°C).

The functional unit is kg water evaporated which is calculated by Equation 3 (W_{evap}) according to an output water content (W_o) of 13%. To account for differences in input water content (W_i). a triangular distribution was entered in the model with an average value of 25% and 30% and 20% as maximum and minimum values respectively.

Thermal energy is obtained from the combustion of light fuel oil with an average gross calorific value of 42.5 [MJ/kg]. Energy used for water evaporation was calculated as 4.448 [MJ/kg $_{H2O\ evaporated}$] using literature data (Busto 2006).

EQUATION 3

$$\text{Wevap} = \frac{(\text{Wo} - \text{Wi})}{(\text{Wi} - 100)}$$

4.7.9. *IRRIGATION*

Irrigation was modeled according literature data (Blengini and Busto 2009) and a mean water consumption value of 27500 l/ha was used with a reuse factor or 28%.

4.8.LCA IMPACT ASSESSMENT

Life cycle impact assessment (LCIA) translates emissions and resource extractions into a limited number of environmental impact scores by means of so-called characterisation factors. To perform Life Cycle Assessment (LCA) studies, there is a variety of Life Cycle Impact Assessment (LCIA) methods available. The International Standard for LCA (ISO 14040-14044) does not specify which LCIA method should be used, which means the choice for LCIA method is up to the author of the study.

There are two mainstream ways to derive characterisation factors, at midpoint level and at endpoint level. Midpoint indicators focus on single environmental problems, for example climate change or acidification. Endpoint indicators show the environmental impact on three higher aggregation levels, being the 1) effect on human health, 2) biodiversity and 3) resource scarcity. Converting midpoints to endpoints simplifies the interpretation of the LCIA results. However, with each aggregation step, uncertainty in the results increases. In this study ReCiPe 2016 method was chosen using midpoint impact indicator were used in order to reduce uncertainty.

The figure below provides an overview of the structure of ReCiPe.

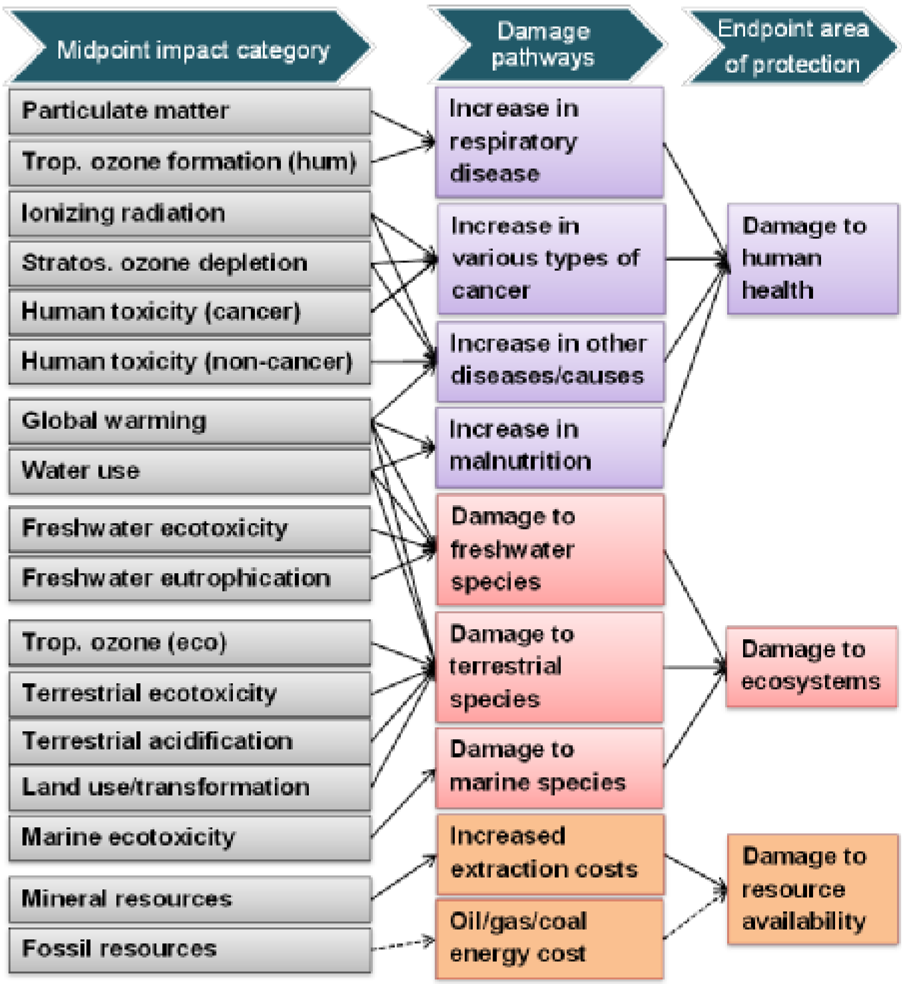


Figure 24 RELATIONSHIP BETWEEN LCI PARAMETERS (LEFT), MIDPOINT INDICATOR (MIDDLE) AND ENDPOINT INDICATOR (RIGHT) IN RECIPE 2016.

Following the same strategy as in ReCiPe2008, different sources of uncertainty and different choices were grouped into a limited number of perspectives or scenarios, according to the "Cultural Theory" (Thompson et al., 1990). These perspectives do not claim to represent archetypes of human behaviour, they are merely used to group similar types of assumptions and choices.

Three perspectives were included in ReCiPe2016:

1. The individualistic perspective is based on the short-term interest, impact types that are undisputed, and technological optimism regarding human adaptation.
2. The hierarchist perspective is based on scientific consensus regarding the time frame and plausibility of impact mechanisms.
3. The egalitarian perspective is the most precautionary perspective, considering the longest time frame and all impact pathways for which data is available.

In this study the hierarchist perspective was chosen.

4.9. RESULTS AND DISCUSSION

Categoria d'impatto	Unità	Total	Rice drying	Fertilizer production	Field operations	Direct field emissions	Machinery	Transport	Rice straw burning	Pesticide	Irrigation
Global warming	kg CO2 eq	2.79E+00	9.15E-02	1.59E-01	1.26E-01	2.36E+00	4.64E-02	4.18E-04	0.00E+00	1.33E-02	0.00E+00
Stratospheric ozone depletion	kg CFC11 eq	4.35E-06	3.61E-08	8.24E-08	6.28E-08	4.10E-06	1.41E-08	1.80E-10	0.00E+00	5.32E-08	0.00E+00
Ionizing radiation	kBq Co-60 eq	1.62E-01	1.04E-01	2.61E-02	1.11E-02	0.00E+00	1.68E-02	6.93E-05	0.00E+00	4.36E-03	0.00E+00
Ozone formation, Human health	kg NOx eq	2.65E-03	1.30E-04	3.75E-04	1.37E-03	0.00E+00	9.94E-05	2.87E-06	6.45E-04	2.83E-05	0.00E+00
Fine particulate matter formation	kg PM2.5 eq	9.91E-04	7.65E-05	4.50E-04	3.35E-04	2.64E-05	6.42E-05	8.90E-07	1.17E-05	2.63E-05	0.00E+00
Ozone formation, Terrestrial ecosystems	kg NOx eq	3.09E-03	1.35E-04	3.82E-04	1.39E-03	0.00E+00	1.12E-04	2.94E-06	1.04E-03	2.93E-05	0.00E+00
Terrestrial acidification	kg SO2 eq	2.73E-03	2.22E-04	1.36E-03	6.55E-04	2.16E-04	1.50E-04	1.72E-06	4.04E-05	8.18E-05	0.00E+00
Freshwater eutrophication	kg P eq	1.83E-04	2.60E-05	1.00E-04	1.78E-05	0.00E+00	2.64E-05	9.91E-08	0.00E+00	1.19E-05	0.00E+00
Marine eutrophication	kg N eq	1.21E-04	1.62E-06	5.80E-06	9.12E-07	1.09E-04	1.17E-06	5.45E-09	0.00E+00	3.05E-06	0.00E+00
Terrestrial ecotoxicity	kg 1,4-DCB	2.13E+00	8.87E-01	5.12E-01	3.13E-01	0.00E+00	3.43E-01	1.11E-03	0.00E+00	7.18E-02	0.00E+00

Freshwater ecotoxicity	kg 1,4-DCB	3.69E-03	7.28E-05	9.81E-05	6.95E-05	0.00E+00	8.53E-05	4.81E-07	0.00E+00	3.36E-03	0.00E+00
Marine ecotoxicity	kg 1,4-DCB	1.74E-03	5.05E-04	4.17E-04	2.48E-04	0.00E+00	2.92E-04	1.20E-06	0.00E+00	2.76E-04	0.00E+00
Human carcinogenic toxicity	kg 1,4-DCB	2.08E-02	4.94E-03	5.15E-03	4.08E-03	0.00E+00	5.98E-03	2.93E-05	0.00E+00	5.83E-04	0.00E+00
Human non-carcinogenic toxicity	kg 1,4-DCB	1.39E-01	9.41E-03	1.80E-02	9.86E-02	0.00E+00	1.08E-02	1.09E-03	0.00E+00	1.04E-03	0.00E+00
Land use	m2a crop eq	7.87E-02	7.30E-02	2.16E-03	2.99E-03	0.00E+00	4.64E-04	3.09E-05	0.00E+00	1.43E-04	0.00E+00
Water consumption	m3	6.45E+00	1.32E+00	3.65E-01	2.33E-01	0.00E+00	4.13E-01	2.44E-03	0.00E+00	5.25E-02	4.05E+00

TABLE 23 RESULTS FOR 1 KG OF RICE CALCULATED WITH RECIPE 2016 MIDPOINT INDICATORS

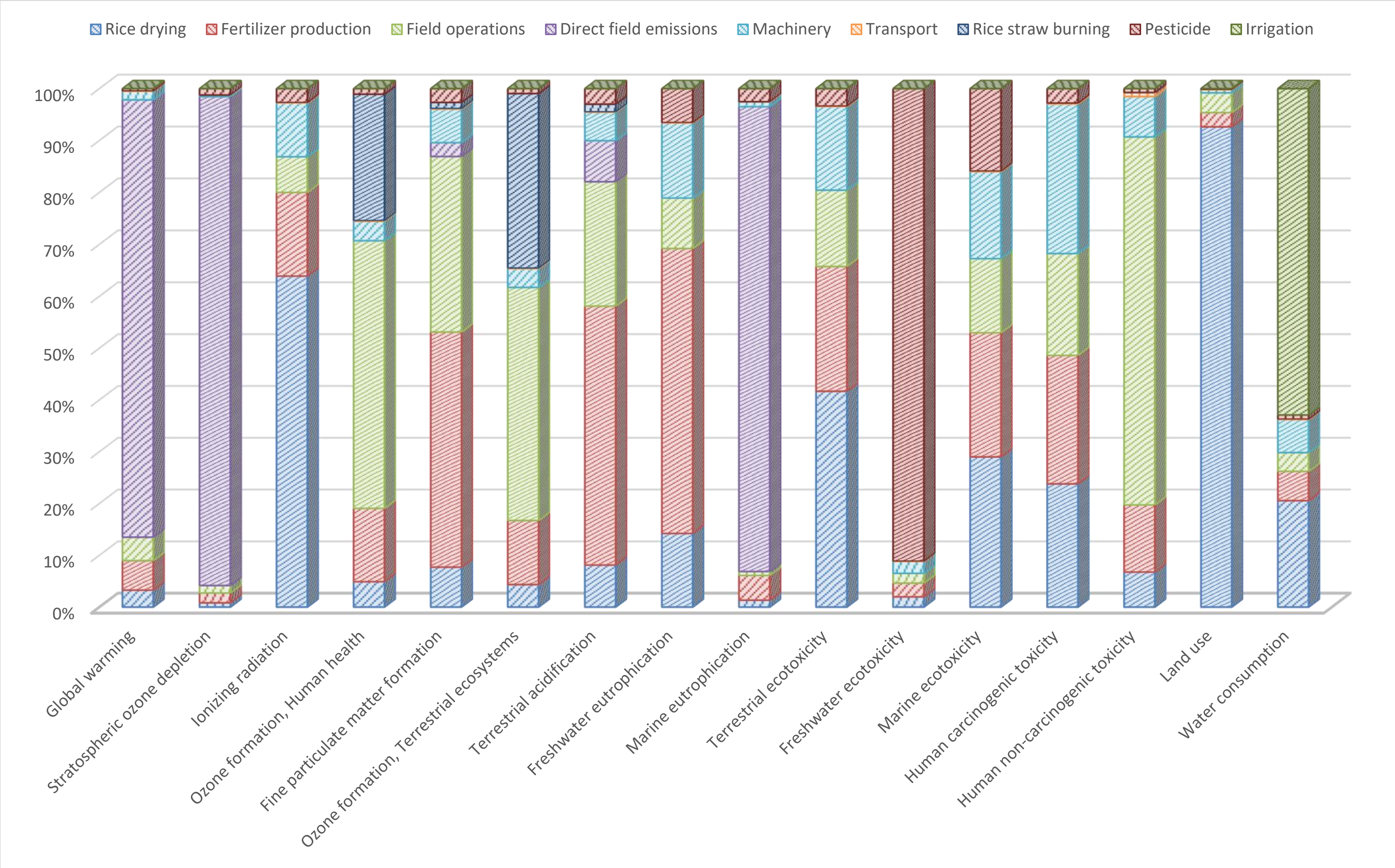


Figure 25 Contribution analysis for 1 kg of rice calculated with ReCiPe 2016 Midpoint indicators

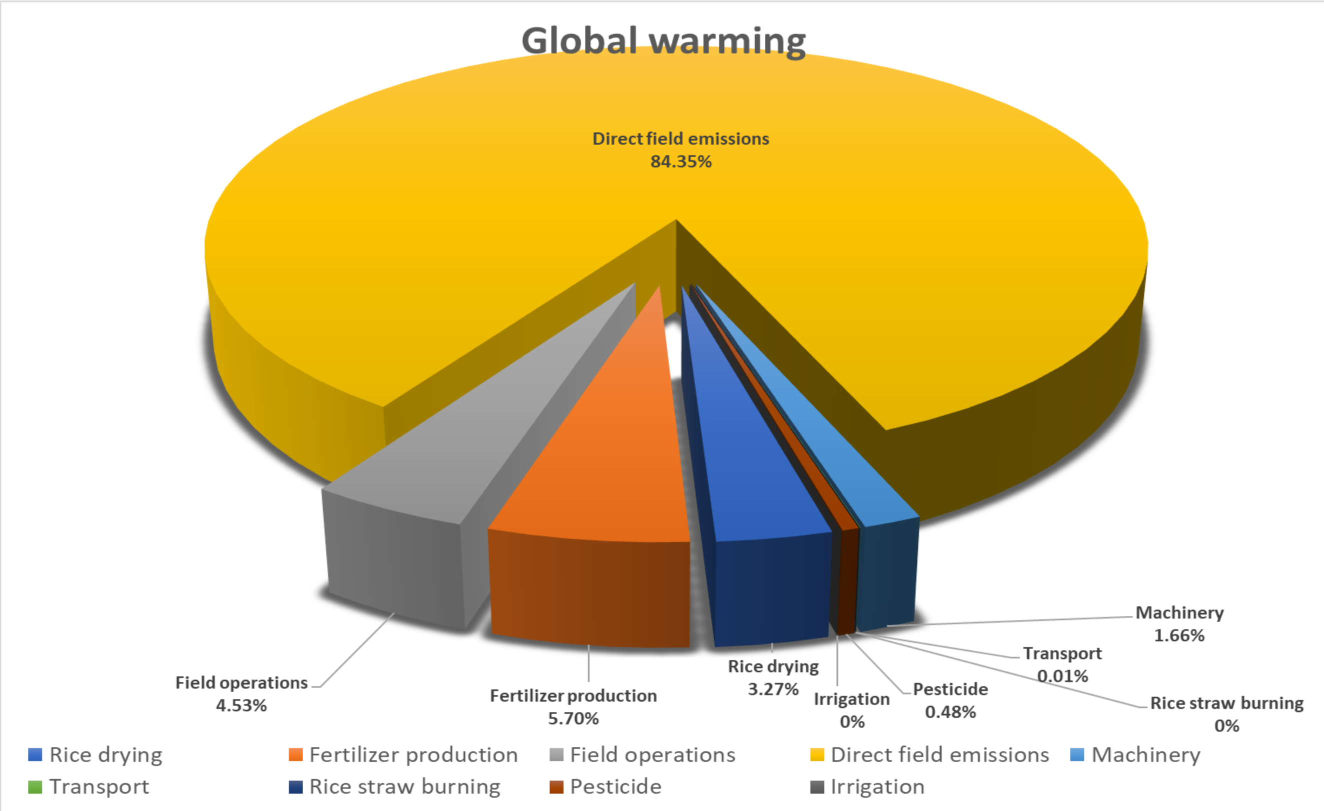


FIGURE 26 CONTRIBUTION ANALYSIS OF GLOBAL WARMING FOR 1 KG OF RICE

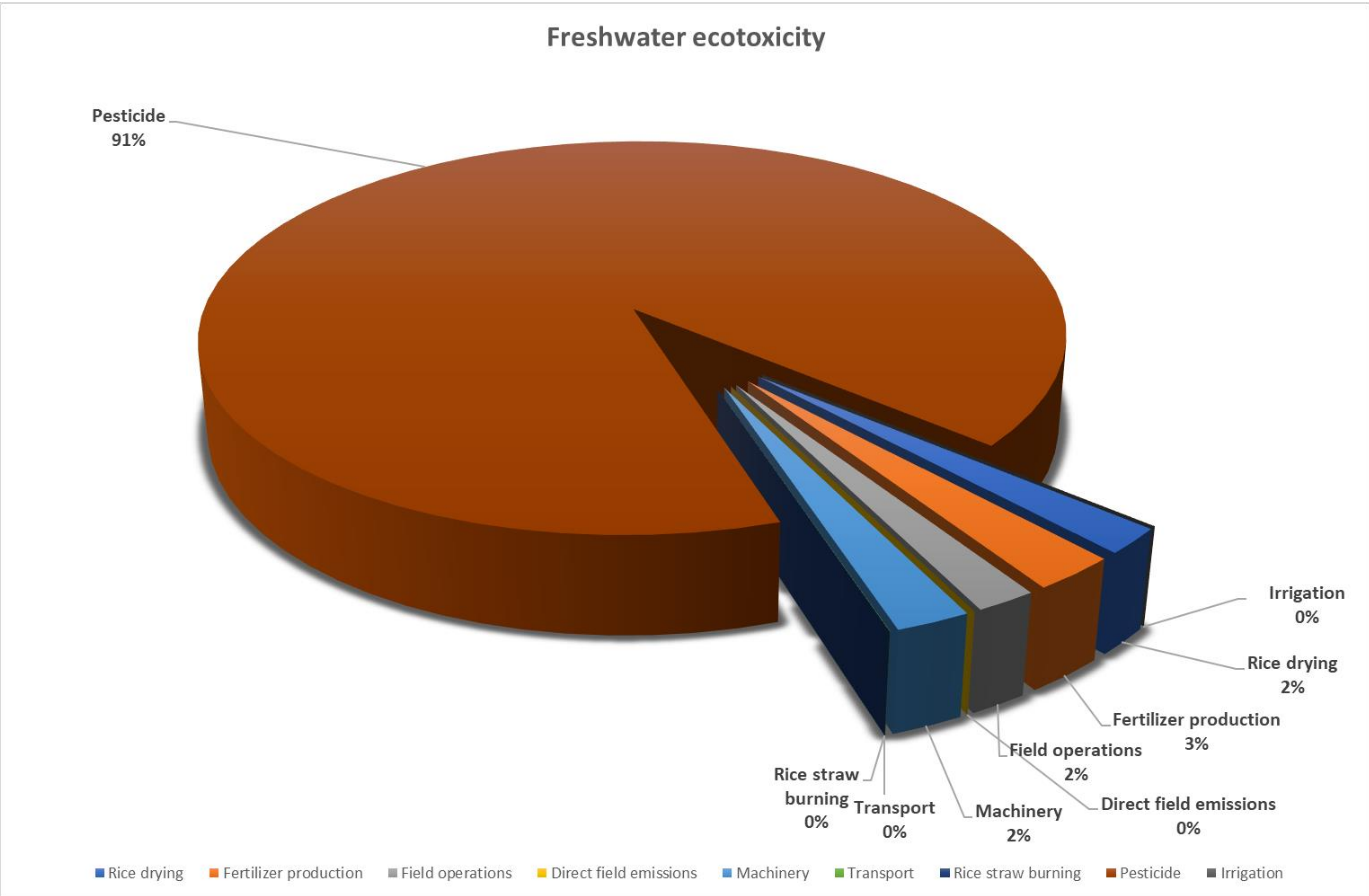


FIGURE 27 CONTRIBUTION ANALYSIS OF FRESHWATER ECOTOXICITY FOR 1 KG OF RICE

4.10. REFERENCES

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5. Conclusions and perspectives for future research

The main goal of this PhD thesis was to put forward LCA methodology enhancing his suitability to model agricultural and agricultural based productions.

The research first focused on LCA case studies. In particular LCA application to rice production in Piedmont highlined some of the limits of this methodology in particular related to the difficulty of estimating direct emissions from the field and to model pesticide impacts.

Field emissions are extremely variable spatially and temporally and they heavily contribute to some impact categories and hence on the overall impacts of products. Our research over the LCA of rice demonstrated their important contribution in global warming (68%), acidification (41%), eutrophication (76%) and photochemical oxidants (92%) indicators.

This said, when determining the impact of a generic product making use of agricultural processes (e.g. food or agro-energy products), this high variability reflects on overall life cycle impacts greatly increasing uncertainty. In fact, is it is quite common to analyze agricultural productions coming from different areas and (sometimes) cultivated in different periods. Even in surrounding areas, emissions can be subject to strong variations due to soil or crop management differences and even the same field can yield different emissions in different periods due to climatic differences.

Simplified approaches, such as IPCC tier 1 or 2 methodology, correlate emissions to few parameters (mainly nitrogen input only) and disregard other important influencing aspects such as soil, climatic and other crop management practices. This approach is very helpful to give a rough estimate of emissions especially at national scale. At plot, landscape or regional scale, simplified approaches are associated to a very high degree of uncertainty.

Initially LCA has been developed as a site and time independent tool. As a result, estimations of on-farm pollutant emissions and pollutant fate models do not consider field scale and regional variability in soil, catchment, climate and farmer practices. It is hence very difficult to estimate emissions which can be considered representative of a geographical area and of a particular year time representative emissions to improve impact assessment in LCA by better considering this variability by coupling LCA with a dynamic simulation model that considers soil, catchment, climate and farmer practices. This will allow the determination of total life cycle environmental impacts at the catchment/landscape spatial level.

Another development in LCA driven by its application for agricultural systems is the improvement of impact assessment through a better description of the fate mechanisms following emissions. Initially LCA has been developed as a site and time independent tool. Impact values result from the integration across emission/impact locations (world) and over time (infinite horizon) in an assumed steady state (Guinée et al. 2002). This over simplification is considered acceptable for global scale impacts (climate change) but considered simplistic for more regional ones, such as aquatic eutrophication. In fact, both ISO standards (ISO 2006) and SETAC (Society of Environmental Toxicology and Chemistry) recommend adopting spatially differentiated characterization factors and, in the last few years, several sets of country-specific factors have been proposed for eutrophication, as well as new methodologies (RECIPE, TRACI, EDIP2003, LUCAS). This research topic is currently being pursued on regional (Gallego et al. 2009) and national scales (Toffoletto et al. 2007), but also at catchment scale, considered appropriate to better model regional impacts such as eutrophication (Basset-mens et al. 2006a; Hauschild and Potting 2005; Seppala, Knuuttila, and Silvo 2004), because it allows considering spatial variability of soils and climate as well as the complex interactions among farms at supra-farm spatial levels.

In summary, improving our understanding of environmental impacts of farming systems requires coupling LCA with dynamic models, to consider spatial and temporal variability of emissions, as well as a better understanding of the complex fate mechanisms that link them to impacts. For eutrophication in particular, the fate factor of eutrophying pollutants in catchments (emission at the outlet of the catchment over emission from the catchment's soils) and particularly of nitrates, reflects one of these complex environmental mechanisms (Basset-Mens et al. 2006b). Establishing fate factors of nitrates in catchments can also identify trade-offs between impact categories. In fact, in riparian zones, nitrates can be transformed into N gases by heterotrophic denitrification (the anaerobic microbial conversion of nitrate to nitrous oxide and nitrogen gas) thus swapping eutrophying emissions with important greenhouse gases fluxes, N2O's Global Warming Power at 100 years being 310 times greater than that of CO2 (Guinée et al. 2002).

In the short to medium term, this thesis project aims at contributing to methodological development in LCA by (i) comparing the estimation of pollutant emissions using dynamic models to the common practice of adopting emission factors and (ii) comparing the use of spatialized fate factors for nitrate to the common practice of assuming a fate factor equal to 1, i.e. assuming that all of the nitrate emitted from the catchment's soils contributes to eutrophication and that no transformation effects (e.g. denitrification or plant uptake) occur. The integration of dynamic models into LCA as pursued in this project will potentially present a major scientific achievement.

In the medium to long term, it will develop the potential of LCA as a tool for the analysis of policies in different fields (e.g. renewable energy or sustainable agriculture), offering an integrated framework capable of grasping the complex interactions among farms at supra-farm spatial levels such as the catchment or agricultural landscape. This will help to strike a better balance between the environmental and economic impacts of agriculture and is thus very relevant to society.

In the long term, it has the potential to contribute to a sustainable development of agriculture by pointing out the most appropriate policies aiming at this goal.

6. Annex A: The life cycle of rice: LCA of alternative agri-food chain management systems in Vercelli (Italy).

Journal of Environmental Management 90 (2009) 1512–1522

Contents lists available at ScienceDirect

Journal of Environmental Management

journal homepage: www.elsevier.com/locate/jenvman

The life cycle of rice: LCA of alternative agri-food chain management systems in Vercelli (Italy)

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ARTICLE INFO

Article history:
Received 15 August 2007
Received in revised form
24 September 2008
Accepted 22 October 2008
Available online 30 November 2008

Keywords:
Life cycle assessment
Rice
Capital goods
Organic farming
Sustainable agriculture

ABSTRACT

The Vercelli rice district in northern Italy plays a key role in the agri-food industry in a country which accounts for more than 50% of the EU rice production and exports roughly 70%. However, although wealth and jobs are created, the sector is said to be responsible for environmental impacts that are increasingly being perceived as topical. As a complex and comprehensive environmental evaluation is necessary to understand and manage the environmental impact of the agri-food chain, the Life Cycle Assessment (LCA) methodology has been applied to the rice production system: from the paddy field to the supermarket. The LCA has pointed out the magnitude of impact per kg of delivered white milled rice: a CO_{2eq} emission of 2.9 kg, a primary energy consumption of 17.8 MJ and the use of 4.9 m^3 of water for irrigation purposes. Improvement scenarios have been analysed considering alternative rice farming and food processing methods, such as organic and upland farming, as well as parboiling. The research has shown that organic and upland farming have the potential to decrease the impact per unit of cultivated area. However, due to the lower grain yields, the environmental benefits per kg of the final products are greatly reduced in the case of upland rice production and almost cancelled for organic rice. LCA has proved to be an effective tool for understanding the eco-profile of Italian rice and should be used for transparent and credible communication between suppliers and their customers.

1. Introduction

The Vercelli rice district, which extends over a cultivated area of 68 000 ha in the River Po valley in NW Italy (approx. 45° long N), is known as one of the world's most technologically advanced rice cultivation areas and plays a key role in the Italian agri-food industry, representing 33% of the national rice production.

Among the rice producing countries, Italy only ranks 27th, with a national production rate for paddy rice (Rough Rice: threshed unmilled rice) of 1.44 million tons in the year 2006 and an average grain yield of 6.34 t/ha (Ente Risi, 2007). Nonetheless, this amount represents fully 50% of rice production in the European Union. The most striking characteristic of the Italian rice industry is that roughly 70% of Italian rice is exported, while in other countries the average domestic consumption is 94% (FAOSTAT, 2007).

Rice production generates wealth and jobs, but also creates environmental impacts that some believe to be unacceptably high (Tilman et al., 2001; Wenjun et al., 2006). Apart from soil and water pollution and consumption of energy and raw materials, paddy fields (irrigated or flooded land used for growing rice) are in fact claimed to be responsible for 10–13% of worldwide methane anthropogenic emissions (Neue, 1997), thus contributing to a great extent to the global warming phenomenon.

For these reasons, it is becoming more and more crucial to understand and manage the environmental impacts of rice production. However, according to Breiling et al. (2005), solutions that try to mitigate the impacts, but disregard the dependencies between the single processes in the agri-food chain and between agriculture and other sectors, are likely to fail. Thus, Life Cycle Assessment (LCA) is becoming more and more important in agri-food industries, as it can be used to assess the environmental performances of products, from their very beginning, throughout their whole life cycle.

With that said, applying LCA to the agri-food chain is complicated by the nature of the processes involved. Although modern and technological farming can be compared to the industrial systems in which LCA has become well known and accepted, there are some peculiar aspects that must be taken into account. In fact, while manufacturing can be regarded as a sequence of industrial processes which depends on human decisions and control, agriculture should, however, be considered more as a sequence of natural and industrial processes which man can drive, but cannot control completely. Thus, great care must be taken when designing an LCA for an agri-food chain to ensure that both human and natural processes are captured accurately by the model design.

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0301-4797/$ – see front matter © 2008 Elsevier Ltd. All rights reserved.
doi:10.1016/j.jenvman.2008.10.006

Important international studies have in fact focused on the harmonisation of LCA applications in agriculture (Audsley et al., 1997) and detailed research reports exist which deal with the life cycle of the most important agricultural commodities. Among such studies, Williams et al. (2005) have supplied a detailed report on the LCA of 10 agricultural and horticultural key commodities, rice being a notable exception.

Moreover, although there is abundant literature on the measurement of greenhouse emissions from paddy fields and associated mitigation issues (Dan et al., 2001; Deepanjan, 2003; Hou et al., 2000; Kewei and Patrick, 2003; Kruger and Frenzel, 2003; Leip et al., 2007; Neue, 1997; Sahrawat, 2004; Schütz et al., 1989; Singh et al., 1999; Watanabe et al., 1995), only the few studies discussed below applied the LCA methodology to the rice agri-food chain as a whole and, to authors' knowledge, there are no papers dealing with LCA of Italian rice. Breiling et al. (2005) applied LCA to understand the potential of no-tilling cultivation on greenhouse emissions in Japan. Harada et al. (2007) used a top-down LCA, based on economic input–output tables to assess CH_4 rice related emissions in Japan. Roy et al. (2007) assessed the potential of resource conservation and greenhouse saving of different parboiling processes in Bangladesh.

Given the importance of the sector in the EU and Italy, and growing concerns about its environmental impacts, the research was aimed at integrating and extending the findings of different studies from both international and Italian literature that were focused on specific environmental aspects of rice farming and processing. The overall objective was to determine the magnitude of impact per kg of delivered rice and to understand where impacts are concentrated within the rice production chain from paddy field to supermarket.

A from-cradle-to-gate LCA model of the white milled rice agri-food chain, which represents 97% of the production in the Vercelli district, was created according to the ISO 14040 standard (ISO 14040, 2006). The model was set up to represent the average rice farming and transformation processes using data gathered from two important rice farms in the area (A.A. Palestro and Cascina Canta), two important rice factories (Riso Gallo SpA and Riseria Re Carlo) and taking into consideration information supplied by the CRR (Research Centre for Rice) of Ente Risi (the National Agency for Italian rice development).

Due to the particularly intense mechanisation in the Vercelli farms, the influence of capital items has been investigated paying attention to the farm size: 50, 150 or 300 ha. In fact, according to Berruto and Busato (2007) an available tractor power of 3.2 kW/ha is expected for a 50 ha rice farm in Vercelli, while, according to Bailey et al. (2003), the available tractor power in UK is about 1 kW/ha. The inclusion of the environmental burdens of capital goods (production of machinery, roads, irrigation facilities and buildings) is in fact one of the most controversial issues when dealing with LCA in agriculture. The influence of capital goods is sometimes excluded due to lack of data (Roy et al., 2007; Høgaas Eide, 2002), but carefully addressed in other studies (Breiling et al., 2005; Harada et al., 2007; Williams et al., 2005). Audsley et al. (1997) addressed this issue with reference to arable crops in the UK and concluded that, with the exception of harvesting equipment, the contribution is insignificant because of the very high throughput of work over machine's life.

The study included analysis of LCAs for three alternative rice farming and food processing methods: organic farming, upland farming and parboiling (improvement scenarios). The LCA model for white milled rice (baseline scenario) was modified in order to investigate the potential for improvement in environmental performance of the rice industry. The quantitative environmental performance of organic farming was of particular interest. Organic rice, which presently accounts for 3% of the overall production, is in fact considered more and more as a product innovation that can help diversify production and create green market outlets. In such context, LCA can be used to supply the scientific basis for a transparent and credible communication with the general public about the value of applying this farming method.

2. Methodology

The Life Cycle Assessment (LCA) methodology has been used to evaluate the environmental profile of alternative rice farming and food processing methods. According to ISO 14040 (2006), an LCA comprises four main stages: *goal and scope definition, life cycle inventory, life cycle impact assessment, and interpretation* of the results.

Goal and Scope Definition is aimed at identifying the objectives, functional unit, system boundaries, cut-off criteria, data sources and data quality requirements.

Life Cycle Inventory (LCI) consists of a detailed compilation of all the environmental inputs (material and energy) and outputs (air, water and solid emissions) at each stage of the life cycle.

Life Cycle Impact Assessment (LCIA) aims at quantifying the relative importance of all the environmental burdens identified in the LCI by analysing their influence on selected environmental effects.

According to ISO 14044 (2006), the general framework of an LCIA method is composed of several mandatory elements (classification and characterisation) that convert LCI results into an indicator representative of each impact category and of optional elements (normalization and weighting) that lead to a single indicator comprehensive of all the impact categories using numerical factors based on value choices.

As there is still neither consensus on weighting nor on the best weighting method to adopt, the LCIA has been focused on the characterisation step, selecting the following indicators:

- GER (Gross Energy Requirement), used as an indicator of the total primary energy resource consumption: direct + indirect + feedstock (Boustead and Hancock, 1979);
- NRER (Non-Renewable Energy Requirement), used as an indicator of non-renewable energy use;
- GWP_{100} (Global Warming Potential), used as an indicator of the greenhouse effect;
- ODP (Ozone Depletion Potential), used as an indicator of the stratospheric ozone depletion;
- AP (Acidification Potential), used as an indicator of acid rain phenomenon;
- EP (Eutrophication Potential), used as an indicator of surface water eutrophication;
- POCP (Photochemical Ozone Creation Potential), used as an indicator of photo-smog creation;
- WU_t (Water Use total), used as an indicator of direct and indirect fresh water use;
- WU_d (Water Use direct), used as an indicator of the direct use of water for rice irrigation and processing.

The characterisation factors of GWP_{100}, which corresponds to the CO_2 equivalent emission for a period of 100 years (IPCC, 2001), were 1, 23 and 296 for CO_2, CH_4 and N_2O, respectively.

The characterisation factors of the ODP, AP, EP and POCP indicators can be found in SEMC (2000).

The final step of the LCA methodology involves the interpretation of LCI and LCIA stages, in order to find hot spots and compare alternative scenarios.

The SimaPro 7 (2006) software application and the Ecoinvent 1.3 (2004) database were used in this study to implement the LCA model and carry out the assessment.

3. LCA model of white milled rice (baseline scenario)

This section describes the application of the LCA methodology to the average rice farm and to the subsequent processes in the white milled rice chain. The model describes a typical farm in the Vercelli district in Italy that makes use of an average amount of products (fertilizers, pesticides, etc.) and which harvests an average yield of 7.03 t/ha of paddy rice, roughly corresponding to 6.12 t/ha of dried paddy rice (1995–2005 average value according to FAOSTAT). The cultivar has not been specified as the study is based on average grain yields.

3.1. System boundaries and functional unit

The setting of system boundaries in an LCA is a very delicate step that can influence the results to a great extent. It determines which processes will be included in the analysis and which will be left out, whether intentionally or unintentionally. If the boundary is set too narrowly, some important impacts may be undetected. If it is set too broadly, impacts other than those generated by the process of interest may be included. For the present analysis, the LCA model was carried out by including three subsystems of white milled rice chain, agricultural processes, drying and storing, and refining and packaging. System boundaries, as well as input and emission categories are shown in Fig. 1.

The indirect environmental burdens of capital goods where included in the agricultural subsystem because of the high level of mechanisation of the Vercelli farms. However, capital goods relevant to the post-agricultural phase were excluded based on the fact that the contribution of buildings and machinery used in the post-harvesting processes is virtually negligible (Williams et al., 2005). Rice straw was excluded from the agricultural phase. The reasons for this choice are discussed in the section on Allocation criteria and also in sensitivity analysis.

As the main function of the system under study is to supply rice as an agri-food product, the functional unit selected is 1 kg of refined rice packed and delivered to the supermarket. Thus consumer waste and the packaging waste management phases were not considered.

3.2. Data sources

The input data was obtained from different sources: on site records, interviews with farmers, agronomists and rice processing

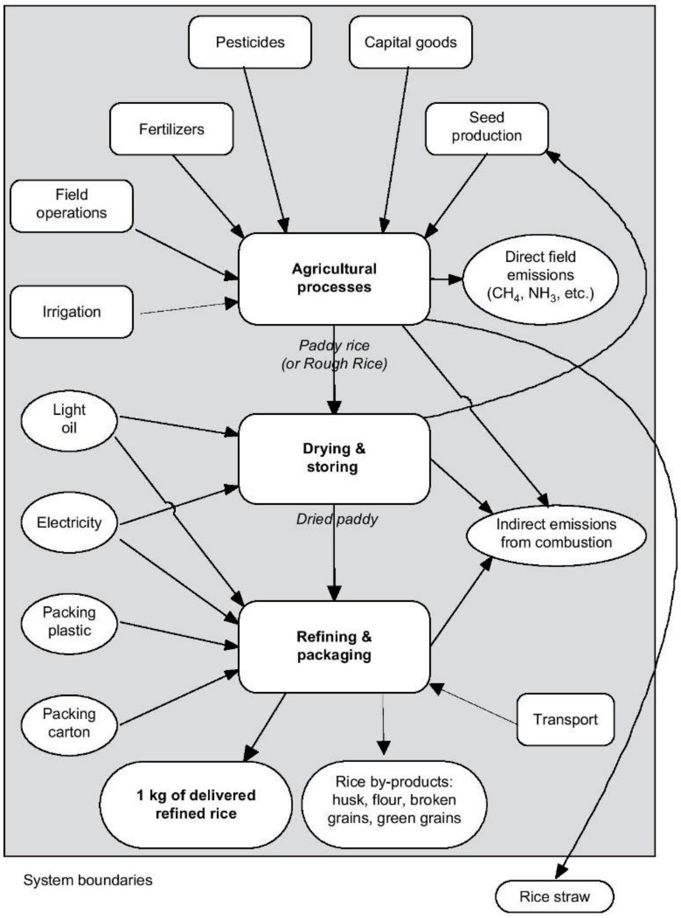


Fig. 1. System boundaries for white milled rice chain.

technicians, as well as specific literature on the Vercelli district and international literature (Table 1).

The crop management practices were investigated by interviewing the farmers and by using information supplied by Ente Risi. The key data relevant to fertilizers and pesticides were supplied by the rice farms or retrieved from literature. The data relevant to greenhouse emissions from paddy fields were either taken from the Regional Greenhouse Inventory (Regione Piemonte, 2005), estimated according to the Intergovernmental Panel on Climate Change models (IPCC, 2006) or retrieved from specific literature on the Vercelli district (Table 1). The water use was estimated according to Regional statistics (Regione Piemonte, 2007) and from the available literature. The direct energy use and ancillary materials used for farming and processing were obtained from field measurements or estimations. The inventory data for the energy and transport systems were retrieved from the Ecoinvent database.

Table 1
White milled rice production phases and data sources.

Production phase	Subsystem	Sources of data
Agricultural	Mechanical field operations	- Measured data from A.A. Palestro rice farm - Literature data (Finassi, 1990) - Ecoinvent database
	Fertilizers	- Measured data from A.A. Palestro rice farm - Literature data (Zavattaro et al., 2006) - Ecoinvent database
	Pesticides	- Measured data from A.A. Palestro rice farm - Literature data (Ferrero and Tabacchi, 2000) - Ecoinvent database
	Irrigation	- Measured data from Consorzio irriguo Ovest Sesia - Literature data (Allavena and Monti, 2007; Baldi et al., 1974; Greppi et al., 2007; Moletti, 1989; 1990; Regione Piemonte, 2007)
	Field emissions	- Literature data (Dan et al., 2001; Deepanjan, 2003; Hou et al., 2000; IPCC, 2006; Kewei and Patrick, 2003; Kruger and Frenzel, 2003; Leip et al., 2007; Prasuhn, 2002; Regione Piemonte, 2005; Schütz et al., 1989; Watanabe et al., 1995; Zavattaro et al., 2006)
	Seeds	- Measured data from A.A. Palestro rice farm - Personal communications from CRR Ente Risi - Ecoinvent database
	Capital goods	- Measured data from A.A. Palestro rice farm - Literature data (Finassi, 1990) - Personal communications from CRR Ente Risi - Ecoinvent database
Post-harvest processing	Drying and storing	- Measured data from A.A. Palestro and Riso Gallo SpA
	Rice refining and packaging	- Measured data from Riseria Re Carlo and Riso Gallo SpA - Ecoinvent database
Transportation	Field – farm	- Estimates from A.A. Palestro rice farm
	Farm – processing	- Estimates from A.A. Palestro rice farm
	Delivery	- Estimates from Ente Risi annual report (2007)

3.3. Allocation criteria

Rice production is a multifunctional process that generates marketable products and by-products, as well as residuals: refined rice, broken grains, rice flour, husk straw, etc. For this reason, allocation criteria have to be adopted in order to distribute environmental burdens among the chain subsystems. According to ISO 14040, when allocation is necessary, the criteria should reflect the physical relationship between the environmental burdens and the functions delivered by the system ISO 14040 (2006). In this research, the allocation of burdens to the co-products was based on relative economic value, as suggested by Williams et al. (2005). Rice straw was allocated no environmental burdens, due to its negligible market value. This leads to a simplification as straw might be harvested and used for electricity generation in a biomass power plant. However, according to current farming practices, around 30% of straw is burned on the site and the rest is incorporated in the soil. Straw burning results in a biogenic air emission that is considered neutral. In fact, according to the conventional approach relevant to the carbon cycle in agriculture (Williams et al., 2005), biogenic emission of CO_2 can be ignored, as the consumption of carbon dioxide during growth is balanced when the crops are consumed and the same quantity is released into the atmosphere. Incorporation leads to an improvement in soil properties that is already accounted for as a consequence of current farming practice. The exclusion of straw is, however, discussed further in the sensitivity analysis. Actual allocation values are based on the details of the model and so are reported at the end of the section on Inventory analysis.

3.4. Inventory analysis

This section explains how the collected data were adapted to the LCA model and gives details on assumptions that were made.

3.4.1. Field operations

Field operations include the maintenance of watering canals, bank management, ploughing, fertilising, harrowing, sowing, application of plant protection products and harvesting (Table 2). These operations generate emissions from fuel burning, consumption of resources and environmental impacts from machinery use in agricultural processes. The paddy field bank management was modelled by adapting ploughing from the Ecoinvent database, according to Finassi (1990) and by considering the average yearly working hours. It was assumed that the

Table 2
LCI of white milled rice: model entries for mechanical field operations.

Field process	Database entry	Reference unit	Q.ty
Maintenance of watering canals	*Excavation hydraulic digger*	m^3	5
Bank management	*Ploughing*	ha	0.5
Ploughing	*Tillage, ploughing*	ha	1
Fertilising	*Fertilising, by broadcaster*	ha	1
Harrowing	*Tillage, harrowing, by rotary harrow*	ha	1
Sowing	*Sowing*	ha	1
Application of plant protection products	*Application of plant protection products, by field sprayer*	ha	1
Harvesting	*Combine harvesting*	ha	1

maintenance of the watering canals is conducted with a hydraulic excavator which moves 5 m^3 of soil per hectare. All other operations were modelled using Ecoinvent agricultural processes. The preliminary and concluding work and equipment transportation from farm to field were also included.

3.4.2. Fertilizers

The use of fertilizer is very different from farm to farm and mainly depends on the rice species and soil. The average mix of fertilizers was estimated using the data and personal communications from rice farms, which gave values that were substantially similar to the literature data (Zavattaro et al., 2006). A mix of single and multicomponent commercial fertilizers was then used in the model according to the overall content in terms of active ingredients (Table 3). The manufacturing processes were taken from Ecoinvent.

3.4.3. Pesticides

As the use of pesticide is also quite variable, data from both farmer interviews and literature (Ferrero and Tabacchi, 2000) were used. Commercial pesticides were modelled according to the active ingredients and according to inventory data from Ecoinvent (Table 4). In some cases, no precise match was found and an active ingredient of a similar chemical class was therefore adopted.

3.4.4. Irrigation

Irrigation in the Vercelli district is based on a complex network of canals where water flows by gravity without the use of any pumping systems. According to Regione Piemonte (2007), the yearly water intake for the irrigation of paddy fields in Vercelli is 15–20 000 m^3/ha. The available agronomic literature reports similar or higher figures: 15–20 000 m^3/ha, according to Greppi et al. (2007), 15–40 000 m^3/ha, according to Allavena and Monti (2007) and 40 000 m^3/ha, according to Baldi et al. (1974). Nevertheless, only part of the water intake is consumed by evapo-transpiration and plant growth: 6500–7500 m^3/ha, according to Allavena and Monti (2007). Another part is lost due to soil infiltration and the rest is discharged as surface runoff and partially re-used in downstream paddy fields. Hence, when conducting an LCA model, it is not appropriate to consider the total volume of water delivered to an individual paddy field. However, to authors' knowledge, there are no published data referring to water re-usage in the Vercelli district, therefore, for the purpose of this analysis, an estimate had to be made according to the information from international literature. Wichelns (2001) has in fact calculated the water requirement for rice farming in Egypt and reported that the water intake is 20 952 m^3/ha, the water re-usage being 28%. Bearing this in mind, a mean value of between 15 000 and 40 000 m^3/ha and a 28% water re-usage was considered, thus a value of 19 800 m^3/ha was used in the LCA model. This assumption is discussed in the sensitivity analysis.

Table 3
LCI of white milled rice: model entries for fertilizers.

Mix of commercial fertilizers	Database entry	Active ingredient	Q.ty (kg/ha)
Cuoio torrefatto (12%N); ORVET 8 (8%N); Urea (46%N); Calce Fosfopotassica (8%P_2O_5–22%K_2O–20%CaO); Complesso 18-0-36 (18%N–36%K_2O); ORVET 10-5-15 (10%N–5%P_2O_5–15%K_2O); Complesso 11-12-36 (11%N–12%P_2O_5–36%K_2O)	*Horn meal (cornunghia), at regional storehouse*	12%N	171.8
	Urea, as N, at regional storehouse	46% N	234.3
	Triple superphosphate as P_2O_5	21% P_2O_5	83.3
	Potassium chloride as K_2O	50% K_2O	270.1

3.4.5. Field emissions

Field emissions include direct air emissions of methane, nitrous oxide and ammonia, as well as emissions of phosphorus and nitrates to water. Carbon dioxide was considered neutral (Williams et al., 2005).

Anaerobic decomposition of organic matter and the consequent methane production are caused by water management practices, mainly due to long submersion times. The magnitude of such an emission is also influenced by fertilizer use and soil typology (Dan et al., 2001; Deepanjan, 2003; Harada et al., 2007; Kruger & Frenzel, 2003; Schütz et al., 1989; Watanabe et al., 1995).

In order to consider an average value of methane emission, while considering the general framework of the present study, different sources were examined (Table 1). The LCA model used the data from the Regional greenhouse gas inventory (Regione Piemonte, 2005), which reports the data on CH_4 emission from paddy fields in Vercelli, according to IPCC Guidelines (1996). Italy is in fact the only country in Europe which uses country-specific factors to estimate CH_4 emissions in the annual greenhouse gas inventory (Leip et al., 2007). According to Regione Piemonte (2005), a value of 48 g of methane per kg of paddy rice was therefore used in the LCA model. For comparison purposes, Schütz et al. (1989) monitored methane emission from a paddy field in northern Italy for a period of three years and the reported measurements correspond to a CH_4 emission of 25.7–107.1 g/kg of paddy rice.

As far as nitrous oxide is concerned, the LCA model was based on the data from the Regional greenhouse gas inventory (Regione Piemonte, 2005). Thus, a value of 0.2 g of N_2O per kg of paddy rice was used. Also N_2O emission from paddy fields is an important source of greenhouse gas (Deepanjan, 2003), but there is still a lack of the measured data. For instance, Dan et al. (2001) reported that field measurements of N_2O from a paddy field in Vercelli showed emissions below the detection limit ($<20\ \mu g\ N_2O\ m^{-2}\ h^{-1}$). According to Hou et al. (2000) and Kewei and Patrick (2003), methane and nitrous oxide from paddy fields are closely correlated and depend on the changes in the soil redox potential. According to Breiling et al. (2005), high uncertainties arise when CH_4 and N_2O are estimated together and no measurements exist of the total greenhouse gas emissions for a whole year: N_2O and CH_4 are likely to be in a tradeoff, but this relationship is not yet well understood. However, due to the remarkable uncertainties of data relevant to CH_4 and N_2O emissions, the robustness of the model was checked in the sensitivity analysis, using measured data that have recently been published by Leip et al. (2007).

Regarding ammonia emissions, data from the literature relevant to the study area were considered (Grignani et al., 1997) and a value of 1.14 g/kg of paddy rice was used. Phosphorus releases were calculated using Federal Agricultural Research Centre models (Prasuhn, 2002). The emission of P, due to leaching out to ground water, was 0.084 kg/ha, while loss for run-off to surface water was 0.3615 kg/ha. The nitrate emissions to ground water and surface water were taken from specific literature data (Grignani et al., 1997): 0.021 kg/kg and 0.085 kg/kg, respectively.

3.4.6. Seed production

According to the average data from the A.A. Palestro rice farm (Table 1), the average use of seed is 200 kg/ha. Although edible rice farming and rice seed farming are slightly different, the same unit process for dried rice was used in the LCA model in a closed loop fashion. The model also included road transportation of the seed (750 km), preservative products (2.4 g/kg of phtalamide compounds) and the use of electricity for storage (0.21 MJ/kg).

Table 4
LCI of white milled rice: model entries for pesticides.

Pesticide product	Active ingredient	Chemical class	Associated chemical class in Ecoinvent	Q.ty (kg/ha)
Gulliver	Azimsulfuron	Pyrimidinyl sulfonylurea	*[sulfonyl] urea compounds*	0.056
Londax 60 DF – Square 60 WDG	Bensulfuron	Pyrimidinyl sulfonylurea		
Pull 52 DF	Bensulfuron metil + metsulfuron metil	Pyrimidinyl sulfonylurea		
Sunrice	Ethoxysulfuron	Pyrimidinyl sulfonylurea		
Karmex	Diuron	Phenylurea	*diuron*	0.128
Buggy - Clinic 360	Gliphosate	Organophosphorus	*glyphosate*	0.774
Stratos ultra	Cycloxydim	Cyclohexene oxime	*phenoxy-compounds*	0.084
Aura	Profoxydim	Cyclohexene oxime		
K-Othrine	Deltametrin + Piperonyl butoxide	Pyrethroid	*pyretroid-compounds*	0.003
Dipterex	Triclorfon	Phosphonate	*organophosphorus compounds*	0.21
Heteran	Oxadiazon	Oxadiazolone	*cyclic N-compounds*	0.251
Nominee	Bispyribac-sodium	Pyrimidinyl oxybenzoic acid		
Rifit	Pretilaclor	Chloroacetanilide	*pesticide unspecified*	4.635
Cannicid – Poladan	Dalapon	Halogenated aliphatic		

3.4.7. Capital goods

The indirect environmental burdens, caused by the manufacturing of capital goods (machinery) used in the agricultural phase (Fig. 1), were calculated. Data from farmers, machinery producers and from CRR Ente Risi, as well as literature studies concerning rice mechanisation in Italy (Berruto and Busato, 2007; Finassi, 1990) were used. The survey emphasised the high mechanisation of rice farms in Vercelli, mainly due to the fact that mechanical field operations have to be carried out simultaneously and must be concluded in a short time. Moreover, it has been observed that this equipment is exclusively used for rice production, the number and size of tractors, harvesters, trailers and tillage tools per hectare being strongly correlated to the farm size (Berruto and Busato, 2007). An average amount of rice produced during the lifetime of the machinery was calculated. Therefore, assuming an average life of 13 years and assuming a yield of 7.04 t/ha, the total production was 91.49 t/ha. This leads to an allocation factor which depends on the farm size (50, 150 and 300 ha), as reported in Table 5. Inventory data for the manufacturing, maintenance and disposal of machinery were taken from Ecoinvent.

3.4.8. Drying and storing

Harvested paddy rice must be dried to reduce moisture in the grains in order to prevent mould formation during storage. The model for rice drying and storing was created according to the average measured data (Table 1). The parameter relevant to the energy used for water evaporation was set at $E = 4.06$ [MJ/kg H_2O evaporated]. For 1 kg of dried paddy, 0.15 kg of water is evaporated. Therefore, 0.7 MJ of light fuel oil and 0.03 MJ of electricity per kg of dried rice were used in the model. Such an electricity use also included refrigeration, which may be necessary after drying to prevent the grains from recovering moisture, and storage.

Table 5
LCI of white milled rice: model entries and allocation factors for capital goods.

Ecoinvent entry	kg of machinery (50 ha farm)	kg of machinery (150 ha farm)	kg of machinery (300 ha farm)
Tractor	10 600	27 650	40 550
Harvester	9000	12 000	24 000
Trailer	3900	5400	6900
Agricultural machinery, general	600	800	1300
Agricultural machinery, tillage	3850	6150	9950
Allocation factor (yield, t/ha × years × farm size, ha^{-1})	2.19×10^{-7}	7.29×10^{-8}	3.64×10^{-8}

3.4.9. Rice refining and packaging

Dried paddy has a non-edible husk or hull surrounding the kernel. During milling, all the stalks and other unwanted materials are removed from the rough rice by a sequence of processes which make use of electricity: cleaning, hulling, milling or whitening, polishing, grading and sorting. The LCA model was based on the average measured data (Table 1). The energy consumption for refining and packaging was 0.277 MJ/kg of dried rice. Rice is packed inside an internal low density polyethylene (LDPE) bag (10 g/kg) and an external carton box (50 g/kg). The plastic film around the pallet (0.36 g/kg) was also accounted for. The inventory data for the packaging materials were taken from Ecoinvent. An allocation based on the economic value of refined rice and co-products was used, thus about 91% of the total impact was assigned to white milled rice (Table 6).

3.4.10. Transportation

Transportation of rice products from the farm to the retailer, in Italy and in the rest of Europe, was modelled according to the routes, the type of transport and distances shown in Table 7 and according to the sources quoted in Table 1. All the transport operations in the pre-harvesting phase were included in their specific subsystems, i.e. the transportation of raw materials for fertilizer manufacturing was included in the fertilizer subsystem. An empty return trip was only assumed for the processing plant to storage route. The environmental burdens relevant to the use of transport systems were taken from the Ecoinvent.

4. LCA models of alternative rice farming and processing

LCA models of alternative rice farming and food processing, such as organic farming, upland farming and parboiling (improvement

Table 6
Allocation criteria for rice products and by-products after processing.

Product	Percentage by weight	Market value (€/t)	Market value for 1 t of dried rice (€/t)	Allocation factor
Refined rice	62%	500	310	90.75%
Rice husk	20%	20	4	9.25%
Rice flour	8%	100	8	
Broken rice	8%	220	17.6	
Green grains	2%	100	2	
	100%	–	341.6	100%

Table 7
Estimated transportation distances.

Transport route	Ecoinvent entry	Distance (km)
Field to farm	*Tractor and trailer*	1
Farm to processing plant	*Lorry 16 t*	20
Processing plant to storage	*Lorry 28 t*	40
Storage to local distribution	*Lorry 32 t*	200
Storage to international distribution	*Lorry 32 t*	880
	Train	220

scenarios), were carried out by modifying the LCA model for the exported white milled rice (baseline scenario). As 40% of the rice farms have between 30 and 100 ha, representing 49% of the rice cultivation area in the Vercelli district (Ente Risi, 2007), the models were based on the 50 ha farm. Only the differences between the improvement and the baseline scenarios are here reported.

4.1. Organic rice

Organic rice farming involves 3% of the cultivation area in the Vercelli district (Ente Risi, 2004). The consequence of this is a general lack of data. The model relevant to organic rice was therefore based on the measured data from the Cascina Canta rice farm, which harvests 4.4 t/ha of organic dried paddy and information obtained from CRR Ente Risi, integrated with the available literature data. No pesticide or related field operations were accounted for. Fertilization was considered to be performed only using organic fertilizers (Tinker, 2001), in compliance with legislative prescriptions (Table 8). Thus, "solid manure loading and spreading" from Ecoinvent was used in the model.

The variations in the methane and nitrous oxide emissions were estimated using the IPCC models (IPCC, 2006). In comparison to traditional farming, CH_4 and N_2O emissions per hectare decreased by 7% and 31%, respectively. However, due to the lower grain yield, the emission of methane per kg of paddy rice increased by 29%, while the reduction of nitrous oxide was limited to −5%. A crop-in-crop rotation was considered and thus the rice cultivation is substituted by soy, every three years. However, a crop-in-crop rotation does not affect the LCA model for organic rice, due to the fact that soy is allocated its own environmental burdens. The capital goods were re-allocated according to the grain yield.

4.2. Upland rice

Upland rice is rice cultivated without submersion and grown under a reduced water regime. Watering is therefore provided either by rainfall or by artificial irrigation. As there is not sufficient rainfall in the study area, two irrigation systems were considered: furrow and sprinkler irrigation. However, as upland rice cultivation in Vercelli is still in an experimental phase, all the data were relevant to experimental fields (Callegarin, 2000; Greppi et al., 2007; Moletti, 1989; 1990).

The absence of submersion required some changes in the LCA. Nitrogen fertilization and pesticide use were increased by 20% (Moletti, 1990), while potassium and phosphorus remained unchanged. Emissions of CH_4 were reduced to 2 g/kg of paddy rice and N_2O was increased to 0.29 g/kg, according to the IPCC model.

Table 8
LCI of organic rice: model entries for fertilizers.

Ecoinvent entry	Quantity (kg/ha)
Poultry manure, dried	1 000
Horn meal (Cornunghia)	366
Potassium sulphate	315

The grain yield was reduced by 25% for both furrow and sprinkler irrigation (Callegarin, 2000). Water use was set at 5600 m^3/ha for furrow irrigation and 2700 m^3/ha for sprinkler irrigation (Callegarin, 2000). The "Irrigating" process from Ecoinvent was used in the model, according to the water quantity utilised for sprinkler irrigation.

4.3. Parboiled rice

Parboiled rice is the rice that has been boiled in the husk. Among other advantages, parboiling improves milling yield, storability and nutritional content (Roy et al., 2007). Before milling, dried paddy undergoes a soaking process followed by pressure steaming and drying (Carpi et al., 1992). The average data for the LCA model of parboiled rice were obtained from Riso Gallo SpA.

Rice parboiling consumes 1.57 MJ/kg or, assuming a 70% efficiency, 0.061 m^3 of natural gas. This consumption can be ascribed to the drying process (70%) and to steam production. All the other processes consume electricity (0.364 MJ/kg). The water use for soaking and steaming is about 1 l/kg. The treatment of wastewater after parboiling (0.75 l/kg with a COD – Chemical Oxygen Demand of 2200 mg/l) was also included in the LCA model.

Allocation between the co-products after parboiling and milling had to be adapted. In comparison to white milled rice, the percentage of by-products is smaller (35% of the total production) and the average selling price of parboiled rice is higher (0.6 €/kg). Therefore, 93% of impact was allocated to parboiled rice.

5. Results and discussion

5.1. Impact assessment (white milled rice)

The impact assessment phase was carried out by analysing the results of the inventory (LCI) in order to calculate the category indicators described in Section 2. Table 9 shows the indicators relevant to the LCA model for white milled rice.

As it can be seen in Table 9, the production and delivery of 1 kg of exported white milled rice from the 50 ha rice farm require 17.8 MJ of energy resources of which 16.6 MJ are non-renewable. The GWP_{100} indicator shows a carbon dioxide equivalent emission of 2.9 kg, which seems to be in contrast with the value of 1.1 kg CO_2eq reported in the Italian Greenhouse Gas Inventory (APAT, 2005). However, the difference can be explained in terms of life cycle phases and system boundaries. In fact, the greenhouse emission of rice, according to APAT (2005), corresponds to direct methane emissions from the paddy field: 48 g of CH_4 multiplied by a characterisation factor of 23. However, when adding up direct and indirect greenhouse emissions relevant to the subsequent life cycle steps and when considering the loss of weight after drying, as well as when allocating impacts between the refined rice and its by-products, the GWP indicator rises to almost 3 kg of CO_2eq per kg of delivered white milled rice. This result highlights the great potential of LCA, which can take into account a large number of parameters and can help manage the complexity of a production system where both natural and industrial processes co-exist.

Although LCA can help manage life cycle issues concerning rice production, it should be pointed out that the economic allocation, which assigned 91% of impact to white milled rice, could be perceived as an issue which disadvantages refined rice in comparison to its by-products. A simple mass allocation criterion would have assigned 62% of the total impact to white milled rice. However, in that case, white milled rice would have been assigned the same impacts as rice husk, which seems to be highly unlikely. The adopted allocation criteria must take into account the relative quality of the co-products and the economic value does fulfil this requirement.

Table 9
Impact indicators for 1 kg of delivered white milled rice (absolute values for 50 ha farm and % of variation for 150 and 300 ha).

Impact indicator	Unit	Local distribution			Exported rice		
		50 ha	150 ha	300 ha	50 ha	150 ha	300 ha
GER	MJ	15.72	(−2.1%)	(−3.0%)	17.81	(−1.9%)	(−2.7%)
NRER	MJ	14.59	(−2.1%)	(−3.0%)	16.64	(−1.8%)	(−2.6%)
GWP_{100}	kg CO_2eq	2.76	(−0.6%)	(−0.8%)	2.88	(−0.5%)	(−0.8%)
ODP	mg CFC11eq	0.10	(−1.4%)	(−2.1%)	0.12	(−1.2%)	(−1.8%)
AP	mol H^+	0.25	(−0.9%)	(−1.3%)	0.28	(−0.8%)	(−1.2%)
EP	g O_2eq	328.3	(−0.2%)	(−0.2%)	334.7	(−0.2%)	(−0.2%)
POCP	g C_2H_4eq	0.52	(−0.1%)	(−0.1%)	0.53	(−0.1%)	(−0.1%)
WU_t	m^3	8.0	(−2.3%)	(−3.2%)	8.2	(−2.3%)	(−3.1%)
WU_d	m^3	4.9	(0.0%)	(0.0%)	4.9	(0.0%)	(0.0%)

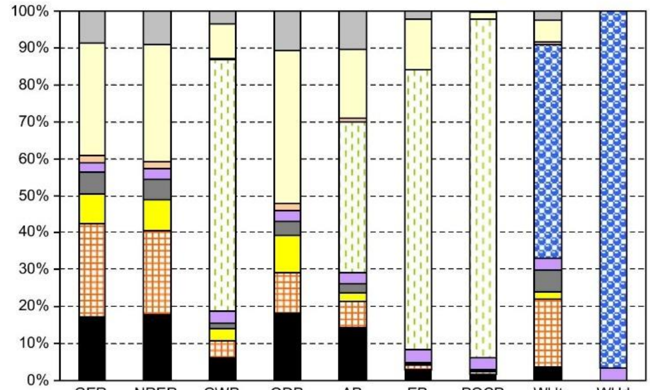
The direct use of water for irrigation appears to be particularly intense: almost 5 m^3 per 1 kg of delivered rice. However, if the indirect use of fresh water is also considered, the WU_t indicator would rise to around 8 m^3/kg. The result is not far away from that reported in Oki et al. (2003) which have estimated the "irrigation water requirement" of rice in Japan. According to an estimate based on Japanese agricultural withdrawals, water rights and national rice production, the unit water requirement for irrigation is 5 m^3/kg. According to another estimate by the same authors, based on a paddy field model which considered a 15 000 m^3/ha water consumption, a 6.46 t/ha grain yield and a 65% milling yield, the unit water requirement is 3.6 m^3/kg.

Table 9 is useful to analyse the influence of farm size over the life cycle impacts of rice. In recent years, the average size of farms has in fact increased in order to reduce costs (Ente Risi, 2007). However, although Berruto and Busato (2007) have reported that an increase in farm size from 50 to 150 ha can be effective in reducing over-mechanisation and therefore in reducing pre-harvesting costs by 20%, the benefits are much lower in terms of life cycle impacts. When a farm increases from 50 to 300 ha, the environmental gain is around 3% of the energy resources and total water use, as a consequence of the better utilisation of capital goods in the agricultural phase. With the exception of stratospheric ozone depletion, which has an improvement of around 2%, all the other indicators show an improvement of about 1% or lower.

5.2. Contribution analysis (white milled rice)

A contribution analysis has been carried out with reference to exported white milled rice.

Fig. 2 can be helpful to appraise the role and significance of the different subsystems that contribute to the life cycle impacts of rice. From this figure it is possible to understand more about the consequences of the assumptions that were made in the inventory phase. Fertilizers' production is the greatest contributor to the gross energy requirement (30%) and this is followed by refining and



	GER	NRER	GWP	ODP	AP	EP	POCP	WUt	WUd
Field operations	8.9%	9.2%	3.6%	10.8%	10.3%	2.2%	0.5%	2.6%	0.0%
Fertilizers	30.2%	31.5%	9.2%	41.3%	18.9%	13.6%	1.7%	5.9%	0.0%
Pesticides	2.0%	2.1%	0.4%	1.9%	0.8%	0.1%	0.1%	0.6%	0.0%
Irrigation	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	57.9%	96.7%
Field emissions	0.0%	0.0%	68.0%	0.0%	40.7%	75.9%	91.5%	0.0%	0.0%
Seed production	2.7%	2.8%	3.2%	3.1%	3.1%	3.3%	3.3%	3.1%	3.3%
Capital goods	5.6%	5.5%	1.6%	3.9%	2.4%	0.4%	0.2%	6.0%	0.0%
Drying & storing	8.2%	8.6%	3.2%	9.8%	2.4%	0.3%	0.6%	2.1%	0.0%
Refining & packing	25.3%	22.5%	4.7%	10.9%	7.2%	1.5%	0.7%	18.3%	0.0%
Transportation	17.0%	17.9%	6.1%	18.3%	14.2%	2.8%	1.5%	3.5%	0.0%

Fig. 2. Contribution of subsystems to the impacts of exported white milled rice (50 ha farm).

packing (25%) and transportation (17%). Global warming is mainly influenced by field emissions (68%) and then by fertilizers (9%) and transportation (6%). Paddy field emissions have the greatest impact on four indicators (GWP, AP, EP, POCP), thus emphasising the need for further reliable and site specific data. As expected, direct water use is dominated by irrigation (97%), the remaining 3% being used for seed production. The total water use is also dominated by irrigation, but 18% is indirectly used for the production of packaging materials.

The agricultural phase has generally shown the most important contributions to the final impacts, thus representing an environmental hot spot. Nevertheless, the post-harvest processing showed remarkable contributions, therefore identifying further areas of potential improvement, mainly in terms of energy saving and reduction of the ozone depletion and acidification potentials.

As far as transportation is concerned, it should be noticed that there is a remarkable contribution for energy and ODP (17–18%), a contribution which is lower for GWP (6%) and negligible for the remaining indicators.

As the contribution of capital goods was considered a meaningful issue, it should be mentioned that they have a noticeable weight (6%) on energy requirement and WU_t, the contribution to ODP, AP and GWP being 3.9%, 2.4% and 1.6%, respectively. The contribution to EP and POCP is less than 1%.

5.3. Scenario analysis (organic, upland and parboiled)

The results obtained from the LCA modelling of the alternative rice production systems are shown in Fig. 3, where the indicators are compared with the exported white milled rice scenario.

As far as organic farming is concerned, it should be mentioned that the benefits that arise from the avoided use of fertilizers and chemicals are heavily reduced or, for some indicators, cancelled due to the lower grain yield. This result is similar to the conclusions of Audsley et al. (1997) and Williams et al. (2005), for organic wheat, who reported that the lower burdens per hectare corresponded to higher burdens per unit mass of product. Thus, while 1 kg of organic rice allows a 5% saving of energy resources, a 15% saving of ODP and a 10% saving of EP, the remaining indicators show a deterioration. GWP and AP indicators are in fact increased by 20%, while the POCP and total water use rise by 30%. The increased land use, as a consequence of the lower yields, adds further environmental burdens relevant to organic farming. On the contrary, the avoided use of pesticides can assign an environmental benefit to organic farming in terms of lower loss of bio-diversity.

The LCA models for upland rice showed a dramatic improvement in terms of direct water use: around 1 m^3/kg for sprinkler and 2 m^3/kg for furrow irrigation. However, if direct and indirect water uses are considered together, furrow irrigation shows an improvement of almost 30%, in comparison to the baseline scenario, while sprinkler irrigation shows a 5% higher water consumption. The reduction in the GWP is about 50% for both furrow and sprinkler, while the POCP drops to 11–12%. On the contrary, there is a 23% increase in energy use for furrow and 50% for sprinkler, while the ODP, AP and EP are increased by 17–33%.

Finally, an analysis of the parboiled scenario showed a 16% increase in the energy requirement (GER), in comparison to the baseline scenario, caused by steam production and re-drying. A slight reduction in direct water use was recorded (2%), although this may seem unlikely, as parboiling adds more process water. However, this can be explained in terms of a higher milling yield after parboiling. The ODP increased by 26%, while a slight increase was recorded for GWP and WU_t. The remaining indicators EP, AP and POCP were unchanged. As far as allocation is concerned, it should be noticed that an economic partitioning could be regarded as a criterion which disadvantages parboiled rice, in comparison with white milled rice, due to the higher market value of parboiled rice.

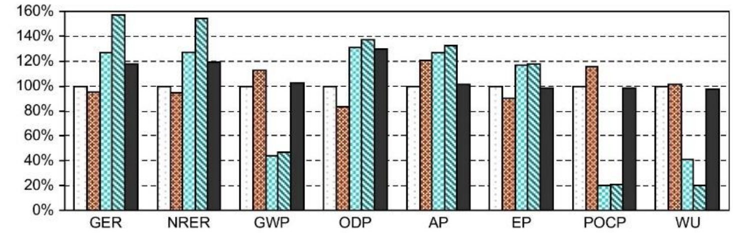
5.4. Data uncertainty and sensitivity analysis

Williams et al. (2005) reported that errors in the measurement of single emissions such as N_2O can be in the range of 70%, while errors in the national inventories of gaseous emissions from agriculture are usually 30%. According to the same authors, a reasonable estimate of the uncertainty, associated with any calculated burden from an LCA model relevant to the agri-food chain, is 30–34%.

Based on these remarks, a sensitivity analysis was carried out in order to test the robustness of the LCA model. Thus, the input data with high uncertainty and some of the assumptions relevant to allocation were re-considered, also taking into account the hot spots highlighted by the contribution analysis.

The sensitivity analysis was limited to the white milled rice scenario because, as reported in Williams et al. (2005), despite the effects of uncertainty on the absolute accuracy, LCA modelling is relatively accurate at performing comparative analyses. Uncertainty is in fact highly correlated between scenarios, thus comparative differences are largely a consequence of differences between systems.

Bearing this in mind, the direct CH_4 and N_2O emissions were substituted by the measured data from a paddy field in Vercelli, recorded in the year 1999 and reported in Leip et al. (2007): 72.4 g/kg of CH_4 and 0.34 g/kg of N_2O for early flooding or 17.5 g/kg of CH_4



	GER	NRER	GWP	ODP	AP	EP	POCP	WU
White milled	100%	100%	100%	100%	100%	100%	100%	100%
Organic	96%	94%	113%	84%	120%	90%	116%	101%
Upland (furrow irrigation)	127%	127%	44%	131%	127%	117%	20%	41%
Upland (sprinkler irrigation)	157%	154%	47%	137%	133%	118%	21%	20%
Parboiled	118%	119%	103%	130%	101%	98%	99%	98%

Fig. 3. Relative comparison between alternative rice farming and processing scenarios.

Table 10
Results of sensitivity analysis: percentage of variation compared to the baseline scenario for white milled rice.

Impact indicator	CH_4 and N_2O emissions according to Leip et al. (2007)		Low level of mechanization	Allocating straw	Low water requirement 15 000 m^3/ha	High water requirement 40 000 m^3/ha
	Early flooding	Late flooding				
GER	–	–	−3%	−5%	–	–
NRER	–	–	−3%	−5%	–	–
GWP_{100}	36%	−34%	−1%	−9%	–	–
ODP	–	–	−2%	−6%	–	–
AP	–	–	−1%	−8%	–	–
EP	–	–	<1%	−10%	–	–
POCP	–	–	<1%	−10%	–	–
WU_t	–	–	−3%	−8%	−27%	27%
WU_d	–	–	–	−10%	−46%	46%

and 0.68 g/kg of N_2O for late flooding. Although the contribution of capital goods and the influence of farm size had already been pointed out, the quantity of agricultural machinery shown in Table 5 has been halved. The economic allocation of burdens to straw was re-considered by assuming a production of 7 t/ha and a consequent sale to an electric power station at the current market price of 30 €/t. Finally, water intake for irrigation was set at 15 000 and 40 000 m^3/ha, according to Allavena and Monti (2007). The model was then re-calculated and the variations are reported in Table 10 and compared with the baseline results.

Table 10 shows that the change in field emissions affected climate change emissions by −34/+36%. The machinery reduction had a limited effect on the energy use and total water requirement of −3%. The maximum variation caused by allocating burdens to straw was −10%. The change in water for irrigation affected the total water requirement by −27/+27% and direct water requirement by −46/+46%.

6. Conclusions

Modelling the life cycle of rice is a demanding task which involves a large number of agricultural and industrial processes, while requiring a multi-disciplinary research team and a methodology that is able to handle and integrate the findings of different investigations.

Although improvements and further research are necessary, mainly relevant to direct field emissions of CH_4 and N_2O, as well as to the use and re-use of water, the present research has supplied quantitative results and information that might be useful in future investigations.

The energetic and environmental profile of white milled rice has been obtained, with reference to a typical farm operating in the Vercelli rice district, considering average farming practices, average grain yield, typical rice processing, packaging and delivery. In a from-cradle-to-gate perspective, the production and delivery of 1 kg of exported white milled rice consume 17.8 MJ of primary energy, produce 2.9 kg of carbon dioxide and use about 4.9 m^3 of water for irrigation.

As far as the over-mechanisation of rice farms in the Vercelli district is concerned, the general increase of farm size goes towards an environmentally friendly direction, due to the lower impact of capital goods. Nevertheless, the benefits are limited to energy use and total water consumption (both 3%).

An analysis of improvement scenarios has shown that mitigation solutions cannot be restricted to single life steps or limited aspects, since the consequences on the subsequent life phases could dramatically reduce the improvements or even cancel them. This is the case of organic farming, which has the potential of lowering the impacts per unit of cultivated area, but which supplies a final product that is characterised by heavier impacts (+20% of GWP) due to the lower production yield.

Upland rice is effective in reducing climate change emissions by 50%, but only upland rice together with furrow irrigation is effective in reducing the total use of water (−30%). This would allow rice cultivation to be continued in the Vercelli district in the case of a different future allocation of water resources.

The LCA model for parboiling has emphasised the role and limits of allocation, as well as the influence of choices relevant to the functional unit. These should be dealt with in more detail in future investigations, while also taking into account the nutritional content of rice products.

In conclusion, the authors would welcome a wider use of the life cycle approach in Italian agri-food chains, as it can supply sound scientific results that can also be used for transparent and credible communication between suppliers and the final consumers.

Acknowledgements

The authors would like to thank the anonymous reviewers for their valuable suggestions and Dr. Deborah Shields of Colorado State University for her help in editing. A.A. Palestro and Cascina Canta rice farms, Riso Gallo and Riseria Re Carlo rice factories are gratefully acknowledged for supplying data. The authors also would like to thank Dr. Maurizio Tabacchi of CRR Ente Risi and Dr. A. Finassi for the stimulating discussions and helpful comments relevant to the LCA research.

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7. ANNEX B: ECO-EFFICIENT WASTE GLASS RECYCLING: INTEGRATED WASTE MANAGEMENT AND GREEN PRODUCT DEVELOPMENT THROUGH LCA.

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Waste Management xxx (2011) xxx–xxx

Contents lists available at SciVerse ScienceDirect

Waste Management

journal homepage: www.elsevier.com/locate/wasman

Eco-efficient waste glass recycling: Integrated waste management and green product development through LCA

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ARTICLE INFO

Article history:
Received 8 July 2011
Accepted 19 October 2011
Available online xxxx

Keywords:
Eco-efficiency
Recycling
Life cycle assessment
Green products
Industrial symbiosis
Waste glass
Foam glass

ABSTRACT

As part of the EU Life + NOVEDI project, a new eco-efficient recycling route has been implemented to maximise resources and energy recovery from post-consumer waste glass, through integrated waste management and industrial production. Life cycle assessment (LCA) has been used to identify engineering solutions to sustainability during the development of green building products. The new process and the related LCA are framed within a meaningful case of industrial symbiosis, where multiple waste streams are utilised in a multi-output industrial process. The input is a mix of rejected waste glass from conventional container glass recycling and waste special glass such as monitor glass, bulbs and glass fibres. The green building product is a recycled foam glass (RFG) to be used in high efficiency thermally insulating and lightweight concrete. The environmental gains have been contrasted against induced impacts and improvements have been proposed. Recovered co-products, such as glass fragments/powders, plastics and metals, correspond to environmental gains that are higher than those related to landfill avoidance, whereas the latter is cancelled due to increased transportation distances. In accordance to an eco-efficiency principle, it has been highlighted that recourse to highly energy intensive recycling should be limited to waste that cannot be closed-loop recycled.

1. Introduction

Due to the fact that many environmental and resource management issues faced by developed and developing nations alike are becoming highly uncertain, urgent, complex, and interconnected, we can no longer afford to address individual environmental and social problems in a convenient isolation of their context, or their spatial or temporal scale (Funtowicz and Ravetz, 2001).

Sustainable development (Bruntland, 1987) is both comprehensive and flexible, thus providing a framework for addressing complex problems through shared roles and responsibilities among the society as a whole and socially responsible companies (European Commission, 2001b; Shields et al., 2002). In such a framework, a growing number of companies are incorporating environmental sustainability in their business strategies, in order to integrate environmental concerns in their business operations and in their interactions with stakeholders (Albino et al., 2009; WBCSD, 1998).

Albino et al. (2009) have discussed the reasons that push firms to go "green", classifying them into three categories: legitimacy, competitiveness, and social responsibility. The authors also pointed out the strategic role of green products (i.e. goods or services that minimise their environmental impact over the whole life cycle) for sustainability-driven companies.

Due to the fact that environmental policies, especially at the EU level, are increasingly focusing on products, the attention of corporate environmental management has been shifting from processes (e.g. clean technologies) to products. The key role of green products in moving towards a 'new growth paradigm and a higher quality of life, through wealth creation and competitiveness' is clearly emphasised in the Green Paper on Integrated Product Policy (European Commission, 2001a). Product-oriented environmental policies offer at least two advantages: (1) they raise awareness that production is not the only source of environmental burdens, but rather production, consumption and post consumption play equally important and inter-dependent roles; (2) they foster shared responsibilities and roles between producers and their customers.

A general definition of a sustainable product could be: a product designed, manufactured, used and disposed of according to criteria

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doi:10.1016/j.wasman.2011.10.018

Please cite this article in press as: Blengini, G.A., et al. Eco-efficient waste glass recycling: Integrated waste management and green product development through LCA. Waste Management (2011), doi:10.1016/j.wasman.2011.10.018

of economic, environmental and social efficiency, which maximise net benefits across generations. However, it should be mentioned that there is still much confusion about what can be considered a sustainable product and what should not. Moreover, the terms "green", "eco" and "sustainable" are often used inter-changeably.

While a comprehensive classification of green products and the definition of their role within environmental policies, strategies and goals are out of the scope of this paper, it should be remarked that the following key issues are currently becoming commonly accepted (Blengini and Shields, 2010):

(1) Companies are increasingly using green product innovation in order to fulfil the environmental quality expectations expressed by their eco-responsible customers.
(2) Companies need reliable tools to make their environmental claims credible and distinguish themselves from firms which merely pursue market targets with green-wash packaging or advertising.
(3) Environmental burdens should be assessed and subsequently minimised throughout the whole product life cycle.
(4) Life cycle assessment (LCA) is one of the most important analytical tools to provide the scientific background for engineering solutions to sustainability, both during the design phase (eco-design) and during life cycle management.

In the above described context of green product development, a new recycling route has been implemented with the goal of maximising resources and energy recovery from post-consumer waste glass through integrated waste management and industrial production.

Life cycle assessment (LCA) has been used to highlight and quantify the eco-efficiency of such an innovative waste-to-production chain, with the objective of identifying engineering solutions to sustainability during the development of new building products to be used in energy efficient buildings (Blengini and Di Carlo, 2010). The purpose of applying LCA in this instance is threefold: quantifying environmental and energy savings and impacts, improving eco-efficiency and, finally, increasing the credibility of sustainability claims.

Both the new process and the related LCA have meaningful aspects that deserve discussion, as they are framed within a case of industrial symbiosis, where multiple waste streams are utilised as input in a multi-output industrial process. In other terms, the waste-to-recycling system under analysis can be considered a hybrid waste management – production system.

The green building product under development is a recycled foam glass (RFG). However, unlike current foam glass products, such as those described in Hurley (2003) and Scarinci et al. (2005), where waste glass is recycled in an open-loop fashion through a energy intensive process, the new waste-to-production route is based on a general eco-efficiency principle according to which RFG should be produced from the part of waste glass that cannot be closed-loop recycled. Hurley (2003) argued that cullet from glass packaging waste should be used in foam glass production only if it is heavily contaminated and not suitable for containers. Accordingly, in the recycling route that will be described in the next chapter, most of post-consumer container glass is not converted into RFG, but rather recovered, purified and sent back to the industries from which the waste originated, increasing eco-efficiency.

2. Materials and methods

The principal output from the innovative recycling route implemented under the EU Life + NOVEDI project by the Italian company SASIL (2009) is RFG, which is an artificial aggregate manufactured from waste glass (Fig. 1).

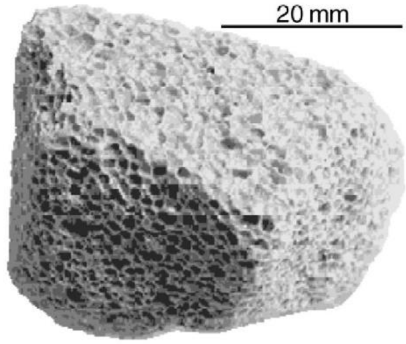

Fig. 1. Recycled foam glass (RFG).

However, the focus of the present paper is not RFG as material, a topic that has been deeply investigated and reported on in the literature (Bernardo and Albertini, 2006; Bernardo et al., 2010, 2005; Fernandes et al., 2009; Herat, 2008; Hurley, 2003; Lebullenger et al., 2010; Méar et al., 2005; Scarinci et al., 2005; Yot and Méar, 2011). Rather the focus is on gaining a better understanding of the eco-efficiency of the proposed recycling route, and the way LCA can be used to measure eco-efficiency and support green product development.

Table 1 reports the main physico-mechanical properties of the RFG that SASIL currently produces as a loose aggregate, which is graded in two ranges of particle size (0–8 mm and 8–16 mm) and shows a good mechanical strength (crushing test 0.62–5.2 N/mm^2 according to the Standard UNI EN 13055-1). Table 1 also shows a selection of mineral based insulating materials for which LCAs are publicly available (Ecoinvent, 2007; Lavagna, 2008; Pittsburgh Corning Europe, 2007). It must be said that, due to commercial confidentiality and limited information in the public domain, there is little availability of detailed and transparent LCAs of foam glass products.

This RFG is intended to be used for several applications in the building sector, where energy saving and resource efficiency are regarded as key issues. Thanks to the combination of low density, low thermal conductivity and good mechanical strength, RFG can be employed in light-weight concrete with good thermal-insulating properties. These RFG-based concrete products, which are presently under testing in co-operation with SASIL SpA, Italcementi Group and the Politecnico di Torino, are expected to open the way towards new engineering solutions for energy efficient buildings. One of these end-uses is represented by mono-material building envelopes that, beyond enhancing energy saving during the operational phase of buildings, are expected to increase recyclability of the building as a whole.

The production of foam glass dates back to the 1930s. It was originally manufactured from a specially formulated glass composition using virgin glass only. Since then, foam glass producers have steadily increased the quantity of post consumer waste glass in their product up to 98% (Hurley, 2003; Scarinci et al., 2005). Some of these RFGs are currently traded as green building products and their environmental claims are often self-declarations based on their status of recycled materials, which avoid waste landfilling and save non-renewable resources.

A granular RFG similar to the one presented in this paper was produced in Switzerland in the 1980s by Misapor AG (www.misapor.ch) in order to provide an alternative market outlet for

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Table 1
Characteristics of the RFG and other mineral-based insulating materials.

		RFG	Expanded clay	Expanded perlite	Expanded vermiculite	Foam glass
Type of product		Granulate	Granulate	Granulate	Granulate	Slabs
Density (slab)	kg/m^3	210–240	–	–	–	110–165
Density (bulk)	kg/m^3	160–180	260–500	80–130	70–140	–
Thermal conductivity	W/(Km)	0.07–0.09	0.08–0.13	0.04–0.07	0.06–0.08	0.04–0.06
Temperature of expansion	°C	900	1200	870–1090	1250–1500	850–1250

separately collected waste glass. Although the foaming process adopted by SASIL is similar to the one developed by Misapor AG (i.e. continuous production of sheets of foam glass that are then broken into loose foam glass aggregate and sized), the principal difference is the sorting process. Unlike most of RFGs, where recycling into building products represents an alternative to closed-loop recycling, the new RFG is produced from a by-product of conventional closed loop recycling, i.e. the waste glass that is rejected by container glass recyclers after their sorting process (between 6% and 15% in Italy), and which is enriched in contaminants (mainly metallic, ceramics and plastic scraps). The principal route for post-consumer soda-lime glass from municipal solid waste separate collection therefore remains recycling into container glass.

Since such a rejected glass is made mostly of glass and ceramic fragments/powders and also contains metals and plastic scraps, SASIL has developed an innovative recycling process that allows the separation of waste streams and makes it possible to sell purified materials back to the industries from which they were generated. RFG is therefore produced from the waste that has previously been rejected by container glass recyclers and, after the sorting step, is not used in glass, ceramic, brick or metal works, or recovered in waste-to-energy facilities.

In a mix with soda-lime glass, SASIL also uses special glass, mostly composed of monitor glass, glass fibres, and glass containing heavy metals that are presently landfilled. Due to the present shift of technologies in the TV/PC monitor industry (plasma, LCD), the problem of CRT (cathode-ray tube) glass disposal is in fact of particular concern. Alternatives for recycling have been reported in the literature (Dondi et al., 2009; Méar et al., 2006; Menad, 1999), including use of CRT glass in RFG production (Bernardo and Albertini, 2006; Bernardo et al., 2005; Méar et al., 2005). Particular attention has been paid to the resistance of RFG to possible leaching of barium and lead, typically contained in the front (Ba) and in the back (Pb) of the screen (Musson et al., 2000; Yamashita et al., 2010; Yot and Méar, 2011). Leaching tests run according to the standards UNI 10802:2004 and UNI EN 12457-2:2004 showed that the potential releases are below the thresholds, being 3–15 μg/l in case of lead and 20–40 μg/l for barium, which is in accordance with the findings of Bernardo et al. (2005).

As far as the environmental management of the recycling route and the role of LCA are concerned, the first environmental gain is the avoidance of waste landfilling, which leads to a saving in terms of waste dump space: a scarce resource nowadays in a country such as Italy. At the same time, sorting of waste glass prior to RFG production allows for the separation and recovery of glass, ceramic, metal and plastic, which improve the overall eco-efficiency.

All the above listed environmental gains are clearly perceived by SASIL, who intends to use them as the basis for environmental claims and green marketing. But, the open question is how the sustainability performance of the new green product can be quantified and communicated in a credible way?

Beyond the environmental gains, recycling is responsible for environmental impacts due to use of additives and fossil fuel in an energy intensive thermal process and it might increase transport-related impacts. Moreover, selection and material recovery efficiencies can lead to lower the overall efficiency of the process (Rigamonti et al., 2009). Consequently, induced impacts might outweigh environmental gains, thus rendering self-declared environmental claims less credible, or even false (Blengini and Garbarino, 2010).

From the previous discussion, it clearly emerges that the environmental profile (eco-profile) of RFG is the final result of a complex and inter-dependent waste-to-production system. LCA is therefore used to outline the eco-profile of RFG, because it is an analytical tool able of capturing complexity and inter-dependencies.

2.1. Definition of a LCA methodology to support environmental claims of RFG

The LCA methodology according to ISO 14040 (2006) and ISO 14044 (2006) has been used in order to capture the multiple environmental gains and the environmental impacts of the waste-to-recycling system under analysis.

The main advantage of using LCA is the possibility of assessing the environmental performance of products throughout their life cycle with a comprehensive perspective. However, there are some specific aspects that must be considered when dealing with waste management (Ekvall et al., 2007; Martínez-Blanco et al., 2010). In fact, when applying LCA to waste management systems, the from-cradle-to-grave and from-cradle-to-gate philosophies, typically adopted when dealing with production systems, must be turned into from-gate-to-grave or sometimes from-gate-to-cradle. In fact, as far as waste management is concerned, the input material is waste, which can either be sent to landfill or re-enter further life cycles in substitution of virgin materials.

Substitution means avoidance of products manufactured from primary resources through secondary materials gathered from recovery and recycling. In other terms, the production of a recycled material that re-enters further life cycles represents a potential credit for avoiding the production of an equivalent quantity of virgin products. The system that recycles the waste into a valuable product is credited with the environmental burdens of the corresponding primary production, but is charged with energy and ancillary materials used for the recycling process. This is currently called system expansion (Finnveden, 1999).

In this research, system expansion has been used in order to associate the multiple benefits of the new recycling process to RFG. Thus, net environmental gains relevant to glass and ceramic fragments/powders recovery, metal scrap recycling, plastic scrap energy recovery and landfill avoidance were allocated to RFG. This is a way to make visible the benefits achieved through industrial symbiosis: one system transfers environmental gains to the other.

2.2. Selection of environmental indicators

Bearing in mind that the choice of indicators and methodologies to express the results of an LCA is a subjective step, the research partners acknowledged the importance of supporting the environmental sustainability claims with sound, objective, understandable and internationally recognised LCA indicators.

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A combined top-down and bottom-up approach has been used in order to adopt a meaningful set of indicators. A top-down approach can be described as one that selects indicators that are representative of broadly recognised areas of environmental concern, as well as based on various international conventions, agreements and guidelines. This type of approach is indeed consistent with the recommendations of ISO (2006). In contrast, a bottom-up approach can be defined as one that identifies indicators based on the interests of the industry, public administrators or stakeholders and/or data availability.

This said, a first set of five mid-point indicators has been identified according to the above mentioned top-down approach (Boustead and Hancock, 1979; SEMC, 2000): Gross Energy Requirement (GER), Global Warming Potential (GWP), Acidification Potential (AP), Eutrophication Potential (EP) and Photochemical Ozone Creation Potential (POCP).

According to the bottom-up approach, in order to further assist decision makers with a simplified overall judgement across areas of environmental concern, the above mid-point indicators were complemented with the Eco-Indicator 99 H/A (Goedkoop and Spriensma, 1999). The latter is based on the so called damage-oriented (end-point) approach, and is aimed at evaluating the environmental implications for human health, ecosystem quality and depletion of non-renewable resources. It must be remarked that, although worldwide used, Eco-Indicator 99 involves both physical and social aspects and introduces subjective value choices and uncertainties that render it not fully consistent with the recommendations of ISO.

2.3. Functional unit, system boundaries and data sources

According to ISO 14040, the functional unit (FU) is a quantified description of product systems' performances. In the case of RFG, the selected FU should provide a reference to which the inputs and outputs can be related, thus allowing comparison among final products. However, RFG-based products are still under development and thus, at this stage, the analysis must be restricted to the environmental profile of RFG as granular material. Consequently, the adopted FU is 1 tonne of RFG aggregate.

The selected FU allows outlining the eco-profile of RFG as material, which will be the necessary background knowledge for subsequent comparative LCAs of building products or engineering solutions to energy saving in the built-environment. Nevertheless, a rough comparison can be based on thermal insulation properties of RFG against other building materials. An example is reported in Ardente et al. (2008) where the chosen functional unit was the mass unit of insulating board with a given thermal resistance R (measured in m^2K/W). It must be remarked that, in a full life cycle perspective, the selected FU would likely emphasise the use phase of buildings insulated with RFG, while it might underestimate some important RFG peculiarities such as lightness, fast assembly and recyclability.

With reference to the insulating materials reported in Table 1, it should also be said that a direct comparison with RFG granulate might be partially (or totally) misleading. In fact, RFG, expanded clay, expanded perlite and expanded vermiculite are granular insulating materials that might be used inter-changeably for some, but definitely not for all, possible end-uses. For instance, as far as concrete is concerned, expanded perlite and expanded vermiculite, which show good thermal insulating performance, have unacceptably low mechanical strength. On the contrary, expanded clay, which shows a mechanical strength similar to RFG, has higher thermal conductivity. Finally, the foam glass (last column of Table 1) that is described in the Ecoinvent database (2007) shows some important analogies with RFG, the most evident of which is that it is produced using nearly 70% of post-consumer glass. However, it is produced with a different process and traded in slabs and pre-cut shapes (Pittsburgh Corning Europe, 2007), therefore intended for different end-uses. With these limitations in mind, a comparison in terms of environmental indicators will be given in the next chapter.

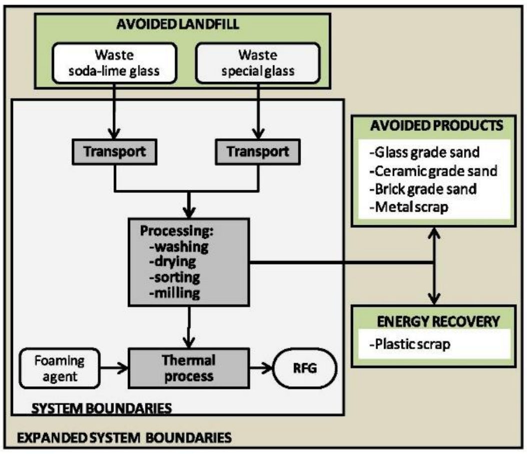

Fig. 2. System boundaries of the RFG waste-to-production chain.

As system boundaries are concerned, these encompass (Fig. 2): waste glass collection and transportation, processing (i.e. washing, sorting, drying and milling), production of additives and the thermal process.

As will be described in the inventory analysis, in the LCA model waste glass collection corresponds to landfill avoidance, while sorting corresponds to the recovery of commercial granulate materials, which in turn displace primary production of silica sand (glass grade sand and ceramic grade sand) and clay (brick grade sand). Moreover, sorting corresponds to avoided production of primary metals and fossil fuels.

As far as data sources are concerned, primary data were supplied by SASIL SpA, while secondary data were retrieved from the database Ecoinvent 2 (2007). The LCA model has been implemented with the software SimaPro (PRé Consultants, 2006).

2.4. Inventory analysis

RFG production can be schematically divided into the macro-processes shown in Fig. 2. Waste soda-lime glass and waste special glass are collected and transported to the plant (waste glass collection) where glass is separated from other components and transformed into a powder through a sequence of processes (washing, sorting, drying and milling). RFG grade powder is then mixed with a foaming agent prior to undergoing a thermal process where the foam glass expands.

The sequence of industrial processes (or avoided processes) in the waste-to-production chain is briefly described in the following paragraphs, with emphasis on input data used for LCA modelling. Unless otherwise specified, data refer to 1 t of input glass waste.

2.4.1. Glass landfilling (avoided)

Environmental gains related to landfill avoidance depend on the type and origin of waste glass, which can be grouped into two main categories:

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Table 2
Classification and average composition of waste glass used for RFG production.

EWC code	Glass type	Description	%
19 12 05	Soda-lime	Glass (waste glass rejected by post-consumer glass recyclers)	90.1
10 11 03		Waste glass-based fibrous materials	3.8
10 11 12		Waste glass other than those mentioned in 10 11 11	3.3
10 11 10		Waste preparation mixture before thermal processing, other than those mentioned in 10 11 09	0.8
20 01 02	Special glass	Glass (post-consumer special glass)	0.5
6 05 03		Sludge from on-site effluent treatment other than those mentioned in 06 05 02	0.3
10 09 08		Casting cores and moulds which have undergone pouring other than those mentioned in 10 09 07	0.4
10 09 12		Other particulates other than those mentioned in 10 09 11	0.8

– Soda-lime glass; including flat glass, windshield glass, light bulbs, tableware and containers.
– Special glass; glass with different compositions used for special applications such as coloured glass, tempered glass, hard glass, laminated glass, UV glass, fibre glass, optical fibres and various physical, chemical and industrial applications.

Table 2 shows the origin and the percentage of waste glass collected by SASIL according to the European Waste Catalogue (EWC). Soda-lime glass (EWC 19 12 05) refers to waste glass that is rejected by container glass recyclers after their sorting process. According to company measures, although quite variable over time, the average composition of rejected cullet is: glass (94%), plastics (2%), paper (1%), ceramics (2%), metals (0.5%) and organic compounds (0.5%). Special glass (all other EWC codes in Table 2) usually contains little non-glass material and its composition is rather constant over time.

According to Italian legislation (Ministero dell'Ambiente, 2005), unless recycled or reused, waste glass must be disposed of in an inert waste or non-hazardous waste landfill.

As far as the LCA model is concerned, it has been assumed that EWC 10 11 10, 10 11 12 and 06 05 03 would partially be disposed of in an inert waste landfill (50%) and partially (50%) in a non-hazardous waste landfill, while other EWC codes would be sent to inert waste landfills. Inventory data necessary to quantify the avoided environmental impacts were retrieved from the Ecoinvent database: "Disposal, glass, 0% water, to inert material landfill" and "Disposal, inert material, 0% water, to sanitary landfill".

2.4.2. Waste glass collection

Waste materials are transported from waste management facilities to the RFG production site. Company data were analysed in order to establish average routes and calculate average distances. Glass with EWC code 19 12 05 is transported on 30-tonne payload trucks for 246 km (weighted average). However, this distance must be reduced according to the avoided average transportation to the landfill facility (80 km). Thus, the net collection distance is 166 km. The same procedure has been used to estimate transportation of all other EWC codes, obtaining an average distance of 179 km. Road transportation by trucks has been modelled using the Ecoinvent datasets.

2.4.3. Washing

With respect to water consumption, input waste glass washing is a quasi-closed loop and the quantity of fresh water collected from a well is 0.83 m^3/t. Water is injected with oxygen to prevent anaerobic fermentation and it is then treated with a physico-chemical process; it enters with a chemical oxygen demand (COD) of 600 mg/l and leaves with a COD of 300 mg/l. Although the treatment is not sufficient to allow discharge to surface waters (160 mg/l being the maximum allowed COD according to Italian laws) it is enough to permit its re-use.

Washing 1 tonne of input material requires 1.4 kWh of electricity for running hydraulic pumps and treating waste water. Moreover, 2.2 kg of aluminium sulphate are used as coagulant, while 0.49 kg of liquid oxygen is used to prevent anaerobic fermentation and speed up the natural treatment by aerobic bacteria. In the washing process, 65 kg of wet sludge per tonne of washed glass are also produced. This sludge is added to the brick grade sand produced by the sorting process and sold to brick manufacturers.

2.4.4. Sorting, drying and milling

According to direct measures, sorting and milling of soda-lime glass require 25 kWh of electricity per tonne of input material. Bulk solid handling consumes 0.5 l of diesel, while drying consumes 103.59 MJ of natural gas. Natural gas drying has been modelled according to Ecoinvent data relevant to the unit "heat, natural gas, at industrial furnace >100 kW".

As previously mentioned, the composition of soda-lime and special glasses can vary over time; consequently, the sorting process must be flexible with respect to possible changes in input material composition, as well as with respect to the variation of market demand for RFG co-products (see Table 3).

Some co-products are internally re-used and some sold to third parties. The composition and market value of outputs are given in Table 3. "Ceramic grade sand" is the commercial name for a granular material made of glass and ceramic fragments/powders with characteristics suitable for ceramic tiles industries. Ceramic grade sand can either be sold, or used in mix with special glass, as the input materials in the foaming process. "Glass grade sand", whose chemical composition is reported in Bernardo et al. (2010) as "glassy sand", is the commercial granular materials sold to the glass industry. "Brick grade sand" is suitable for the production of bricks. A nil price means that the products are delivered free of charge, while R stands for internal reuse.

As previously stated, in the LCA model glass grade sand and ceramic grade sand displace primary production of silica sand, while brick grade sand displaces extraction of brick clay. Plastic scrap is internally used to produce electrical and thermal energy in a waste-to-energy facility. However, since the data related to the SASIL waste-to-energy system are not yet available, and given that post consumer plastic scrap in the study area is often co-incinerated in cement kilns in partial substitution of petcoke (Genon and Brizio, 2008), plastic scrap recovery has been modelled as an

Table 3
Composition and selling price of outputs (R = internal reuse; NA = not applicable).

Products	Output (%)	Destination	Selling price (€/t)
Ceramic grade sand	15	Ceramic industry/ RFG	22
Glass grade sand	80	Glass industry	36
Brick grade sand	3	Brick factories	3
Plastic scrap	1	Waste-to-energy	R
Metal scrap	0.5	Recycling	0
Organic materials	0.5	Water treatment	NA

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avoided production of petcoke. In the LCA model, a low heat value for plastic scrap (30 MJ/kg) and a 10% process loss were utilised. Also in the case of metals, an avoided production of the corresponding virgin materials has been assumed, according to Rigamonti et al. (2010).

Processing of waste special glass requires less electricity (12 kWh/t), as it contains less contaminants. This is likely a consequence of the different route for separate collection, in comparison to waste container glass. However, given the major difficulties related to closed-loop recycling, especially in case of waste CRT (Bernardo et al., 2005; Méar et al., 2005), and considering that no co-products can be recovered from waste special glasses presently treated by SASIL, no avoided impacts can be accounted for in the LCA model.

2.4.5. Thermal process

Glass/ceramic powder from soda-lime and special glass (97%) is mixed with silicon carbide (2%), calcium sulphate (0.5%) and calcium carbonate (0.5%) and is subsequently sent to a furnace in which the raw mix undergoes preheating, foaming (T = 900 °C), cooling and subsequent sizing. Although a detailed description of the foaming process is out of the scope of this paper, as it has been extensively described elsewhere (Hurley, 2003; Scarinci et al., 2005), some remarks will help clarify why the industrial partner has selected this specific mix, which is an evolution of a process presented in a previous paper (Bernardo et al., 2010). In fact, other chemical products and industrial minerals such as graphite, coal, MnO_2 and sulphates were also tested, but with less satisfying results or major costs. In rough terms, during the thermal process, silicon carbide (SiC) and calcium carbonate ($CaCO_3$) both generate CO_2 bubbles, which are the foaming agent. While the carbonate provides CO_2 by decomposition (Scarinci et al., 2005), SiC provides CO_2 by oxidation; such oxidation is operated by the oxygen available in the foaming furnace and also by the oxygen provided via the reduction of sulphates ($CaSO_4$) into sulphites and sulfides. The silica released as by-product of SiC oxidation is incorporated by the glass.

The structure of the foam glass traps a large part of the emitted CO_2, the rest being released into the atmosphere. Based on the total carbon in SiC and assuming a 50% entrapment, a direct emission of 11 kg of CO_2 per tonne of RFG has been entered in the LCA model. During the reaction, SO_4 is reduced to SO_3 and remains in the RFG matrix, therefore no SO_x air emissions needed to be accounted for. As far as heavy metals contained in the special glass are concerned, a set of laboratory tests showed that their volatility starts to significantly increase at temperatures above 1200 °C; this is also confirmed in the literature (Bernardo and Albertini, 2006; Bernardo et al., 2005). Consequently, no significant lead and barium air emissions are released from the furnace (T = 900 °C).

The required thermal energy of 1800 MJ/t is supplied by an electric furnace. A scenario with a natural gas furnace and a scenario with an electric furnace fuelled with electricity from natural gas co-generation were considered for comparison. Inventory data related to the Italian electricity mix, electricity from co-generation, use of a natural gas furnace and silicon carbide production were retrieved from the Ecoinvent database.

In order to take into account the variability of input materials, three possible mixes of soda-lime and special glass were considered (Table 4). As far as density and insulating properties of the RFG are concerned, these were found to remain within the range given in Table 1.

Table 4
Composition of the raw mix for RFG production.

	Soda-lime glass (%)	Special glass (%)
Mix 1	50	50
Mix 2	80	20
Mix 3	20	80

3. Results and discussion

RFG produced from Mix 1 (50% of soda-lime glass and 50% of special glass) has been chosen as the baseline scenario.

Table 5 displays mid-point indicators and the single score Eco-Indicator 99 relevant to both electric heating (E), natural gas heating (NG) and electric heating from natural gas co-generation (C). The differences in terms of environmental performance are remarkable: −52% in the case of AP and −54% in the case of POCP.

3.1. Contribution analysis

Impacts are due to transportation, processing and firing, while savings come from avoided landfill and recovery of co-products (Fig. 3).

It can be observed that the environmental gains related to the avoided landfill are cancelled by the transport-related impacts. Thus, it is not sufficient to base environmental claims on the statement that RFG is sustainable because it avoids landfilling, as the related gains are lower than the induced impacts.

An important contribution to improve the environmental profile of RFG is represented by recovered plastic, metals and glass fragments/powders, whose environmental gains are higher than those corresponding to landfill avoidance. This suggests that, in order to improve the RFG eco-profile, the raw mix should preferably be made of soda-lime glass rather than special glass, which does not contain recoverable metals and plastics. This finding highlights that industrial symbiosis can play a key role in eco-efficient glass recycling and further supports the recommendation of Hurley (2003) according to which closed-loop container glass recycling remains a preferable option.

Production of SiC and RFG firing represent the highest induced impacts. In spite of the small amount used, SiC is an important contributor to the overall impacts. Consequently, although SiC proved to be an excellent foaming agent (Bernardo et al., 2007), a more environmentally friendly additive is preferable. A possible solution to cutting SiC-related impacts might be replacing primary SiC with waste SiC, which is an option already investigated in the literature (Bernardo et al., 2007; Fernandes et al., 2009). Another source for waste SiC, which is likely to be pursued by SASIL SpA is the use of waste SiC from end-of-life roll-conveyors currently used in ceramic and sanitaryware production. Waste SiC would correspond to zero impacts related to SiC production and limited impacts for transportation and grinding.

3.2. Sensitivity and improvement analysis

The influence of the raw mix composition (Table 4) has been investigated, as reported in Fig. 4.

Table 5
Eco-profile of RFG made from Mix 1 according to different thermal processes.

Indicator		RFG-E	RFG-NG	RFG-C
GER	MJ/t	7761	5118 (−34%)	5405 (−30%)
GWP	kg CO_{2eq}/t	513	349 (−32%)	386 (−25%)
AP	mol H^+/t	77	37 (−51%)	37 (−52%)
EP	g O_{2eq}/t	7907	4338 (−45%)	5583 (−29%)
POCP	g C_2H_{4eq}/t	12.6	8.6 (−31%)	5.8 (−54%)
EI-99	Pt/t	23	15 (−35%)	17 (−25%)

E = electric heating using electricity from the Italian mix; NG = natural gas heating; C = electric heating using electricity from natural gas co-generation.

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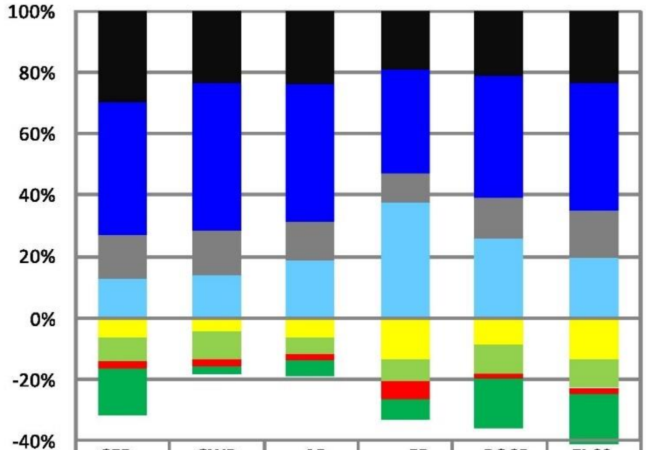

	GER (MJ)	GWP (kg CO_2)	AP (mol H+)	EP (g O_2)	POCP (g C_2H_4)	EI-99 (Pt)
Foaming agent	3382	144.6	22.5	2237	4.1	9.1
RFG firing	4928	304.2	42.8	4078	7.9	16.4
Processing	1595	92.7	12.2	1113	2.6	5.8
Transport	1491	86.1	17.8	4458	5.1	7.8
Avoided landfill	-773	-27.7	-6.4	-1605	-1.7	-5.3
Glass recovery	-876	-55.3	-5.0	-881	-1.9	-3.6
Metal recovery	-213	-18.1	-1.8	-663	-0.3	-0.9
Energy recovery	-1772	-13.1	-5.1	-827	-3.1	-6.2

Fig. 3. Induced and avoided impacts in the RFG waste-to-production chain (Mix 1, electric heating) – (100% = sum of induced impacts).

Mix 2 shows the best performance according to GER, POCP and EI-99. Mix 3 has the lowest impact according to GWP, AP and EP.

Since the electricity from the Italian grid showed a heavy contribution to the environmental impacts of RFG, an improvement scenario could be switching from an electric to a natural gas fuelled kiln. Such a proposed change of technology was regarded as too costly and thus a second improvement scenario was proposed, namely using electricity from a natural gas co-generator instead of drawing electricity from the grid.

The comparison between the eco-profiles of RFG produced using electricity from the grid, RFG produced with a natural gas fuelled kiln, and with an electric kiln powered with a co-generator are shown in Table 5. The solution adopted by the industrial partner, i.e. electric kiln plus natural gas co-generator, shows reasonably good environmental performance at reasonable costs. New energy-saving foaming processes are described in the literature (Hurley, 2003; Scarinci et al., 2005), but are not presently used for commercial purposes.

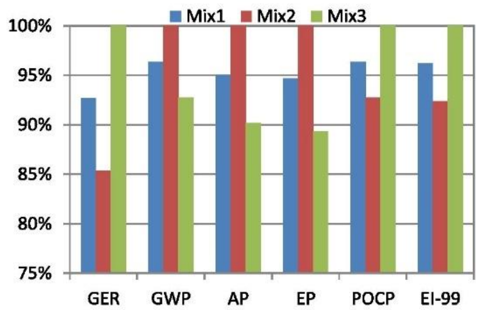

Fig. 4. Comparison among RFGs produced from different raw mixes (electric heating) – (results are normalised to the case with the highest impact).

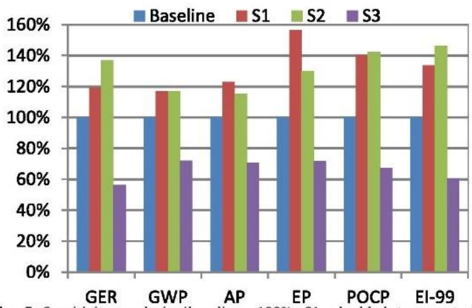

Fig. 5. Sensitivity analysis (baseline = 100%; S1 = doubled transport distances; S2 = exclusion of avoided products; S3 = use of waste SiC).

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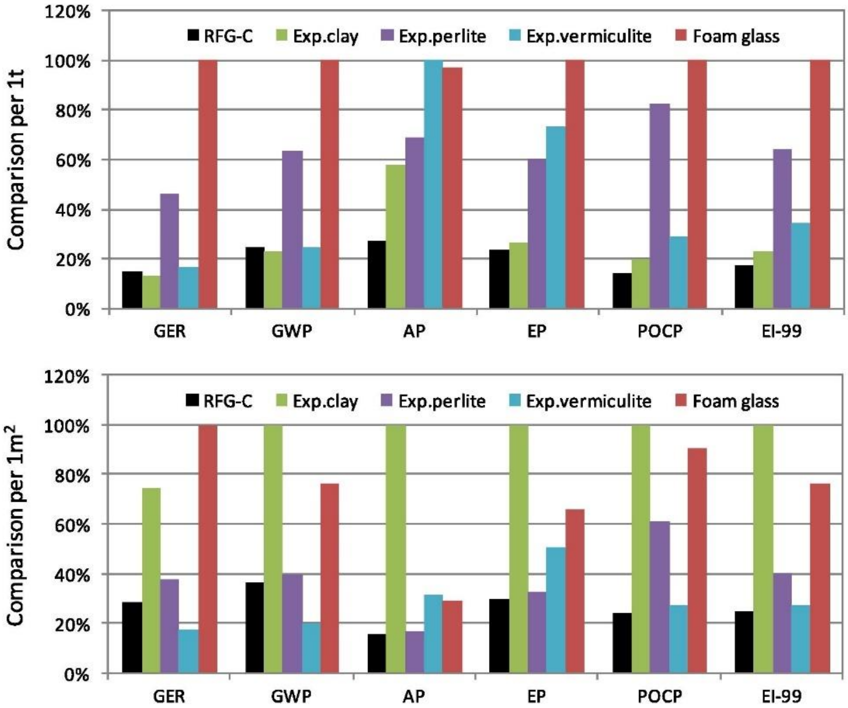

Fig. 6. Comparison among RFG (RFG-C = electric heating from gas cogeneration) and some selected mineral-based insulating materials – (results are normalised to the case with the highest impact).

Given the remarkable influence of transport-related impacts, recovery of co-products, and SiC production, a sensitivity analysis is presented in Fig. 5, where, one at a time, the parameters are changed and compared to the baseline scenario: transport distances are doubled; avoided products are excluded; SiC is substituted by waste SiC.

Bearing in mind that a meaningful comparison among insulating materials should be carried out only once their end-use and the function of the system under study have been identified, Fig. 6 shows energy and environmental indicators relevant to the materials shown in Table 1. The RFG eco-profile is that for electric heating and natural gas co-generation (RFG-C). Inventory data for the remaining insulating materials are taken from the Ecoinvent. The comparison is given per unit of mass (1 tonne) and per unit of area of an insulating board with the same thermal resistance, using average densities and thermal conductivities from Table 1. No further comments are provided because the aim of this paper has been to describe the recycling route under study and stimulate interest in more transparent LCAs of other building products made from recycled materials.

4. Conclusions

LCA led to a deeper understanding of the RFG eco-profile, and improved environmental management of the waste-to-production chain. Moreover, the results provided scientific background supporting the producer's environmental claims and information essential for conducting LCAs of future end-uses of RFG in energy efficient buildings.

The main environmental strengths and weaknesses of the waste-to-production chain can be summarised as follows:

- The environmental gains related to landfill avoidance are offset by increased transport-related impacts.
- The environmental gains related to recovery of co-products are higher than those from avoided landfilling. This emphasises the role of an eco-efficient waste glass recycling chain, in which an innovative multi-output process made it possible to re-process low quality glass rejects and sell most of them back to their original industries (closed-loop recycling), while only a small fraction of post-consumer glass is used in RFG production. This is a synergy obtained from integrated waste management and production in the context of industrial symbiosis and eco-efficient recycling.
- As energy use for the thermal process is a hot spot, LCA results suggested switching to a natural gas powered kiln or an electric kiln powered with a natural gas co-generator, the latter being the solution adopted by the industrial partner.
- A further important improvement could be obtained through the substitution of silicon carbide for a more environmentally friendly additive, or the use of waste silicon carbide from end-of-life roll-conveyors currently used in ceramic and sanitary-ware production.
- For a more comprehensive comparison between RFG and other building materials it will be necessary to better define the end-uses. Otherwise, it will not be possible to fully understand the direct and indirect environmental gains that RFG will transfer to the final product (i.e. the building as a whole). Nevertheless, the eco-profile of RFG has been contrasted against those of

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other insulating materials. This should be helpful in order to discuss the relative importance of single subsystems that transfer environmental gains and/or impacts to green products made from waste, for instance: landfill avoidance, change in transportation distance, recovery of co-products and, finally, energy/material saving in highly energy intensive thermal processes.

Acknowledgements

The research was carried out in co-operation with SASIL SpA within the EU Project NOVEDI (www.sasil-life.com/), financed under the call LIFE+ 2007 (contract ENV/IT/00361). The authors would like to thank Dr. Enrico Bernardo (University of Padua, Italy) for the support and for the fruitful and stimulating discussion.

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Please cite this article in press as: Blengini, G.A., et al. Eco-efficient waste glass recycling: Integrated waste management and green product development through LCA. Waste Management (2011), doi:10.1016/j.wasman.2011.10.018

8. ANNEX C: PARTICIPATORY APPROACH, ACCEPTABILITY AND TRANSPARENCY OF WASTE MANAGEMENT LCAS: CASE STUDIES OF TORINO AND CUNEO.

Waste Management 32 (2012) 1712–1721

Contents lists available at SciVerse ScienceDirect

Waste Management

journal homepage: www.elsevier.com/locate/wasman

Participatory approach, acceptability and transparency of waste management LCAs: Case studies of Torino and Cuneo

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ARTICLE INFO

Article history:
Received 23 November 2011
Accepted 13 April 2012
Available online 12 May 2012

Keywords:
Participatory approach
Stakeholders engagement
Waste management
Municipal solid waste
Attributional life cycle assessment
LCA assumptions

ABSTRACT

The paper summarises the main results obtained from two extensive applications of Life Cycle Assessment (LCA) to the integrated municipal solid waste management systems of Torino and Cuneo Districts in northern Italy. Scenarios with substantial differences in terms of amount of waste, percentage of separate collection and options for the disposal of residual waste are used to discuss the credibility and acceptability of the LCA results, which are adversely affected by the large influence of methodological assumptions and the local socio-economic constraints. The use of site-specific data on full scale waste treatment facilities and the adoption of a participatory approach for the definition of the most sensible LCA assumptions are used to assist local public administrators and stakeholders showing them that LCA can be operational to waste management at local scale.

1. Introduction

Life Cycle Assessment (LCA) applied to sustainable municipal solid waste management has rapidly expanded over the last few years as a tool that is able to capture and handle complexities and interdependencies typically characterising modern integrated waste management systems (I-WMS).

A recent, fairly comprehensive and extensive literature review by Pires et al. (2011b) pointed out to what extent a system approach is becoming strategic in order to take into account many technical and non-technical aspects of solid waste management systems. In fact, I-WMSs should be analysed as a whole, since they are inter-related with one another and developments in one area frequently affect practices or activities in another area. The same authors (Pires et al., 2011b) classified nine system assessment tools commonly used in waste management (WM), among which LCA clearly emerged as the most popular, scientifically sound and worldwide appreciated.

The context, objectives and operational conditions that characterise the growing number of recently published LCA applications to WM are quite variable.

With reference to the European countries and the most recent legislation published by the European Commission (EC), i.e. the Waste Directive 2008/98/EC (EU, 2008), in the geographical areas where WM is closer to the EU targets of sustainability, LCA is mostly used to rationalise technological choices and management strategies, while in less advanced regions LCA is used to develop measures to implement more integrated solid waste management and reach EU directives.

As reported in Rigamonti et al. (2010), several LCA studies deal with the I-WMS as a whole (i.e. from a system perspective), while other studies are focused on single subsystems (or groups of subsystems taken individually) devoted to the treatment of single waste fractions.

Although the LCA methodology and the WM related tools are rapidly expanding, there are still uncertainties and open issues, which are challenging the scientific community and that, are limiting the diffusion among end-users. Thus, the question "What life-cycle assessment does and does not do in assessments of waste management" raised by Ekvall et al. (2007) is still partially unanswered and still of great interest.

One of the key issues is understanding what LCA can do for local waste authorities and operators. Moreover, it is still unclear to what extent these subjects are aware of the potential of WM LCAs and/or are willing to put into practice the results.

LCA can supply objective and comprehensive information, but, in Italy and elsewhere, the final decision lies mostly with public

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http://dx.doi.org/10.1016/j.wasman.2012.04.010

administrators seldom aware of the great potential of LCA. Such public administrators often set up priorities and take decisions more on financial constraints rather than on environmental optimisation issues (Blengini, 2008).

Beyond the great advancements of the scientific community, the central question is therefore: "is WM LCA fully operational to business?" In other words: "is LCA accepted, used, understood and put into practice by all the stakeholders?".

LCAs of complex and inter-dependent systems such as WMs necessarily reflect complexity, which is also influenced by non-technical factors, site-specific aspects and local socio-economic constraints. Therefore, LCAs are difficult to be handled and/or developed by non-experts. Moreover, the results of a LCA applied to an I-WMS are unique and should never be generalised, though a lesson can be learned. Consequently, the most important message from a WM LCA should not be the final results, but rather a combination of the results and the way LCA was conducted.

In order to develop meaningful LCAs of I-WMS: (1) goal and scope must be clearly identified, justified and outlined; (2) both input data and inventory results must be fully made available and it should be possible to mathematically manipulate them. Numerical results unsupported by assumptions and the full dataset might be of limited importance.

Two examples taken from the literature might help clarify the context. Rigamonti et al. (2009, 2010) have shown to what extent the selection efficiencies, the adopted technologies and the methodological assumptions related to avoided products might drastically change the overall environmental performance of WM subsystems. A similar picture is presented by Merrild et al. (2008) in case of wastepaper recycling vs. incineration, where the overall energy and environmental indicators can change from positive (net impact) to negative (net gain) depending on the combination of the adopted technologies.

The consequence of the large variability of the environmental performance of subsystems, which heavily depends on assumptions, becomes exponential when dealing with an I-WMS. Moreover, when also socio-economic constraints are taken into account, including, for instance, the preferences of stakeholders relevant to different areas of environmental concern, it is very possible that LCA results become subjective to a large extent, with consequent increased scepticism and loss of credibility and acceptance. This is a very important area of concern that represents an obstacle to the diffusion of WM LCAs, and is also the central point of the present article, where two extensive LCAs run by the Politecnico di Torino in the years 2008 and 2009 (Blengini et al., 2008, 2009) are used in order to discuss on strategies to boost adoption of LCA in WM in northern Italy, and elsewhere, and increase the credibility and acceptability of results.

The original contribution of the present paper can be summarised as follows:

- Use of site-specific data on full scale waste treatment facilities in the study area in order to cover all the WM activities in the I-WMS and the full life cycle of waste;
- use of the participatory approach in order to address the most sensible LCA assumptions and propose solutions in order to enhance the acceptability of results;
- assist the local public administrators in order to verify and quantify the effectiveness of EU strategies on WM using site-specific data and taking into account the local socio-economic constraints, emphasising that LCA application is both useful and feasible.

2. Model and data development

The paper presents a synthesis and the main results of two research programmes focused on the application of LCA to a set of WM scenarios in Torino and Cuneo Districts in northern Italy (Blengini et al., 2008, 2009). The study area covers a population of nearly 2800,000 inhabitants with an annual generation of nearly 1500,000 tons of municipal solid waste (Fig. 1). In both cases, the overall objective was identifying scenarios with best energy and environmental performance. A detailed energy and environmental analysis was carried out for the main components of the I-WMS and for the I-WMS as a whole in order to support public administrators towards sustainable waste management.

The above research programmes were developed by the Politecnico di Torino and funded by the WM Authorities of Torino and Cuneo Districts. LCAs were implemented using the SimaPro 7 software (SimaPro7, 2006).

All the subsystems included in the I-WMS were considered and analysed paying attention to energy and environmental implications and inter-dependencies. Separate collection (SC) and its downstream recycling chains were investigated in terms of environmental benefits and impacts, in order to quantify advantages and drawbacks that can be ascribed to the new objectives of SC (65% by the year 2012) introduced by the law presently in force in Italy (Dlgs.152/06). At the same time, the role and environmental implications of energy recovery from residual waste were analysed, paying attention to the consequences of possible pre-treatment options of the residual waste, and considering both incineration and co-incineration.

2.1. Definition of goal and scope through a participatory approach

The LCA methodology according to ISO 14040 (2006) is worldwide accepted and appreciated because it allows an objective evaluation of the environmental performances of products and processes (Guinée, 2002).

However when applying LCA to WM, there are some sector-specific aspects that must be considered and assumptions to be undertaken that might affect the results to a large extent (Ekvall et al., 2007; Finnveden, 1999; Merrild et al., 2008; Rigamonti et al., 2010).

In order to keep under control the negative influence that assumptions might have in terms of acceptability of the results, a participatory approach was adopted since an early stage of the research. When applied from the very beginning, a participatory process may be of help in reducing possible conflicts among opposite interest groups, which is typical in waste management, and contribute towards defining acceptable solutions for all involved parties (Salhofer et al., 2007b). As it was observed in other case studies, where a structured participative approach was applied to waste management, different stakeholders have different objectives (Pires et al., 2011a) and some of them might try to influence the results by changing the criteria in a late stage (Salhofer et al., 2007b). Setting up clear and shared rules and preferences since the beginning is therefore a key issue (De Marchi et al., 2000).

A panel of stakeholders and experts, including participants from Politecnico di Torino, WM authorities of Torino and Cuneo Districts and Environmentalist NGOs, was set up. The trans-disciplinary nature of the panel was similar to those presented in De Marchi et al. (2000) and in Salhofer et al. (2007b), where attention was paid to include all local actors and give them equal opportunities to express their opinion.

An initial brainstorming and subsequent structured meetings were used in order to reach a shared definition of the following aspects that, as the participants revealed, can highly increase the acceptability of the LCA results:

- Identification and description of the scenarios to be compared: amount of waste, composition, percentage of SC, definition of technologies/strategies not yet defined in the local WM policies/plans;

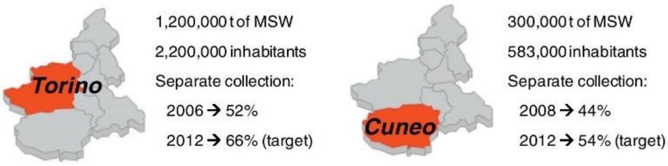


Fig. 1. Municipal solid waste production and population in the Torino and Cuneo districts.

- definition of the case-specific LCA methodological assumptions: system boundaries, avoided/substituted production, cut-off criteria;
- data collection, sources and responsibilities: mass balances, energy consumption and emissions for all the WM subsystems.
- selection of meaningful energy and environmental indicators;

while the detailed description of input data, assumptions and subsystems are reported as Supplementary content, and are fully available in the research reports quoted in the references (Blengini et al., 2008, 2009), in the following paragraphs the main aspects relevant to the discussion within the panel of stakeholders are summarised.

The following sources of data were also used and quoted in the Supplementary content: (AEA, 2001; ANPA, 2000; Blengini et al., 2007; Bovea and Powell, 2006; Cernuschi et al., 2003; Ecoinvent, 2007; Favoino and Hogg, 2008; Giugliano, 2007; Rigamonti et al., 2009).

2.2. Identification and description of scenarios to be compared

With reference to the Torino District, four scenarios relevant to the whole I-WMS were identified: 1A, 1B, 2A, 2B. Scenarios 1 (A/B) depict the status of SC in the year 2006, which was 52.1%, whereas scenarios 2 (A/B) represents the target of SC for the year 2012, which was set at 65.7%.

It must be said that, at the time of the study, the construction of a new incinerator had already been launched, but some important aspects related to the residual waste management chain were still undefined. Thus, inventory data to model residual waste management had to be retrieved from design documents provided by the main contractor, or assumed via consultation with the panel of stakeholders. Therefore, scenarios 1 (A/B) depict a past situation where management of separately collected waste was modelled on real data, while residual waste management is modelled as if incineration was already operational.

The discussion among the panel of stakeholders about adoption or exclusion of a mechanical–biological treatment (MBT) of the residual waste prior to incineration was particularly controversial. An agreement was reached that two out of four scenarios include MBT (1B/2B) and the remaining exclude MBT (1A/2A). However, the panel was not able to reach a complete agreement and left the description of the MBT characteristics and operating conditions vague to some extent.

The destination of biowaste out of MBT was felt of particular importance. It was agreed that the baseline scenarios would consider landfilling of biowaste out of MBT, while possible improvement scenarios were left for a sensitivity analysis that will be presented in Section 5, and for future more detailed investigations. A graphical description of the four scenarios under analysis is available in the Supplementary content.

As Cuneo District is concerned, it must be remarked that the size, socio-economic constraints and organisation of the WM facilities and infrastructures are quite different in comparison to Torino District. In that case, a scenario analysis was carried out to better understand the environmental and energy implications of the I-WMS re-organization in progress at the time of the study (2009).

However, to the aim of the present paper, none of the results relevant to the I-WMS as a whole are provided, but rather a comparison among four residual waste management alternatives (chains) is carried out:

- chain 1: residual waste to MBT, dry waste to residue derived fuel (RDF) production and co-incineration in an existing cement kiln;
- chain 2: residual waste to bio-drying, dry waste to RDF production and co-incineration in an existing cement kiln;
- chain 3: residual waste to a dedicated incinerator;
- chain 4: residual waste to MBT and dry waste to a dedicated incinerator.

Chains 1–4 encompass the whole sequence of activities in the life cycle of 1 ton of residual waste, from collection to the final disposal of residues, including substitution of primary energy and/or recovery of secondary materials. Data on quantities of waste and composition are presented in Table 1. Detailed data are supplied as Supplementary content.

2.3. Definition of the case-specific LCA methodological assumptions

It is well known that identifying and clearly describing system boundaries is a very important step that can heavily influence LCA results. Although the expansion of the principal system boundaries in order to avoid allocation is warmly recommended since long (Finnveden, 1999; ISO 14040, 2006), it must be noticed that case-specific choices related to expanded system boundaries are still controversial (Ekvall et al., 2007).

System expansion avoids allocation, but introduces subjective choices relevant to the substituted primary production, for instance the electricity produced from primary sources that is displaced by energy recovery from waste. Moreover, this introduces a crediting system (negative figures) that is sometimes source of confusion and/or misinterpretation among non-LCA experts. In such a context, an important step is the choice between attributional or consequential LCA modelling (EC et al., 2010). In rough terms, attributional modelling means taking a picture of the present operating conditions, while consequential modelling implies that the LCA model is made representative of an evolution of the present situation towards a given target.

This said, although consequential modelling is certainly more relevant for long-term decision making, this introduces value choices that were again source of conflict among the panel. Some citizen's groups feared that the bargaining power of public authorities could drive the choice of primary energy to be substituted and therefore distort the results, thus increasing scepticism. On the other hand, everybody was interested to know the environmental performance of the full-scale local WM activities and inter-dependencies. It was therefore decided to adopt an attributional principle and create a LCA model that (where possible) represents the actual fate of waste in the study area and the actual displacement

Table 1
Total waste, waste composition and separate collection in Torino and Cuneo Districts.

Waste type	Torino District		Cuneo District	
	2006; SC = 52.1%; ton × 1000	2012; SC = 65.6%; ton × 1000	2008; SC = 43.7%; ton	2012; SC = 54.4%; ton
Organic	190	219.7	6247	13,299
Green	51.7	51.8	16,866	19,319
Plastic	35.1	47.4	10,806	17,264
Paper	193.4	260	51,745	61,930
Wood	54	52.3	6526	7399
Glass	67.5	74.2	23,566	27,635
Metals	20.2	24.7	6200	7029
Other	45	63.2	12,178	16,585
Residual waste	603	414	172,605	142,820
Total	1260	1203	306,738	313,279

of primary energy/materials. Moreover, the panel of stakeholders recommended to include in the LCA model also those waste flows (and the related waste treatment) that go outside the administrative territory, according to actual data.

2.4. Criteria for missing inventory data

An important topic that was discussed among the panel of stakeholders at the initial stage was that relevant to the influence of transport-related impacts. Public administrators didn't have comprehensive and quantitative data on collection systems (from the place where waste is generated to the collection centres or transfer stations), but tended to put much emphasis on this aspect, saying that SC increases transportation distances and consequently impacts might outweigh savings from SC and downstream recycling. On the opposite side, citizen's groups, which are often more ecologically oriented than public administrators (Salhofer et al., 2007b), pushed more towards recycling. According to them, although transport distances might increase to some extent, SC is carried out with smaller and more efficient vehicles, which should compensate impacts.

Given the absence of comprehensive and reliable data on collection distances, the two opposite parties agreed to assume a transportation distance of 50 km for all types of collected waste. According to the field experience of public administrators, such a distance would be an overestimate of the real conditions. All the participants where interested to obtain a proxy quantitative estimate of transport-related impacts and contrast it against the environmental gains of recycling. However, they didn't want such a rough estimate influence the environmental comparison among WM scenarios.

For the sake of clarity, it must be said that transportation distances from transfer stations to waste treatment facilities were all available, and therefore were included in the LCA model (see Supplementary content).

2.5. Selection of meaningful energy and environmental indicators

As the selection of impact indicators is concerned, these were chosen through a consultation with the panel of stakeholders.

As suggested by Kruse et al. (2009) a combined top-down and bottom-up approach was adopted in order to develop a meaningful suite of indicators. A top-down approach can roughly be described as one that selects indicators that are representative of broadly recognised areas of environmental concern, as well as based on various international conventions, agreements, and guidelines. This approach is consistent with the International Standards Organization's (ISO) recommendations for LCIA methods (ISO 14040, 2006). In contrast, a bottom-up approach can be defined as one that identifies indicators based on industry, public administrators or stakeholder interests and/or data availability (Kruse et al., 2009).

This said, a first set of two impact categories was initially proposed according to the above mentioned top-down approach: emission of greenhouse gases and use of non-renewable energy.

Based on the bottom-up approach, the panel confirmed that energy and climate change are meaningful impact categories and expressed an interest to include some indicators relevant to human health. However, after a discussion, it clearly emerged that a careful modelling of human health impacts would have been much more complicated and not compatible with the time frame and data availability. There are in fact some important methodological aspects to be taken into account when incorporating local-scale environmental impacts in LCA (Kruse et al., 2009).

A comprehensive discussion about local-scale environmental impacts in LCA is beyond the scope of this article, however some remarks can help the reader. LCA is in fact mostly relying on additive indicators, i.e. that can be measured quantitatively and that are additive through the chain. Such indicators are calculated through characterization factors that can capture regional and global burdens, but which mostly neglect local and site specific aspects. This is obviously a problem that deserves to be accurately addressed in WM LCAs when human health is concerned. It is in fact very possible that some emissions from the foreground system and the background system are treated as additive, while they are not.

Selected indicators are therefore: GER (Gross Energy Requirement) expressing the total primary energy resource consumption (Boustead and Hancock, 1979); NER (Non-renewable Energy Resources) as the non-renewable part of GER; GWP_{total} (Global Warming Potential – 100 years) as an indicator of the greenhouse effect, including biogenic carbon dioxide (IPCC, 2006); GWP_{fossil} as part of GWP_{total} with exclusion of biogenic carbon dioxide.

Given the data reported as Supplementary content, these allow calculation of environmental indicators typically included in WM LCAs, including human toxicity. However, for the above consideration, although Eco-Indicator 99 (Goedkoop and Spriensma, 1999) was used in order to provide a rough estimate to the panel, human health and ecosystem quality issues were left outside the final panel discussion and are therefore not reported in this paper.

2.6. Methodology for accounting of biogenic carbon dioxide emissions

Accounting and reporting of biogenic carbon emissions and/or sequestration was another important topic under panel discussion. It was agreed that all biogenic carbon emissions were to be accounted and included in GWP_{total}, whereas GWP_{fossil} excludes biogenic carbon dioxide.

Since the greenhouse effect is determined either by fossil and biogenic carbon dioxide (Blengini, 2008; Hogg et al., 2008), in LCAs applied to WM the assumption excluding the biogenic carbon dioxide is not scientifically correct. In fact, the simplification whereby the biogenic carbon dioxide cycle is neutral (the amount of CO_2 absorbed during the plant growth is the same released in its end of life) is an oversimplification when comparing disposal scenarios with different potentials of biogenic carbon dioxide generation.

Here it must be said that there are different accounting methodologies to handle carbon uptake, carbon sequestration in landfill, sequestration associated to the use of compost in agriculture and biogenic emissions. It is therefore necessary to consistently handle C-uptake and emissions throughout the whole life cycle (Christensen et al., 2009; Rabl et al., 2007).

According to Christensen et al. (2009), a simple and transparent model for the calculation of C-balances is recommended. Beyond that, in the authors of this paper opinion, it is also important to report in a transparent way, i.e. separating the biogenic carbon dioxide contribution.

As the present study is concerned, the following accounting rules for biogenic carbon were adopted:

- Generated waste holds no carbon credits (and no environmental burdens) according to the so-called "zero burden assumption" (Ekvall et al., 2007). This assumption can also be supported by the statement of Vergara et al. (2011): "if waste carries with it no environmental burdens, then it should not carry with it any environmental benefits either".
- All biogenic carbon dioxide emissions associated with WM are accounted for (GWP + 1).
- Biomass recycling corresponds to permanent locking of carbon according to the mass flow that is permanently re-circulating in the loop. In practical terms, recycling corresponds to locking of carbon dioxide absorbed during the growth of the biomass. This is automatically obtained in the present LCA according to the following example: if the biomass is incinerated, a CO_2 emission is recorded (GWP + 1), while recycling has no direct emissions (GWP 0).
- Biomass landfill and use of compost in agriculture is assigned a carbon sequestration potentials (GWP − 1).

Detailed data and assumptions are reported in Supplementary content.

3. Results

The following results reflect field data from the study area (Piedmont, Italy) and reflect the full-scale performance of existing (or under construction) WM plants. All the activities in the I-WMS were considered and analysed with focus on their energy and environmental implications and inter-dependencies.

SC and its downstream recycling/treatment are investigated first, paying attention to the net environmental gains, i.e. the eco-balance between recovered materials/energy and the environmental impacts of WM activities. The analysis turns then on the I-WMS as a whole, in order to better understand the role (and weight) of WM subsystems in a more holistic perspective. The environmental implications of energy recovery from residual waste are then presented, with focus on the consequences of possible pre-treatment options of the residual waste, and considering both incineration and co-incineration.

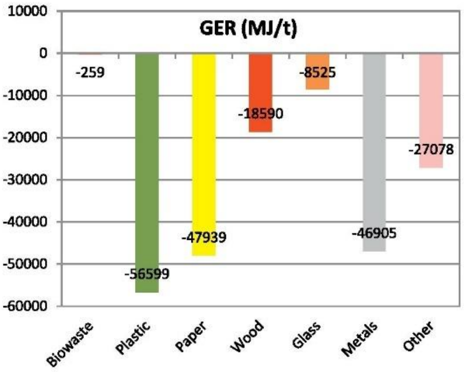
3.1. Energy and carbon balance of recycling/treatment of single materials from SC

With reference to 1 ton of separately collected waste, Fig. 2 shows the energy and carbon balances of SC and subsequent recycling/treatment in a life cycle perspective. The sequence of activities starts after collection (not included) and encompasses transportation, selection, recycling/treatment and substitution (avoided products/energy). Both the main waste flows and residues were included in the analysis, whereas residues are either landfilled of sent to energy recovery. Negative indicators mean that environmental gains are higher than induced impacts. Biowaste refers to a mix of composting and anaerobic digestion (AD), which reflects the current situation in Torino (33% AD and 67% composting), while metals refer to a mix of ferrous and non-ferrous metals (detailed data are reported as Supplementary content).

With reference to Fig. 2, GER savings were found to be substantially similar to non-renewable energy savings, except paper (NER = −13770 MJ/t) and wood (NER = −3559 MJ/t) where most of the GER is renewable energy from biomass. As carbon emissions are concerned, GWP_{total} and GWP_{fossil} were found to be substantially the same, except for biowaste where GWP_{fossil} shows a net saving of −163 $kgCO_{2eq}$/t.

3.2. Analysis of whole I-WMS: comparison of the four scenarios in Torino District

With reference to 1 ton of total waste, Table 2 shows the energy and carbon balances related to the four scenarios. According to both energy and climate change indicators, scenarios with 65.6% of separated collection appear to be more eco-efficient than those with 52.1%. Scenarios which include MBT (1B and 2B) show a more favourable carbon balance, but perform worse in terms of energy balance.



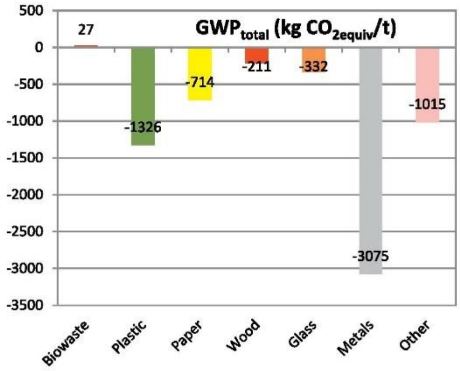
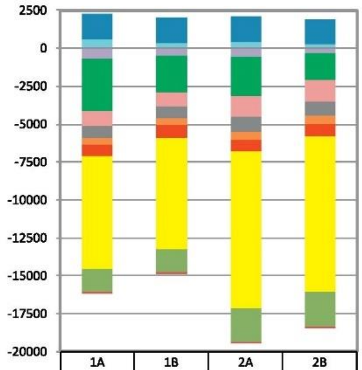


Fig. 2. Energy and carbon balance of separately collected waste materials.

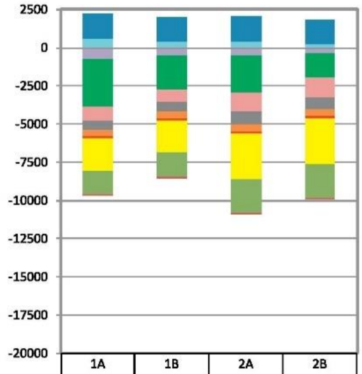
Table 2
Energy and carbon balance of the four scenarios under comparison (Torino District).

Impact category	Unit	Scenario 1A	Scenario 1B	Scenario 2A	Scenario 2B
GER	MJ/t	−13,898	−12,858	−17,362	−16,497
NER	MJ/t	−7476	−6499	−8811	−8001
$GWP_{100total}$	kg CO_2eq/t	233	142	26	−46
$GWP_{100fossil}$	kg CO_2eq/t	−156	−160	−230	−241



GER [MJ/t]	1A	1B	2A	2B
Collection	1633	1633	1633	1633
Biowaste	-50	-50	-58	-58
Plastic	-1575	-1575	-2219	-2219
Paper	-7356	-7356	-10299	-10299
Wood	-797	-797	-804	-804
Glass	-456	-456	-523	-523
Metals	-753	-753	-959	-959
Others	-967	-967	-1411	-1411
Residual Waste	607	368	436	254
Electricity (WtE)	-3458	-2400	-2606	-1743
Heat (WtE)	-726	-504	-550	-367

Fig. 3. Contribution of subsystems to the energy balance of the I-WMS of Torino District (GER).



NER [MJ/t]	1A	1B	2A	2B
Collection	1613	1613	1613	1613
Biowaste	-50	-50	-59	-59
Plastic	-1561	-1561	-2199	-2199
Paper	-2113	-2113	-2958	-2958
Wood	-152	-152	-154	-154
Glass	-396	-396	-454	-454
Metals	-642	-642	-818	-818
Others	-869	-869	-1269	-1269
Residual Waste	593	378	426	261
Electricity (WtE)	-3174	-2203	-2392	-1600
Heat (WtE)	-724	-502	-549	-365

Fig. 4. Contribution of subsystems to the energy balance of the I-WMS of Torino District (NER).

A contribution analysis was conducted in order to quantify the relative importance of various subsystems over the environmental performance of the I-WMS as a whole (per 1 ton of total waste). Figs. 3–6 report the contributions of collection and WM activities related to single waste fractions. The contribution of separately collected wastes encompass the same activities already described in Section 3.1. Such a contribution depends on the overall recycling efficiency (Fig. 2) and the quantities shown in Table 1.

In the case of residual waste, recovered electricity and heat (WtE) are reported separately in the lower part of the tables embedded in Figs. 3–6, whereas "residual waste" encompass the impacts from pre-treatment (scenarios 1B/2B), incineration and landfill of residues. In order to provide more quantitative information on the impacts related to residual waste, in scenario 2B the contribution of pre-treatment to the GWP_{total} is 16 kg CO_{2eq}/t, that of incineration is 300 kg CO_{2eq}/t and that of stabilised organic fraction landfill is 23 kg CO_{2eq}/t.

3.3. Comparison among alternatives for energy recovery from residual waste: Torino and Cuneo districts

A comparison among alternative chains for energy recovery from residual waste was felt of strategic interest by the panel of stakeholders. The results are presented in Table 3 and are based on both Torino and Cuneo Districts LCAs (detailed data are reported as Supplementary content).

With reference to 1 ton of residual waste, Chains 1 and 2 are based on two alternative processes for the production of RDF and subsequent co-incineration in an existing cement kiln located in the surroundings of Cuneo town, which currently produces 1.6 Mt of clinker per year and hold a co-incineration capacity of 100 kt.

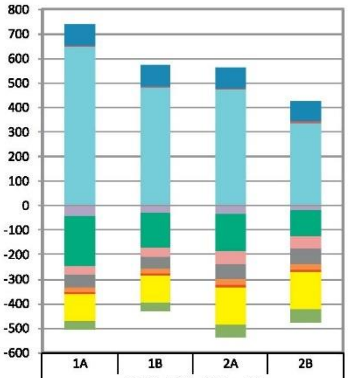
Chains 3 and 4 reflect the operating conditions of the above described incinerator under construction in the town of Torino. In particular, Chain 3 is based on inventory data retrieved from the scenario 1A of Torino District LCA (without MBT), while Chain 4 is based on inventory data retrieved from the scenario 1B of Torino District LCA (with MBT).

Table 3 shows that Chain 1 is the most efficient in terms of energy recovery and greenhouse emissions.

4. Discussion

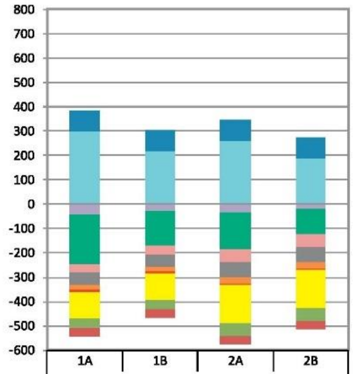
The results from the LCA were discussed with the panel of stakeholders.

A first important discussion was that relevant to the ecological relevance of transport. Public administrators in the panel were unaware of the quantitative impacts of collection and transporta-



GWPtotal [kg CO2 eq/t]	1A	1B	2A	2B
Collection	84	84	84	84
Biowaste	5	5	6	6
Plastic	-37	-37	-52	-52
Paper	-110	-110	-153	-153
Wood	-9	-9	-9	-9
Glass	-18	-18	-20	-20
Metals	-49	-49	-63	-63
Others	-36	-36	-53	-53
Residual Waste	649	483	472	339
Electricity (WtE)	-204	-142	-154	-103
Heat (WtE)	-42	-29	-32	-21

Fig. 5. Contribution of subsystems to the carbon balance of the I-WMS of Torino District (GWP_{total}).



GWPfossil [kg CO2 eq/t]	1A	1B	2A	2B
Collection	84	84	84	84
Biowaste	-31	-31	-37	-37
Plastic	-37	-37	-52	-52
Paper	-110	-110	-153	-153
Wood	-9	-9	-9	-9
Glass	-21	-21	-24	-24
Metals	-49	-49	-63	-63
Others	-36	-36	-53	-53
Residual Waste	300	220	262	190
Electricity (WtE)	-205	-142	-154	-103
Heat (WtE)	-42	-29	-32	-21

Fig. 6. Contribution of subsystems to the carbon balance of the I-WMS of Torino District (GWP_{fossil}).

tion and tended to overestimate them, or believed that impacts of collection and transportation might outweigh savings from SC and downstream recycling. The numerical results showed that the impacts of collection and transportation are much lower than environmental gains from recycling, which is in accordance with other studies (Merrild et al., 2012; Salhofer et al., 2007a). Moreover, it was pointed out that only part of the impact of collection can be ascribed to transportation (60% of energy and 75% of GHG), the remaining impact being associated with the manufacturing of waste bags and containers. Public administrators committed themselves to made data on collection available in future LCAs, but, at the same time, they agreed that having assigned the same transportation distance to all waste types in all scenarios would not substantially change the conclusions of the study.

A very recent paper by Merrild et al. (2012) confirmed that SC corresponds to a higher diesel use (4.1 l/t) than collection of residual waste (3.6 l/t). However, such a difference is relatively low. This substantially supports the position of environmentalist NGOs, according to which, although transport distances remarkably increase, SC is carried out with smaller and more efficient vehicles, which partially compensate impacts.

Another important discussion was that on ecological efficiency of recycling. The numerical results of Fig. 2 and the quantitative description of the recycling chains (reported as Supplementary content) where extremely helpful to show to the panel how recycling is modelled in LCA. Many were unfamiliar to basic concepts such as selection and recycling efficiencies and partially unaware that recycling avoid not only manufacturing, but also its upstream activities. Thus, while incineration can recovery part of the feedstock energy, recycling can recovery part of feedstock energy, but also direct and indirect energy (Boustead and Hancock, 1979). Such a discussion helped the panel to correctly interpret the LCA results and highly contributed to enhance their acceptability.

Table 3
Comparison among four alternative residual WM chains (Torino and Cuneo Districts).

Impact category	Unit	Chain 1	Chain 2	Chain 3	Chain 4
GER	MJ/t	−10,117	−8065	−6354	−3187
NER	MJ/t	−9933	−8080	−5805	−2902
$GWP_{100total}$	kg CO_2eq/t	90	149	906	686
$GWP_{100fossil}$	kg CO_2eq/t	−327	−229	178	180

At the end of the process, in fact, all participants expressed their satisfaction on how the LCA was conducted. Similarly to other cases reported in literature (Salhofer et al., 2007b), all the participants took the opportunity to express their opinion and support the LCA with their contribution, but none could influence the results to a large extent.

Bearing in mind that data were collected from plants and activities well representative of the study area, at the end of the discussion on SC and recycling, the panel agreed that the net environmental gains obtained in this study are not overestimated. On the contrary, the research highlighted that there is room for improving the eco-efficiency of the collection-recycling chain, which is not fully optimised (in the case of plastic, only 49.4% is effectively recycled, see Supplementary content). An ex post confirmation on the actual recycling efficiencies came from another LCA study in northern Italy (Rigamonti et al., 2009), where similar numerical results are reported.

4.1. Sensitivity analysis of Torino District LCA

A sensitivity analysis was used to address some of the most controversial issues raised by the panel of stakeholders. With reference to the pre-treatment of residual waste prior to incineration, the debate concentrated on the mass balance at the MBT plant,

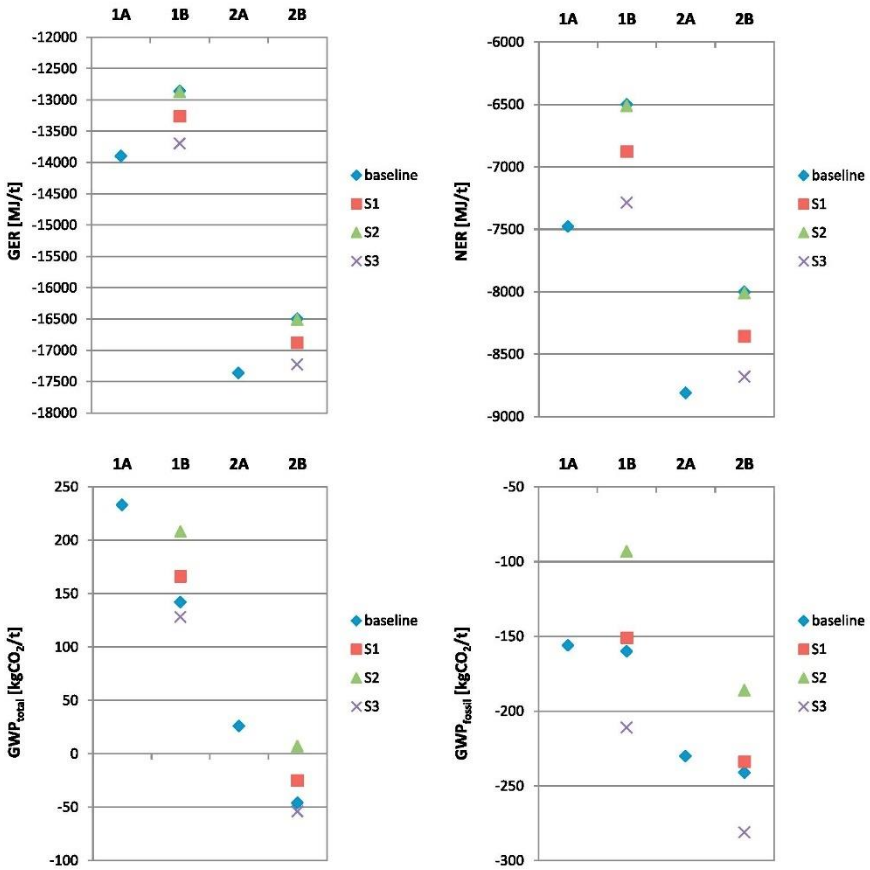


Fig. 7. Sensitivity analysis of Torino District LCA.

i.e. quantity and quality of light/dry waste and stabilised organic fraction, and the final destination of biowaste out of MBT.

Bearing that in mind, the baseline scenarios, i.e. the four scenarios (1A/B, 2A/B) summarised in Section 2.2 and described in the Supplementary content, were contrasted against the following sensitivity scenarios, where some of the key assumptions were changed.

- Sensitivity scenario S1: a different set of removal coefficients at the separation stage is considered (see Supplementary content). Consequently, mass balance and heat values of the light/dry fraction are recalculated, as well as carbon dioxide emissions;
- Sensitivity scenario S2: this is a pessimistic scenario where aerobic stabilisation of biowaste out of MBT is assumed to have a 50% reduced effectiveness and where biogas collection from landfill has a reduced efficiency (from 55% to 27.5%);
- Sensitivity scenario S3: this is an optimistic scenario were the organic fraction out of MBT is sent to AD, sludge from AD is sent to composting prior to landfill. Given the low quality of compost, no fertilizers substitution is accounted for.

The results of the sensitivity analysis (Fig. 7) helped understanding and interpreting the previously presented Tables and Figures. Note that in Fig. 7 scenarios that exclude MBT (1A/2A) remain unchanged.

With reference to the baseline scenarios, energy indicators highlighted that scenarios that includes pre-treatment plus incineration appear to be slightly less efficient (−5%) than scenarios with direct incineration. However, this can be ascribed to landfill without energy recovery of the biowaste fraction out of MBT. A possible improvement could be AD of biowaste out of MBT, which would bring scenarios A and B to a similar energy saving performance. In case of AD, also logistic, technical and economic aspects should be considered, which go outside the framework of the present study.

The carbon balance has emphasised that the pre-treatment of the residual waste sensibly improves the climate change impacts in comparison to incineration without pre-treatment. An important aspect is the dynamic of the carbon cycle of landfilled biowaste fraction after MBT and the actual efficiency of biogas collection. As GWP is concerned, the pessimistic scenario in the sensitivity analysis showed that scenarios A and B are nearly equivalent (scenarios B can be worse than A in case of GWP_{fossil}).

The sensitivity analysis highlighted therefore that the vagueness in the definition of the pre-treatment technologies and subsequent destination of the organic fraction are crucial factors that deserve further and more accurate investigation.

5. Conclusions

Detailed applications of LCA to integrated waste management systems are complex and the subsequent analysis necessarily reflects this complexity. Developing waste management strategies is a challenging task which encompasses several aspects that cannot be fully included in a LCA analysis. Moreover, the research programmes carried out for Torino and Cuneo Districts have once more confirmed that there are not preferable waste management solutions in terms of all the environmental and energy indicators.

The two LCAs summarised in this article confirmed that SC and downstream recycling is the most effective tool to improve energy efficiency and to lower environmental impacts. This conclusion was drawn after considering the whole sequence of activities, thus quantifying the eco-balance of collection, transportation, selection, recycling of the main waste flows and landfill/energy recovery from residues. This important site-specific conclusion confirmed that, under the local operational conditions, SC objectives according to the Italian national law in force (Dlgs.152/06) are consistent with an overall energy and environmental efficiency target. This was an important feedback for the public administrator involved in the research.

While the priority should be given to SC and subsequent recycling, energy recovery from residual waste also plays an important role. As residual waste is concerned, it must be remarked that Torino and Cuneo Districts have substantial differences in terms of existing/under construction WM infrastructures, which, at the time of the research, was a constraint limiting the possible WM scenarios to be compared. In fact, Cuneo District can take advantage of an existing cement factory, which is already authorised to co-incinerate RDF, whereas Torino District was going to build a new incinerator. In both cases, it clearly emerged that the eco-efficiency of energy recovery is much lower than the eco-efficiency of recycling. Based on the mass balances of the whole chain, and bearing in mind the site-specific data and local operational conditions, it appeared that co-incineration corresponds to better energy and carbon performances than dedicated incineration (with or without pre-treatment of residual waste). As far as energy and climate change issues are concerned, and according to the LCA results, an existing co-incineration plant should be preferred to a new incinerator. However, the research has also highlighted that the efficiency of the production of RDF plays an important role.

The research confirmed once more that the results of a LCA applied to an I-WMS are heavily influenced by site-specific aspects and local socio-economic constraints, and, therefore, should never be generalised. Consequently, the most important message from a WM LCA should not be the final results, but rather a combination of the results and the way LCA was conducted.

It was observed that stakeholders were extremely interested in actively contributing to the LCA, but under the condition to discuss the assumptions in details and agree upon the sources of data. Such a shared process highly contributed to the credibility and acceptability of the results. However, it was also observed that the vagueness in the definition of key elements in the I-WMS (in this case the pre-treatment of residual waste) can be an obstacle to the implementation of the LCA results.

Beyond site-specific conclusions, a general, more important, conclusion is that, without a deeper engagement of public administrators and stakeholders in the definition of case-specific methodological assumptions, LCA applied to WM will not easily become fully accepted and operational.

Acknowledgements

The Authors wish to thank all the public administrator of Torino and Cuneo Districts and all the members of the panel of stakeholders that actively participated in the research programmes.

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.wasman.2012.04.010.

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9. ANNEX D: DEVELOPING SPATIALLY STRATIFIED N(2)O EMISSION FACTORS FOR EUROPE

Environmental Pollution 159 (2011) 3223–3232

Contents lists available at ScienceDirect

Environmental Pollution

journal homepage: www.elsevier.com/locate/envpol

Developing spatially stratified N_2O emission factors for Europe

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ARTICLE INFO

Article history:
Received 4 November 2010
Accepted 21 November 2010

Keywords:
DNDC
Soil N_2O emissions
CAPRI
Stratification
GHG inventories
Europe

ABSTRACT

We investigate the possibility to replace the – so-called – Tier 1 IPCC approach to estimate soil N_2O emissions with stratified emissions factors that take into account both N-input and the spatial variability of the environmental conditions within the countries of the European Union, using the DNDC-Europe model. Spatial variability in model simulations is high and corresponds to the variability reported in literature for field data. Our results indicate that (a) much of the observed variability in N_2O fluxes reflects the response of soils to external conditions, (b) it is likely that national inventories tend to overestimate the uncertainties in their estimated direct N_2O emissions from arable soils; (c) on average over Europe, the fertilizer-induced emissions (FIE) coincide with the IPCC factors, but they display large spatial variations. Therefore, at scales of individual countries or smaller, a stratified approach considering fertilizer type, soil characteristics and climatic parameters is preferable.

1. Introduction

Nitrous oxide, N_2O, is an important greenhouse gas. Its major release pathway into the atmosphere relies on soil microbial processes, in conjunction with agricultural activities. Reactive nitrogen, usually added to soils to boost agricultural growth (fertilization), is exposed to conversion processes, specifically nitrification (formation of nitrate from ammonia) and denitrification (reduction of nitrate into molecular nitrogen). N_2O is a side product of both processes.

Release rates of N_2O are known to be spatially and temporally highly variable. Even though these two main processes are based on microbial activities which require opposite chemical regimes, i.e., nitrification occurs under aerobic conditions, while denitrification needs anaerobic conditions, highest N_2O emissions are observed for both processes at intermediate aeration (Firestone et al., 1979; Granli and Bockman, 1995; Groffmann, 1991). Hence, soil conditions like carbon content (soil organic carbon—SOC) and water availability play a key role. The huge variability is also reflected in the results of available field measurements. Attempts to single out driving parameters based on such measurement compilations have been made, but the reported uncertainties remain huge (Stehfest and Bouwman, 2006).

Likewise, a number of soil modelling approaches address N_2O release. Typically, such studies start on a plot scale to validate simulation results directly with plot scale measurements. A few of these models such as DNDC (Li, 2000; Li et al., 1992a,b), EPIC (Williams, 1995), DAYCENT (Del Grosso et al., 2009; Del Grosso et al., 2000; Del Grosso et al., 2006), or the fuzzy model described by Ciais et al. (2010) have been extended to also cover larger regions. Evaluation of such models focused on total emissions and on comparisons between the results from different models and between models and inventories.

Emissions of N_2O by source sector have to be reported to United Nations Framework Convention on Climate Change (UNFCCC) by the parties of that convention. Despite of the availability of models and (clustered) measurement data, these results are considered so unreliable that almost all countries resort in applying the simplest method available in the IPCC national emission inventory guidelines (IPCC, 1997, 2000, 2006). This method, based on the plot measurements mentioned, scales emissions according to a single emission factor indiscriminately to all possible sources of nitrogen input to soils. Being aware of the limitations of the approach, the uncertainty associated with these emissions is so high that it typically dominates the overall uncertainty of national GHG inventories (Leip, 2010; Winiwarter and Muik, 2010)

In this paper, we use the N_2O fluxes obtained from a well-validated model (DNDC-EUROPE, Leip et al., 2008) to consider the influence of three important factors: soil organic carbon content of the soils, fertilizer type (mineral fertilizer or manure), and weather

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doi:10.1016/j.envpol.2010.11.024

conditions (represented by different meteorological years). Each of these factors is assessed in terms of its impact to N_2O fluxes and fertilizer-induced N_2O emission fractions (FIE) at a national scale for 25 countries in the European Union (EU25), distinguishing between FIEs for mineral N fertilizer (FIE_{min}) and manure N inputs (FIE_{man}). The results are meant to allow, at a later stage, cross-validation with actual measurements in order to provide guidance on mitigation options beyond a generic reduction of nitrogen input.

2. Methodology

2.1. Overall approach

We use the process-based model DNDC-EUROPE (Li, 2000; Leip et al., 2008) to simulate direct N_2O fluxes from agricultural soils at a large number of spatial calculation units (about 200 000 for EU25) for different scenarios. Each unit is simulated for up to three crops and for ten different meteorological years. The results presented are based on a total of about 10 000 000 individual simulations. In the following we refer to individual spatial calculation units as 'spatial units' (SU); a simulated crop in an SU is a 'simulation entity' (SE). The SE are chosen such that they represent the major agricultural land use in Europe, based on the Agricultural Land Use Map for Europe (see Kempen et al., 2005; Kempen et al., 2007; Leip et al., 2008), which was adapted to the latest information contained in the CAPRI regional database (Britz and Witzke, 2008) for the base year 2002. Specifically, those crops occurring in an SU were selected that cover at least 10% of the agricultural area of the SU. We included all 31 annual crops included in the CAPRI database with the exception of permanent grassland and pastures. Different scenarios of nitrogen input are used to simulate the processes occurring in an SE. A scenario simulated for one SE within a specific meteorological year is referred to as 'individual simulation' or simply 'simulation'.

2.2. Definition of scenarios

A set of 15 scenarios was defined to assess the behaviour of DNDC-EUROPE under changing agricultural practices (see Table 1). A reference scenario contained estimates of input fluxes obtained from official statistics and downscaled as described below. Changes from the reference scenario with regard to the levels of nitrogen input are grouped into two classes. The 'Mineral fertilizer' scenarios (S01–S06) vary the input of nitrogen through mineral fertilizer in order to test the kind and magnitude of the response of N_2O fluxes to this nitrogen source. For these scenarios, the input of manure nitrogen and other N sources was kept as in the reference scenario. In analogy, the 'manure' scenarios (S07–S12) evaluated the response of N_2O fluxes to changing input of manure nitrogen leaving mineral fertilizer application as in the reference scenario. Scenario S06, has a specific interpretation ('extensification') as mineral fertilizer N input is set to zero and only manure N is applied to the fields without compensation for the mineral fertilizer N-input. Similarly, in S12 manure N input is set to zero and only mineral fertilizer N is applied to the fields. In order to limit computation to those SEs where a response is expected, a selection criterion was applied to the scenario groups 'mineral fertilizer' and 'manure'. Only those SE which have a nitrogen application rate of at least 20 kg N ha^{-1} yr^{-1} of mineral fertilizer or manure were included in the reference scenario.

2.3. Set-up of model simulations

Set-up of the simulations are done as described in Leip et al. (2008) and Britz and Leip (2009). Briefly, land use per spatial unit, actual and potential yield, nitrogen input (mineral fertilizer and manure), and irrigation (irrigated vs. not irrigated) are obtained from the CAPRI regional database (Britz and Witzke, 2008) and downscaled to the SU with the CAPRI-Spat model (Britz and Leip, 2009; Britz et al., 2010; Kempen et al., 2007; Leip et al., 2008). The model uses ground-truth observations from a Land Use/Cover Area Frame Statistical Survey (European Commission, 2003) to estimate the mean and variance for crop shares for each land cover class in the Corine 2000 database (European Topic Centre on Terrestrial Environment, 2000). Consistency with regional totals is achieved using the Highest Posterior Density technique (Heckelei et al., 2005). In addition to crop shares, the downscaling procedure encompasses among others (i) crop yield, based on information of potential yield and irrigation from Genovese et al. (2007) and Siebert et al. (2005), respectively; (ii) manure nitrogen application rates, taking into account nitrogen losses from animal housing and manure management systems, which are estimated according to the MITERRA-EUROPE (Velthof et al., 2009) and the Greenhouse gas–Air pollution Interactions and Synergies (GAINS) (Klimont and Brink, 2004; Winiwarter, 2005) models. Applied manure is characterized by its C/N ratio, which is obtained from CAPRI on the basis of the manure type (farm yard manure and slurry); (iii) mineral fertilizer nitrogen application rates, based on crop demands and nitrogen input from manure nitrogen, atmospheric deposition and biological N-fixation. CAPRI does not distinguish between different types of mineral fertilizer; in the DNDC model, application of urea and ammonium-nitrate is assumed in equal quantities. For each SE, DNDC-EUROPE simulations are calibrated by adjusting the value for the potential yield to its site-specific characteristics using the DNDC-CAPRI meta-model (Britz and Leip, 2009).

Table 1
Overview of scenarios.

Name	Description
Reference (S00)	Default scenario, land use and farm management as downscaled from CAPRI database.
Mineral fertilizer (S01-S06)	Input of mineral fertilizer nitrogen set to 1.25, 1.1, 0.9, 0.75, 0.5, and 0 times the of value in S00
Manure (S07-S12)	Input of manure nitrogen set to 1.25, 1.1, 0.9, 0.75, 0.5, and 0 times the of value in S00

Environmental input data required include top-soil characterization (pH, texture, bulk density and initial soil organic carbon (SOC) content) meteorological information (minimum and maximum daily temperature, and daily precipitation), and information on nitrogen deposition. Soil information is obtained from the European Soil Database (ESBD, 2006). The raster data (Hiederer et al., 2003; Jones et al., 2005) are calculated with the use of pedotransfer-functions taking into account land cover from the Corine1990 dataset. In order to avoid bias due to the different land use/cover used in this study (Corine2000), only matching agricultural land cover was selected from the ESDB raster map. To obtain a full coverage for the SUs, data gaps were filled by calculating a similarity index for each SU based on the classified information of the ESDB and missing data were filled with the values found in the SU with the highest similarity. Meteorological data were taken from Orlandini and Leip (2008). This consists of daily meteorological data at a resolution of 50 km × 50 km from the MARS grid weather (Orlandi and Van der Goot, 2003) combined with ATEAM/CRU data (interpolated monthly climate data at 10′ × 10′ spatial resolution, Mitchell et al., 2004) into a dataset covering the European Union plus Norway, Switzerland and Croatia at 1 × 1 km spatial resolution and daily temporal resolution from 1900–2000. Missing data in MARS were gap-filled through automated spatial and temporal interpolation. Nitrogen deposition is obtained from EMEP (2007) at a resolution of 50 km × 50 km using data for the year 2001.

Simulations were carried out for each scenario for the meteorological years 1990–2000 to reduce bias that can originate from specific meteorological conditions, also with respect to the simulated farm management which is not responsive to the current simulated weather. However, in order to enhance comparability between simulations made in different years, they were each time and for each scenario initialized at identical conditions and not treated as consecutive years of a simulated agricultural field. Thus each simulation represents the situation in the first year of the farm management according to the scenarios. The limitation to one year was necessary in order to keep the number of simulations manageable. The meteorological years of 1975–1989 were used to spin-up the soil conditions for each SU and the simulations were initialized with the archived soil conditions at the end of the spin-up run. Due to problems with this archive, some simulations were—randomly—not properly initialized for scenarios S1–S12 so that we had to discard about 30% of the simulations leaving us for the present analysis with about 6.5 million individual simulations.

2.4. Processing of data

N_2O fluxes were aggregated to all crop types, all simulation years as well as to the national level for 25 member countries of the European Union (Cyprus and Malta have not been simulated). As the focus of the current study is spatial variability and response to management factors rather than the generation of a N_2O inventory for agricultural soils, each SE has been given equal weight regardless of the crop share estimated with CAPRI-Spat.

For the mineral fertilizer and manure scenarios, we performed a linear regression (over all SEs) of the simulated N_2O flux versus N-input per hectare and year as mineral fertilizer and manure. The resulting parameters (number of 'observations', constant and linear regression coefficients, β_0 and β_1, and the coefficient of determination, R^2) were then aggregated over all simulated crop types and years as well as to the national and EU25 level.

Before further processing, simulations were assigned to one of four SOC-classes (low, medium, high SOC content and organic soil) with SOC content of <1%, 1–3%, 3–12% and >12%, respectively. As the DNDC model is not suited to simulate organic soils (Leip et al., 2008), results obtained on organic soils (>12% SOC) were not used in the analysis (about 1.5% of the SEs).

All processing of the data was done in the General Algebraic Modeling System (GAMS) language (V.22.6).

3. Results

3.1. Reference scenario

Average simulated N_2O fluxes for EU25 over all years and crop types are log-normally distributed (Fig. 1) with a geometric mean of

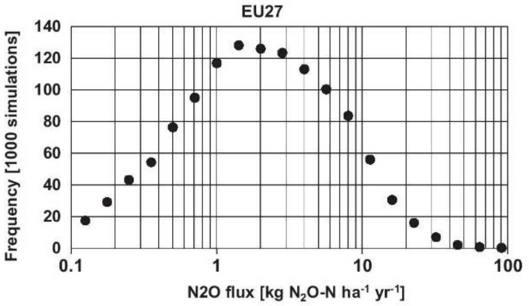

Fig. 1. Histogram of simulated N_2O fluxes on mineral soils in the reference scenario. The simulated fluxes were grouped into 30 classes on a logarithmic scale. Numbers are in 1000 simulations.

1.8 kg N_2O–N ha^{-1} yr^{-1} and a 68% confidence interval (CI; one standard deviation) from 0.5 kg N_2O–N ha^{-1} yr^{-1}–6.4 kg N_2O–N ha^{-1} yr^{-1} or a 95% CI (2 standard deviations) from 0.1 kg N_2O–N ha^{-1} yr^{-1}–22 kg N_2O–N ha^{-1} yr^{-1}. The average N_2O flux is 3.7 kg N_2O–N ha^{-1} yr^{-1}. Log-normal distributions are skewed, thus the confidence intervals are not symmetrical to the median (identical to the geometric mean) for a log normal distribution.

Soil organic carbon strongly influences the simulated N_2O fluxes (Table 2): average fluxes strongly increase going from low SOC content (1.0 kg N_2O–N ha^{-1} yr^{-1}, range at 95%-CI 0.1–4.4 kg N_2O–N ha^{-1} yr^{-1}) to medium SOC content (2.3 kg N_2O–N ha^{-1} yr^{-1}, range 0.2–12.3 kg N_2O–N ha^{-1} yr^{-1}) and high SOC content SEs (7.2 kg N_2O–N ha^{-1} yr^{-1}, range 4.4–37.2 kg N_2O–N ha^{-1} yr^{-1}). More than half of the SEs are on soils with a medium SOC content between 1% and 3% by weight, about 30% of the simulated SEs are on soils with a high content of SOC (3%–12%) and about 16% of the SEs are on soils with low SOC content (<1%). Accordingly, the weighted average of N_2O fluxes calculated for all mineral soils are between the values obtained for medium and high SOC content.

3.2. Comparison of data at country-scale

Mean N_2O fluxes at country-scale are shown in Fig. 2 for soils with low, medium, and high SOC content. The values scatter a lot across countries but also within each country. Note that for reasons of the underlying statistics, Luxembourg and Belgium are always presented as one entity here. Differences between the soil classes are high with highest mean N_2O flux on soils with low and medium SOC content in Slovenia, with 1.8 and 4.4 kg N_2O–N ha^{-1} yr^{-1} respectively, and for soils with high SOC content in Finland with 14 kg N_2O–N ha^{-1} yr^{-1}. The latter case (and the situation is similar for Estonia) may to some extent reflect the generally high SOC contents in those countries, as variation within this subcategory may be particularly high. Lowest per-area emissions, in all three cases, are attributed to Ireland, very low emissions also for Latvia

Table 2
Simulated N_2O fluxes [kg N_2O–N ha^{-1} yr^{-1}] for the reference scenario for the EU countries covered. The table reports the arithmetic mean (mean) of the simulated N_2O fluxes, the geometric mean or back-transformed logarithmic average (median), the lower and upper bound of the 95% CI (low and high), and the number of simulations (n).

SOO	Mean	Median	Low	High	n
All mineral soils	3.7	1.8	0.14	22.1	1 240 306
Low SOC	1.0	0.6	0.09	4.4	190 850
Medium SOC	2.3	1.4	0.16	12.3	652 446
High SOC	7.2	4.4	0.53	37.2	397 010

and the United Kingdom. The data are direct N_2O emissions from simulated SEs, and thus both differences in farm management and environmental parameters influence the level of N_2O emissions, however none of the parameters alone has significant correlation with N_2O fluxes.

3.3. Response to nitrogen input

One of the main objectives of the current study was to assess the response of simulated N_2O fluxes to changes in N-input rather than absolute N_2O flux values. This effect was studied by regressing simulated N_2O fluxes over scenarios where *ceteris paribus* mineral fertilizer or manure nitrogen application rates were changed between 0 and 125% of the value in the default scenario. A second-order polynomial regression gave very good representation of the response of the model with high coefficients of determination larger than 0.999. But even a linear model could explain more than 97% of the variance of N_2O fluxes across the scenarios. For simplicity reasons we used the linear regression model to estimate the rate of fertilizer-induced N_2O emissions (FIE)—the change of N_2O emissions as a consequence of change of fertilizer nitrogen application—for the application of mineral fertilizer (FIE_{min}) and manure (FIE_{man}) nitrogen. This concept neglects the fertilizer-independent component (emissions at zero N-input, or *y*-intercept in a regression analysis): this term comprises N_2O emissions from other N added to the system (N-deposition, N-fixing crops, N released through the mineralisation of crop residues or soil organic matter) as well as "back-ground emissions", a site-specific constant which occurs even in the absence of agricultural activities (Bouwman et al., 1995). The design of our simulations does not allow distinguishing between these sources of N_2O fluxes.

Fig. 3 shows the FIE_{min} and FIE_{man} for the 25 countries included in the simulation. According to the DNDC-EUROPE model, manure causes a slightly higher release of N_2O than mineral fertilizer. At European level, the difference between FIE_{min} and FIE_{man} is about 10% with higher FIE for applied manure than mineral fertilizer. FIE_{man}/FIE_{min} however increases with decreasing SOC content. For soils with low SOC content, FIE_{min} is only about 0.5%, but FIA_{man} is about 0.8%; on soils with high SOC content, FIE is around 1.8% regardless of the nature of the applied nitrogen. This effect can be explained by lack of anaerobic conditions in soils under dry meteorological situations with low SOC content, since the microbial activity necessary to deplete the oxygen is constrained by the lack of carbon substrate. As manure application adds carbon as well as nitrogen to the soil system, it enhances the carbon turnover processes with a higher rate of oxygen consumption and thus increased probability of the existence of anaerobic micro-sites which favour the production of N_2O, (Parkin, 1987; Smith, 1990; Smith et al., 2003).

It is interesting to note the differences between countries in Southern Europe vs. those of Northern Europe, with soils in Northern Europe generally exposed to more water, carbon content and lower temperature. Portugal, Slovenia, Romania and Italy have consistently high FIE on all soils and irrespective of the type of nitrogen. Countries like Denmark, Lithuania, Latvia and Ireland have low FIE throughout the considered cases. Notwithstanding, Denmark and Lithuania have N_2O fluxes that are with 2.9 and 5.1 kg N_2O–N ha^{-1} yr^{-1}, respectively, close or even above the European average indicating high back-ground fluxes.

3.4. Inter-annual variability

The differences observed in simulated FIEs point to a complex interaction of weather conditions, their impact on the soil-vegetation continuum and hence their impact on N_2O emissions. As noted, for all countries considered, FIE_{man} is on average about 10% higher than

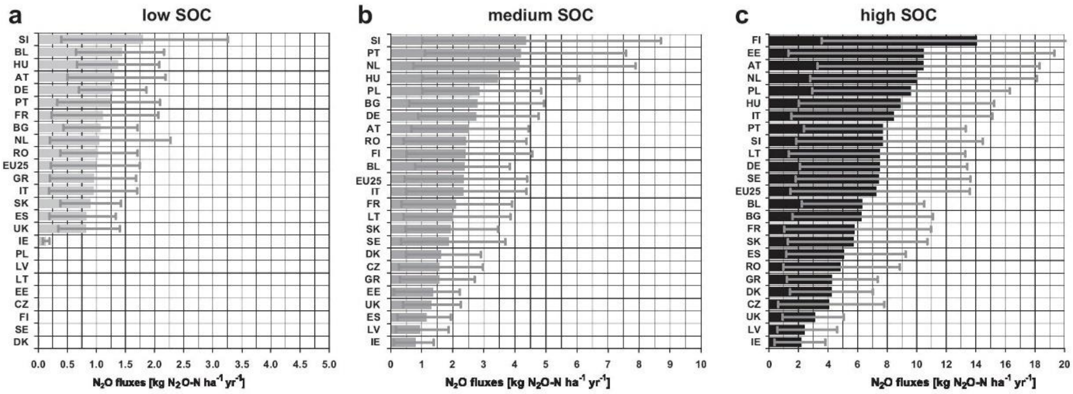

Fig. 2. Mean N_2O fluxes for 25 European Union countries (EU25, not including Cyprus or Malta) for the reference scenario on soils with (a) low SOC content; (b) medium SOC content and (c) high SOC content. The diagrams are sorted by the mean N_2O flux highest mean fluxes shown on top of the figure. In order to fit the error bars into the graphics, a range of ±1.03 standard deviation estimated on the basis of the log-transformed population has been chosen. This corresponds to a ±35% probability-range. Except for Belgium/Luxembourg, which are treated as a common entity, countries are indicated in the plot by their ISO country-acronym: AT: Austria, BG: Bulgaria, BL: Belgium and Luxemburg, CZ: Czech Republic, DE: Germany, DK: Denmark, EE: Estonia, EL: Greece, ES: Spain, FI: Finland, FR: France, HU: Hungary, IR: Ireland, IT: Italy, LT: Lithuania, LV: Latvia, NL: Netherlands, PL: Poland, PT: Portugal, RO: Romania, SE: Sweden, SI: Slovenia, SK: Slovakia, UK: United Kingdom.

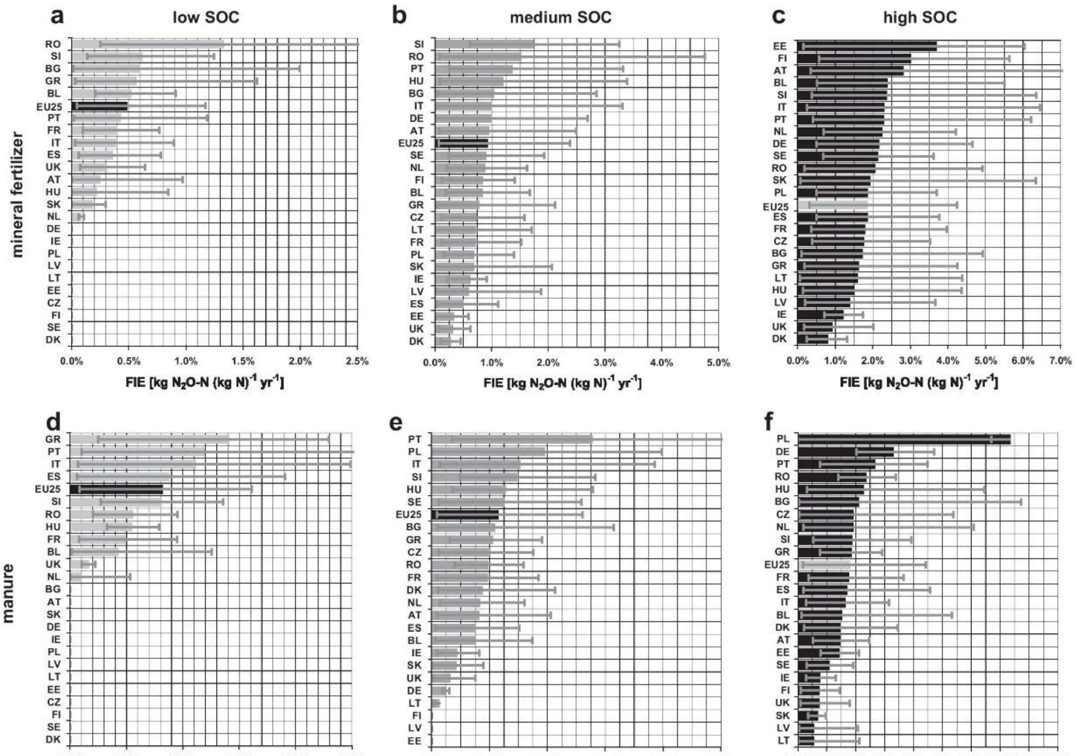

Fig. 3. Mean FIE of N_2O for 25 countries in the European Union for the reference scenario for the application of mineral fertilizer (a–c) and manure nitrogen (d–f). The panels show the FIE on soils with (a,d) low SOC content; (b,f) medium SOC content and (c,g) high SOC content. The diagrams are sorted by the mean FIE shown on top of the figure. The error bars indicate the 70%-CI. Countries are indicated in the plot by their country-acronym as given in Fig. 2.

FIE_{min}, but for individual years the difference can be as large as 30%. This is the case of 1994, a year with the highest average mean temperature during the simulated years (11.4 °C) and average annual precipitation of 690 mm. Yet in 1996, which experienced about the same mean annual precipitation as 1994 but had an average mean temperature of only 9.9 °C (the lowest in our dataset), FIE_{min} is larger than FIE_{man} by 23% (see Table 3). Microbial activities are reduced in cold temperatures, reducing also the occurrence of anaerobic micro-sites in dry soils because of the lower mineralisation rate of manure. Thus, the relative rate of N_2O formation from manure in dry soils with respect to N_2O formation from mineral fertilizer is lower in cold than in warm years, as mineral fertilizers are assumed to be less affected.

The situation is more complex looking at the individual countries (Fig. 4). The ranges of FIE_{min} and FIE_{man} generally overlap, but this is not the case for example in Greece or Denmark (FIE_{min} is smaller for all years) or Sweden or Slovakia (FIE_{man} is always smaller with a small overlap in Sweden). The ratio between highest and smallest FIEs ranges between 1.3 and 8.6 for FIE_{min} (Greece and Estonia) and 1.6 and 6.2 for FIE_{man} (Italy and Lithuania). Also the years which lead to particularly high or low N_2O emission factors are different but the data reveal some trends. Low FIEs are simulated in the first three simulation years (1990–1992) and, to a lesser extent, in 1995 and 1996. High fluxes are simulated in particular in the last five simulation years with a peak of 20% of simulated annual and national FIEs occurring in 1997.

In general, FIE_{min} is higher in countries characterized by cool/humid summers (AT, FI, EE, IR, LT, LV, SE, SK) while FIE_{man} is higher in countries with dry/hot summers (HU, PT, ES, IT, and GR, but also PL and DK which do not fall into the appropriate category). A similar pattern is found at EU25 level and we may argue this effect on a soil/climatic base. In wet soils, which are sufficiently rich in carbon to allow microbial activity, it is the availability of N that determines N_2O production. Mineral N is more readily available, leading to enhanced production. By contrast, dry soils under hot conditions will require carbon as microbial feed, favouring nitrification and denitrification from manure. Denitrification will further depend on water availability, which will determine the occurrence of anaerobic micro-sites. Soil wetness is more related to ambient temperature (especially during summertime) and summer precipitation than to annual precipitation. Summer conditions are usually a better indicator for microbial processes than annual averages; during summer, temperature is likely not to be the limiting factor for N_2O production and thus other factors increase in importance (see Conen et al., 2000). Nevertheless, Fig. 4 shows a clear temperature dependency of the FIE_{man}/FIE_{min} ratio. FIE_{min} is larger than FIE_{man} for countries that have mean summer temperatures below 14 °C (except for Denmark where sandy soils seem to need the organic substrate for N_2O production). The opposite is the case at temperatures above 15 °C (exception here is Romania), with FIE_{man} being larger. Indecisive results are presented for countries of summer temperatures in between. Possibly, in the complex pattern of soil processes, temperature might be used as a good proxy for soil conditions.

Table 3
Average FIE_{min} and FIE_{man} for 25 countries in Europe for 11 different meteorological years.

	FIE_{man}	FIE_{man}
1990	1.01%	1.09%
1991	1.02%	1.13%
1992	1.12%	1.38%
1993	1.20%	1.26%
1994	1.14%	1.50%
1995	1.03%	1.25%
1996	1.25%	1.01%
1997	1.21%	1.36%
1998	1.13%	1.24%
1999	1.31%	1.46%
2000	1.23%	1.15%
all years	1.15%	1.26%

Table 4
Mean fertilizer-induced emissions of N_2O over the 10 simulation years for all countries included in the dataset.

Country	FIE_{man}	FIE_{man}
Austria	1.5%	1.0%
Belgium	1.2%	0.9%
Bulgaria	1.2%	1.4%
Czech Republic	1.3%	1.4%
Denmark	0.6%	1.3%
Estonia	3.4%	1.4%
Finland	3.0%	0.7%
France	0.9%	1.1%
Germany	1.7%	2.6%
Greece	0.7%	1.2%
Hungary	1.3%	1.6%
Ireland	1.1%	0.7%
Italy	0.8%	1.5%
Latvia	1.2%	0.5%
Lithuania	1.3%	0.4%
Netherland	1.7%	1.5%
Poland	1.5%	4.1%
Portugal	1.0%	2.3%
Romania	1.6%	1.0%
Slovakia	1.2%	0.5%
Slovenia	1.8%	1.5%
Spain	0.6%	1.0%
Sweden	2.1%	1.1%
United Kingdom	0.5%	0.4%
EU25	1.1%	1.3%

4. Discussion

4.1. Variability vs. uncertainty

Despite the still large variability of results at the level of the individual SE and also at the different countries, we note that, for the total domain considered, results agree between individual years, with an inter-annual variability that is still large, but clearly reduced compared to the full dataset. Leip (2010) hypothesized that the uncertainty of national estimates of N_2O emissions are likely to be overestimated as current uncertainty ranges used are derived from the variance in experimental data which largely compensate. Similarity of large-scale model studies and the IPCC default factor confirm this hypothesis (Butterbach-Bahl and Werner, 2005; Del Grosso et al., 2005; Leip et al., 2008; Li et al., 2001).

Sampling of statistically independent information, as in measurements, typically yields a range of results. The variability of the sampled data is characterized by the standard deviation. It is obvious that the variability of the mean value calculated from these observations will become much smaller than the variability of the respective inputs as such. Statistical theory describes the standard deviation of the mean as the standard deviation of the distribution divided by the root of the number of elements. Thus the standard deviation of aggregated sets of data may be predicted from the original data. This resulting uncertainty will be clearly smaller than the variability of the individual samples.

The variance of individual simulations in the model was shown to be huge (Table 2). Stehfest and Bouwman (2006) observed a similar variance for plot measurements: mean 3.4; lower and upper bounds of a 95% central interval 0.001 and 22.45 and median 0.85 kg $N_2O–N$ ha^{-1} and measurement period, which for some plots is significantly smaller than a year. In their assessment, they were also able to narrow down the uncertainty thresholds

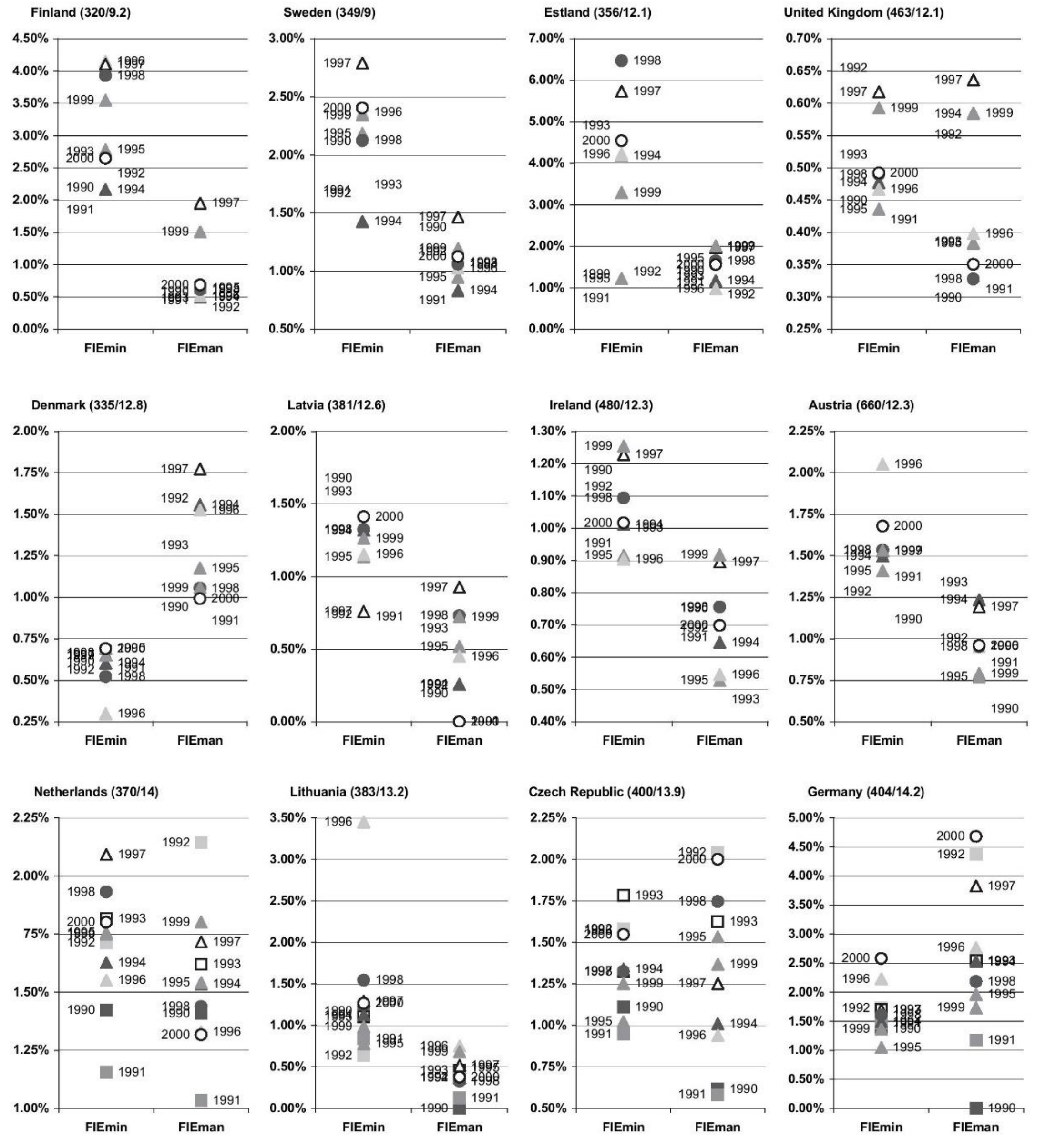

Fig. 4. Mean FIE_{min} and FIE_{man} for 25 European Union member countries over all soil types (mineral soils) and over all crop types for 11 different meteo-years (1990–2000). Countries included are identical to those presented in Fig. 2; Belgium and Luxembourg have been combined into one unit. In the figure, the countries have been clustered by summer precipitation (increasing precipitation from the top row to the bottom row) while within each cluster the countries are sorted by mean summer temperature (increasing temperature from the left to the right). Mean summer temperature (°C) and summer precipitation (mm) in our database are given in parenthesis. (summer values calculated here using data for the months April–September).

(95% confidence interval) to −51% to +107 of the estimated value by appropriate clustering of input data.

Also in aggregating the simulation results, we see that the difference in FIEs across countries (Table 4) or years (Table 3), even though still very large, is significantly lower than the variance in the individual simulations in the total dataset. For the purpose of this paper, we may consider for a moment the model results for individual SEs like measurements and the country-means as robust estimates for the national N_2O emission factors as used in the national greenhouse gas inventories to be annually submitted to

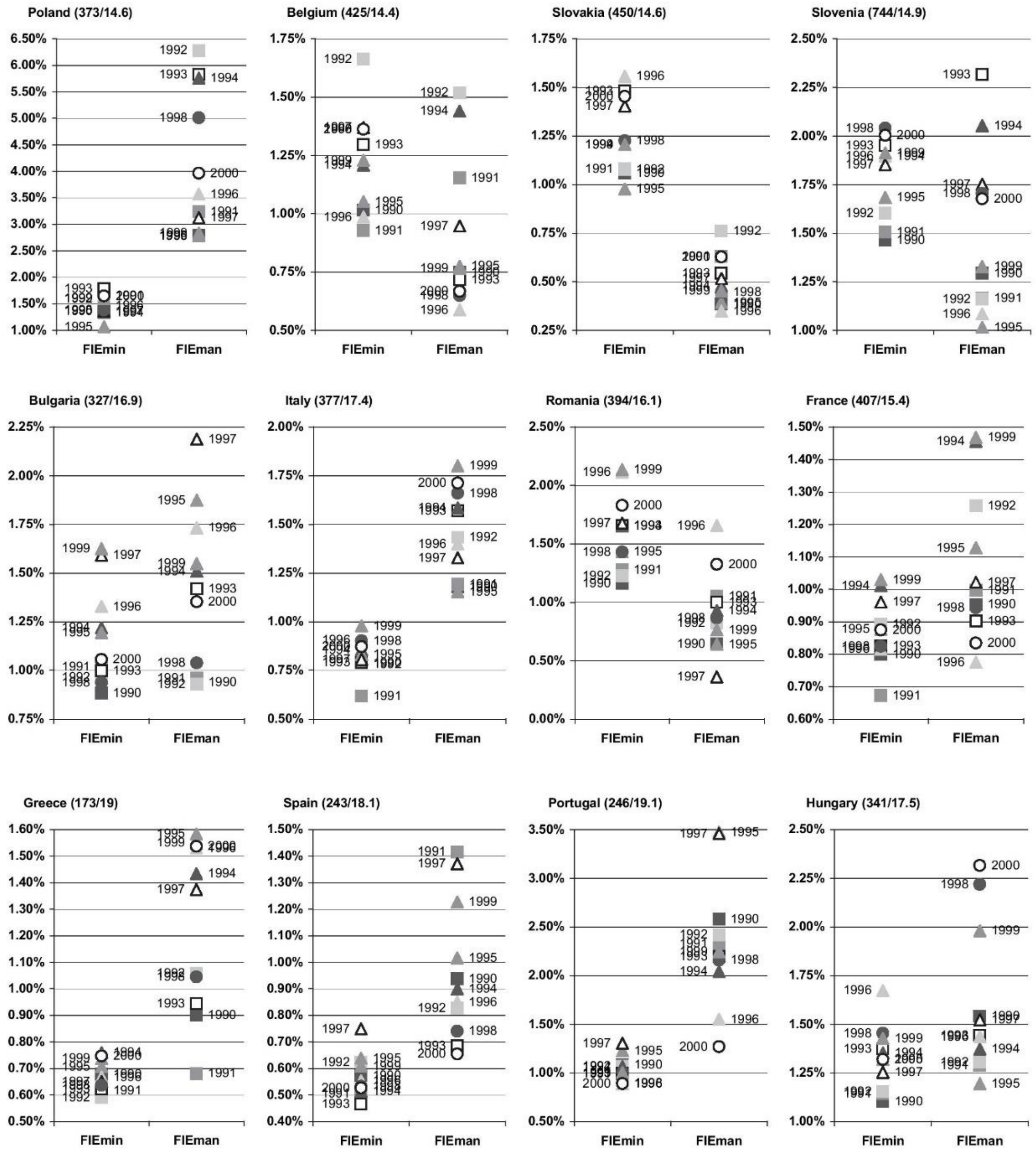

Fig. 4. (*continued*).

the UNFCCC. The range available for a country completely ignoring its own conditions would be from 0.4% to 4.1% of applied nitrogen, with a mean value of 1.3%, somewhat higher than the EU25 weighted averages of FIEs 1.15% and 1.25% for mineral fertilizer and manure nitrogen, respectively.

But on the basis of the real differences between clusters of data, which have been shown to impact the simulated mean FIEs between countries, such as soil and climate conditions, it is possible for a country to even narrow down the range of plausible mean national emission factors. Obviously, this property of the dataset needs to be treated for clustering rather than for ascribing an uncertainty which does not exist.

This discussion refers to model data only, and we need to make clear that also any conclusions at this stage are limited by the extent as the model data reflect the situation of real measurements. Nevertheless, the fact that the model results for the EU25 average is very close to the emission factor proposed by IPCC (1997) [1] is encouraging.

4.2. Comparison with UNFCCC

All the above analysis is based on the assumption that each SE represents an individual datum with equal weight in the subsequent calculations. For comparison with estimates made in GHG inventory calculations, a weighting of the simulated N_2O fluxes with the area of the SE has been done. Area-weighted mean N_2O fluxes for mineral soils are with 3.0 kg N_2O–N ha^{-1} yr^{-1} smaller than the un-weighted data of Table 2, indicating larger crop areas per SE in countries with lower N_2O fluxes. For the different soil classes we obtain 0.8, 1.9, and 6.3 kg N_2O–N ha^{-1} yr^{-1} for soils with low, medium, and high SOC content. The 95%-CI for mineral soils covers 0.14–14.4 kg N_2O–N ha^{-1} yr^{-1}. The resulting simulated emissions amount to ca. 200 Gg N_2O–N yr^{-1} or about 320 Gg N_2O corresponding to a radiative forcing of 100 Tg $CO_{2\text{-eq}}$ on the basis of a global warming potential of 310 kg CO_2 (kg $N_2O)^{-1}$.

This flux is obtained for a nitrogen input of 8.5 Tg yr^{-1} as mineral fertilizer and 4.2 Tg N yr^{-1} as manure, which is about 65% of the nitrogen input reported by the countries to the UNFCCC for the year 2002, for which the data on land use and farm management in the CAPRI database correspond (EEA, 2010). We scale by nitrogen input for the individual countries to extrapolate over the whole area.

Most countries use the IPCC (1997) default factor of 1.25% of N-input to calculate N_2O emissions (in the following referred to IPCC1997-factor). The method separately assesses N_2O emissions from crop residues and N-fixing crops, which is about 20% of the total direct emissions of 462 Gg N_2O. That figure does not contain direct N_2O emissions from histosols and 'other' sources of nitrogen input (mainly from the application of sewage sludge), which both are also not included in our model simulations. Moreover, national data do not contain indirect emissions as N_2O fluxes caused by deposition of nitrogen volatilised from the fields as ammonia or NO_x. For comparison, we thus need to subtract nitrogen input related to atmospheric deposition, and arrive at emissions of 465 Gg N_2O yr^{-1} from our simulations, very close to the national totals. Agreement by far is not as close for individual countries, as national data rely on a constant IPCC1997-factor while our analysis shows a large variability in FIEs.

Calculating an 'effective' implied emission factor (IEF_{eff}) from the country reports by dividing total direct N_2O emissions by the sum of N-input through manure and mineral fertilizer, we obtain 1.5% kg N_2O–N (kg N-input$)^{-1}$, which compares well with the simulated IEF_{eff} of 1.6% kg N_2O–N (kg N-input$)^{-1}$. Our data suggest that in Europe, a slightly smaller fraction of mineral fertilizer applied converts to N_2O than the IPCC1997-factor suggests, while emissions from manure are consistent with this factor. The overall slightly smaller FIEs are compensated in our model calculations by higher emission fluxes caused by nitrogen fixation, crop residues, or mineralisation of organic matter.

4.3. Stratified approach

Several authors suggested that N_2O emission factors stratified according to their main ecosystem characteristics would deliver more robust values with reduced uncertainty in comparison to the figures currently used in national inventories. Freibauer (2003) proposed a regionalised approach to calculate N_2O emissions differentiating between arable soils and grassland; for arable soils by two broad climate regions (temperate oceanic and Mediterranean on one hand and pre-alpine, alpine and sub-boreal on the other hand) and considering main soil parameters such as soil carbon and soil texture next to fertilizer-N input. Jungkunst and Freibauer (2005) proposed a pre-calibration matrix for ecosystem specific emission factors including climate zone, aeration status of the soil, intensity of management etc. Bouwman et al. (2002) and Stehfest and Bouwman (2006) developed a model based on a restricted maximum likelihood approach and on more than 1000 long-term field measurements of N_2O fluxes globally. Their model is based on the effect of five parameters, i.e., N application rate, soil organic content, climate, fertilizer type, and the length of the experiment.

Our results are in line with the main conclusions of these studies, indicating that (i) a stratification of emission factors helps in reducing apparent uncertainty by explaining part of it as the consequence of ecosystem factors; (ii) soil organic carbon is identified as the single most important factor explaining the magnitude of the FIEs, which is also confirmed by a number of modelling studies (see e.g., Butterbach-Bahl and Werner, 2005; Giltrap et al., 2008; Leip et al., 2008; Li et al., 2004; Mulligan, 2006). (iii) climatic factors are important drivers explaining N_2O flux rates, even though the causal relationship is complex, interacts with the soil properties and depends on the type of nitrogen applied (Bouwman et al., 2002; Freibauer and Kaltschmitt, 2003; Stehfest and Bouwman, 2006).

With regard to the differences observed for FIE_{min} and FIE_{man}, it became clear that the sign of the difference varies from country to country and that hence the overall result of slightly higher FIEs for manure than for mineral fertilizer nitrogen applications at the European level is a consequence of the spatial distribution of current agricultural activities. The result is therefore not necessarily in contradiction with the result of Davidson (2009) showing from a global assessment that the overall N_2O emission factors is 2.5% (range 1.6%–2.7%) for manure nitrogen and 2.0% (sensitivity range 1.7%–2.7%) for mineral fertilizer nitrogen. Note that, on an absolute basis, numbers cannot be compared as indirect emissions, by definition, will not be included in a plot-based model like DNDC, but have been covered (and cannot be separated) in the approach of Davidson.

Overall, our analysis suggests that, for EU25, the IPCC default factors give reasonable results for both mineral fertilizer and manure applications, but that stratified approaches would be preferable for assessments at smaller scales.

5. Conclusions

We present results from a dataset generated by a large number of simulations of N_2O fluxes with the biogeochemical model DNDC in order to assess the influence of three important factors: soil organic carbon content of the soils, fertilizer type (mineral fertilizer or manure), and meteorological conditions (represented by different meteo-years) at a national scale for 25 countries in the European Union (EU25). According to the DNDC model, there is a clear relationship between N-input and N_2O fluxes. Average fertilizer-induced emissions (FIEs) are 1.15% of mineral fertilizer and 1.26% of manure nitrogen. National FIEs are ranging from 0.5% to 3.4% of mineral fertilizer-N and 0.4%–4.1% for manure-N. Through the large coverage of the European agricultural landscape in our analysis, the data suggest that the high spatial variability observed both in measured and simulated N_2O fluxes does not translate in an equally high uncertainty of national or supranational emission factors. Instead, it is likely that national inventories tend to overestimate the uncertainties in their estimated direct N_2O emissions from arable soils. For an assessment at scales

[1] Note that the DNDC model used here can only assess direct emissions. Direct emissions are very similar for either IPCC approach (IPCC, 1997: 1.25% of N remaining on the field after correction for losses to the atmosphere; IPCC, 2006: 1% of total N applied).

as large as the EU25 a single emission factor for N_2O fluxes gives reasonable results and we find total direct N_2O emissions from mineral soils very close to those estimated in the latest national greenhouse gas inventories. Yet, a stratified approach considering fertilizer type, soil characteristics and climatic parameters is preferable at scales from individual countries in Europe or smaller. We find clear deviations (up to a factor of four) between countries in terms of area-based emissions, which can be explained in part by fertilizer input. More considerable differences are seen for FIEs. These differences do play a decisive role when considering appropriate abatement strategies. As presented now, the differences are currently model results only and reflect the current understanding of the underlying processes. Only field experiments can show if this process understanding is sufficient to extrapolate the effects presented here to the real situation.

Acknowledgements

The authors would like to thank the European Commission funded research projects NitroEurope IP and the DG-ENV funded LIFE project EC4MACS for supporting the research and collaboration underpinning the results presented in this paper.

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10. ANNEX E: ESTIMATION OF N2O FLUXES AT THE REGIONAL SCALE: DATA, MODELS, CHALLENGES

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Current Opinion in Environmental Sustainability

Estimation of N_2O fluxes at the regional scale: data, models, challenges

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Empirical and process-based models simulating N_2O fluxes from agricultural soils have the advantage that they can be applied at the scale at which mitigation measures can be designed and implemented. We compared bottom-up results from studies providing N_2O fluxes at a regional/country or continental scale with estimates from the process-based model DNDC-EUROPE and from the TM5-4DVAR inverse modeling system. While the agreement between different bottom-up models is generally satisfying, only in a few cases a thorough validation of the result was done. Complex empirical or process-based models do not appear to have a better agreement with inverse model results in estimating N_2O emissions from agricultural soils for countries or country-groups than simple ones. Both bottom-up and inverse models are limited by the density and quality of observations. Research needs to focus on developing tools that inherit the advantages of both methods.

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Current Opinion in Environmental Sustainability 2011, 3:1–11

This review comes from a themed issue on
Carbon and nitrogen cycles
Edited by Carolien Kroeze and Lex Bouman

Received 5 May 2011; Accepted 7 July 2011

1877-3435/$ – see front matter

DOI 10.1016/j.cosust.2011.07.002

Introduction

N_2O fluxes from agricultural soils are the most uncertain emissions source in GHG inventories submitted to the UNFCCC annually [1,2]. The reason is that N_2O fluxes are characterized by a very large spatial and temporal variability due to their strong dependence on environmental factors. Soil type, farm management including nitrogen additions, concentration of organic material in the soil, temperature and precipitation all influence the level of N_2O fluxes that are measured in the field [3,4•]. Furthermore, all these factors interact with each other. For example it is important whether rain events are close to the application of fertilizer or not [5,6].

As a consequence, N_2O fluxes are very difficult to predict at plot scale. The high temporal variation implies also that extrapolation of N_2O fluxes to annual emissions requires a large number of measurement to obtain a representative annual flux estimate [7]. For the whole of EU27, however, less than 650 flux measurements are available from around only 80 sites, which satisfy minimal requirements with regard to frequency and extension of the measurements [8].

Given the large variability of N_2O fluxes, models are important tools for extrapolation over larger regions, countries or even continents. The models should be able to estimate emissions based on spatially explicit input data, such as soil information and meteorological records, encompassing the spatial heterogeneity of the landscape. Various kinds of models have been developed for that purpose, including empirical models of different complexity [9–13], process-based models [14,15] and meta models based on applications of detailed process-based models [16,17]. The advantage of using these models is that they can be applied to any scale, depending on the availability of input data matching the requirements of the model. While only N-input scenarios can be tested with the IPCC model [11], more elaborated empirical and process-based models can be used to test a wide range of scenarios, for example, to assess possible mitigation measures. The extent to which this is possible depends, of course, on the responsiveness of the models.

Despite these advantages, the validity of upscaling bottom-up models to larger regions has never been systematically tested. So far, the validity of regional estimates has only been discussed through a good fit with measured data on plot experiments distributed over parts of the model domain and uncertainty is addressed with sensitivity tests or Monte Carlo simulations (e.g. [18,19•,20–25]). Butterbach-Bahl *et al.* [21] stress that even if the coverage of experimental data is sufficient for obtaining a robust regional estimate, it is still associated with considerable uncertainty. Most studies therefore conclude that the results are not suited for verification of N_2O emission estimates used in official greenhouse gas inventories submitted to the UNFCCC. Indeed, this can only

Table 1

List of studies estimating regional or national emissions of N_2O from agricultural soils. The table reports the main information on the studies and indicates whether or not the model used has been calibrated or validated for the particular study or whether a sensitivity study was performed.

Reference	Region (country)	Year	Simulated area [km^2]	Note	Emission [kg N_2O-N ha^{-1} $year^{-1}$]	SD	Approach	Calibration	Validation[a]	Sensitivity
Durandeau *et al.* [61]	Picardie (FR)	1995–1997	8726[c]	IPCC-EX IPCC-EN CERES-EX CERES-EN	1.97[c] 1.55[c] 1.29[c] 1.07[c]	– – – –	Biophysical soil-crop model CERES-EGC coupled with economic model, AROPAj. Response curves of N_2O emissions to fertilizer nitrogen inputs generated with CERES-EGC and linearized to obtain emission factors (Efs). Ef's fed into AROPAj relates farm-level GHG emissions to production factors	0	(x)	0
Lehuger [65]	Ile-de-France (FR)	2000	5629[b]	Corn Wheat	1.29 0.84	– –	Application of CERES-EGC model to Île-de-France region	x	0	x
Gabrielle *et al.* [25]	Chartraine (FR)	1998–1999	319	–	1.37	–	CERES-ECG model run with geo-referenced input data on soils, weather, and land use to map N_2O emissions from wheat-cropped soils in France	x	x	x
Garnier *et al.* [62]	Seine basin (FR)	–	36,390	Cropland	2.0		Compilation of published N_2O emissions in Europe and calculation of median flux rate for the land use classes cropland, grassland, forest	0	0	0
Neufeldt *et al.* [42]	Baden-Wurttemberg (DE)	2000	4670[b]	–	1.89	0.57	Economic model EFEM used to simulate crop and livestock productive systems and to provide inputs (crop area and fertilizer intensity) to DNDC model to calculate crop emissions	0	x	0
Butterbach-Bahl *et al.* [21]	Saxony (DE)	1996	8730[c]	–	5.60	3.40	DNDC and PnET-N-DNDC models linked to a detailed GIS-database to calculate N_2O emissions	0	(x)	x
Bareth *et al.* [58]	Württembergisches Allgäu (DE)	–	4300[b]	–	3.0	–	Soil-landuse information system to estimate regional N_2O flux. Emission potentials associated to distinct land use and climate using literature data or expert knowledge	0	(x)	0

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Table 1 (*Continued*)

Reference	Region (country)	Year	Simulated area [km^2]	Note	Emission [kg N_2O-N ha^{-1} $year^{-1}$]	SD	Approach	Calibration	Validation[a]	Sensitivity
Roelandt *et al.* [63]	Belgium	5 years	30,600	–	3.87	0.69	Statistical models MCROPS and MGRASS used to analyze the spatial and temporal variability of N_2O emissions from the agricultural soils of Belgium	0	0	0
		1996			3.67					
Van Moortel *et al.* [64]	Belgium	1996	13,500	Direct emission	5.57[c]	–	Application of IPCC 1996 methodology using Belgian statistical data for 1996	0	0	0
De Vries *et al.* [12]	Netherlands	2000	18,860	Grassland and cropland	7.44	–	Simple process oriented soil model predicting N_2O emission as a function of nitrification and denitrification, which in turn depend on the net N input and net N mineralization	0	(x)	0
				Arable	5.7					
				Grassland	12.7					
Brown *et al.* [60]	United Kingdom	1990	51,000[c]	Cropland	1.46[c]	–	DNDC model with UK specific inputs (UK-DNDC) with added ability to simulate daily C and N inputs from grazing animals and applied animal waste	0	x	x
				background	1.70					
Lilly *et al.* [39]	Scotland (UK)		74,000	Arable crops, cereals and roots	1.02		Upscaling measurements of emissions to derive annual emission rates for specific combinations of soil type, land management and fertiliser practices to the national scale. Emission rates were then spatially scaled to derive national figures through the use of a GIS modeling framework	0	(x)	0
Sozanska *et al.* [40]	United Kingdom	1991–2000	325,641[c]	–	3.90	4.5	Regression model within a GIS framework based on published N_2O data. Emissions calculated every 5 km^2 as a function of N input, water filled pore space, soil temperature and land use	x	(x)	x
Freibauer [45]	EU15	1995	1,302,643[b]	Cropland + grassland + farmed organic	3.39[c]	0.89	Emission factors and regression equations derived from European long-term measurements of soil emissions and experimental data on emissions from animal houses and manure management at real scale conditions	x	(x)	x

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Table 1 (*Continued*)

Reference	Region (country)	Year	Simulated area [km^2]	Note	Emission [kg N_2O-N ha^{-1} year^{-1}]	SD	Approach	Calibration	Validation[a]	Sensitivity
Boeckx and Van Cleemput [59]	EU15	1996	1,445,163[c]	Arable land	2.78		IPCC 1996 methodology	0	0	0
Ciais *et al.* [48]	EU-25	–	–		3.06[c]	0.45	Fuzzy logic model used to calculate direct N_2O emissions at EU-25 level	x	(x)	0
Stehfestand Bouwman [8]	EU27	–	980,000	Cropland	3.37	–	Statistical model based on summarized information from N_2O and NO emission measurements for agricultural fields and for natural vegetation	x	(x)	0

[a] Study has performed a validation on independent measurements available for the study area x or provided a comparison with other studies (x).
[b] Area used in our DNDC simulation.
[c] Calculated from paper.

be achieved by a double constraint of emissions using both bottom-up and top-down methods [26•]. In this context, the term 'bottom-up' methods is used for the approaches described above, while 'top-down' methods estimate emissions from measured mixing ratios at atmospheric monitoring stations and inverse modeling techniques, usually based on three-dimensional atmospheric transport (and chemistry) models. Such inverse techniques have been widely applied for the major GHGs, including N_2O [27–29,30•]. These atmospheric inversions integrate over large regions, initially providing emission estimates mainly on the global to continental scale, and more recently also on the regional scale, by using atmospheric transport models with relatively high resolution and continuous regional atmospheric measurements [27,30•].

In this paper we present a collection of bottom-up studies providing N_2O fluxes at a regional/country or continental scale and compare the results with estimates from the process-based model DNDC-EUROPE [31•] and with results of the inverse modeling study of Corazza *et al.* [27•] based on the TM5-4DVAR model. The objective of the paper is to examine reliabilities of both methods and identify regions where a high spread of model results indicate shortcomings in driving data, representativeness of available measurements or process description.

Review of available regional estimates of N_2O fluxes in Europe

Uncertainty in regional emission estimates is composed of four components: random and systematic errors in the input data, and random and structural errors in the model used. These uncertainties can sometimes be mixed since at the large scale, input data are often the product of a model simulation. The optimum level of complexity of bottom-up models is a compromise between a decrease of the structural error in the model and an increase in the parameterization error [32,33]. For processes that are non-linear and highly variable as this is the case for N_2O fluxes, often a higher complexity of the model is recommended with respect to simpler processes [34].

At the large scale very simple models such as IPCC, which use different sources of N-input as the sole explanatory variable, have been found to predict N_2O emissions that are in line with results from other methods [1,2,27,35,36]. At intermediate scales (regions to countries), however, estimates of total N_2O emissions remain very uncertain. A detailed literature review was performed selecting peer reviewed and EU localized studies reporting estimates of N_2O fluxes (Table 1). However, only a small number of EU based studies have attempted to estimate N_2O emissions at a regional scale.

A comparison of these literature data with results from the DNDC-EUROPE model [31•] is shown in Figure 1. Leip

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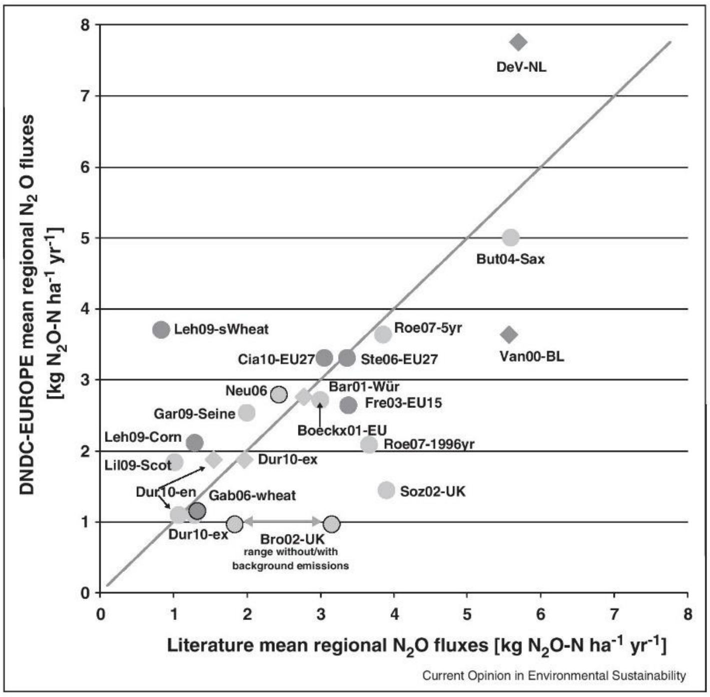
Figure 1



Comparison of regional estimates of N_2O fluxes versus the average N_2O flux rates for mineral soils simulated with the DNDC-EUROPE model [31]. The plot shows data as flux rates [kg N_2O-N ha^{-1} $year^{-1}$]. The shape of the point indicates whether default IPCC methodology (diamond) or another model (circle) has been used. Dark grey dots indicate that the model has been calibrated on regional data, while light grey dots indicate that no specific calibration for the study was undertaken. A black border around the dot indicates that some dedicated validation has been done. References: Bar01-Wür [58], Boeckx01-EU [59], Bro02-UK [60], But04-Sax [21], Cia10-EU27 [48], Dur10-en and Dur10-ex [61], Fre03-EU15 [44], Gab06-wheat [25], Gar09-Seine [62], Leh09-Corn and Leh09-sWheat (S Lehuger, PhD thesis, AgroTech Paris, France), Lil09-Scot [39], Neu06 [42], Roe07-1996yr and Roe07-5yr [63], Soz02-UK [40], Ste06-EU27 [8], Van00-BL [64], and DeV-NL [12]. For additional information see Table 1 and the text.

et al. [31•] presented results of data for the years 1990–2000 for all major crops, with the exception of grassland, using meteorological information for those years. It was thus possible to zoom to the region/period of the studies and compare the simulations with the same crop type or groups of crops. The results of the models derived from the literature are largely based on independent methods, even though in some cases also a model from the DNDC-family has been used [21,37], but in those cases the model-versions and input data were different. If N_2O flux rate per hectare was not directly provided by the study, it was estimated from the cultivated area, either obtained from the paper or as it was used in the DNDC-EUROPE model [23,31].

Despite the scatter in the data, there is a trend that DNDC-EUROPE tends to estimate lower N_2O fluxes than the more specific studies, particularly for the United Kingdom and Belgium. Most of the studies use more detailed input data than are available for a EU-wide study. For example, Roelandt *et al.* [38] provide data for the year 1996 and as average over five years. While simulations of DNDC-EUROPE match very well for the average value, it estimated much lower emissions using the meteo-data for 1996.

While many studies compare their results with previous estimates or with an estimate obtained using a simpler (often IPCC) methodology, only few studies have carried out a validation of the results. A sensitivity analysis is performed by six studies (see Table 1).

Brown *et al.* [37] compared results of UK-DNDC simulations with 16 datasets from contrasting conditions on a

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daily basis. The comparison was judged to be 'generally good', but the model sometimes simulated large emissions peaks which were not seen in the experimental data. The authors conclude that the low frequency of flux measurements makes it likely to miss events of high emissions and thus the validation of the simulation results is limited by the poor observational data. Almost half of their estimate is due to simulated background fluxes in the absence of nitrogen input. It is not clear how much of this is due to emissions from grassland. The data from Lilly *et al.* [39] and Sozanska *et al.* [40] suggest much higher than average emissions from areas dominated by intensively managed grassland.

Gabrielle *et al.* [25] present results of the CERES model calibrated with detailed data measured at three locations [41]. Simulations with a local parameterization were compared with simulations using a regional parameterization. Differences between −40% and +130% were obtained with respect to the local parameterization.

The most detailed validation of the studies included in our assessment has been made by Neufeldt *et al.* [42]. They took opportunity of the fact that Germany belongs to the countries with the highest density of annual observations in Europe [43] and were able to compare fluxes simulated with the DNDC model with a large number of long-term field experiments in Germany. The analysis showed a good agreement, but DNDC generally underestimated the fluxes and the authors applied therefore a scaling factor. DNDC-EUROPE gives, with 2.8 kg N_2O-N ha^{-1} $year^{-1}$ on the average, about 50% higher fluxes than those reported in the paper (1.9 kg N_2O-N ha^{-1} $year^{-1}$). However, if grassland and set-aside are excluded from the area-weighted average of the study, N_2O fluxes of 2.4 kg N_2O-N ha^{-1} $year^{-1}$ match reasonably well with results of DNDC-EUROPE.

Butterbach-Bahl *et al.* [21] compared crop-specific results obtained with the DNDC model for Saxony, Germany with measurements carried out in Central Europe. Even though the comparison shows a large scatter of the data, the authors conclude that the N_2O fluxes simulated with DNDC are well within the span of the reported N_2O emissions, which was particularly encouraging as the model had not been calibrated for this particular study.

Freibauer and Kaltschmitt [44] developed empirical models for N_2O emissions from agricultural soils that were used to estimate EU15 N_2O fluxes [45]. The comparison with plot data showed a large spread, but the order of magnitude of the fluxes was well captured and gave a better match with observations than the IPCC model [11] or the statistical model of Bouwman [46]. The approach of stratified models was further developed by Jungkunst *et al.* [47] showing that substantial benefit can be gained for distinguishing: first, water-logged soils vs. soils under redoximorphic conditions; second, presence/absence of regular soil frost conditions; and third, dry/wet climate zones.

The latter idea has also been taken up by Dechow and Freibauer [48][a] who developed a model based on fuzzy logic. Besides soil properties and management, the model implements seasonal weather conditions to account for the emission potential forced by soil moisture and freeze–thaw events. The model has been calibrated and validated on the emissions database described in Stehfest and Bouwman [8] and was upscaled to EU25 [48] using the CAPRI-Dynaspat input-database [23]. For EU25, the mean annual direct N_2O emissions match well with estimates reported to the UNFCCC, even though the distribution across countries is very different. The estimate by Ciais *et al.* [48], based on this approach, matches also very well with the estimate from Stehfest and Bouwman [8]; both models were developed on the same set of experimental data using different statistical methods. As the database compiled by Stehfest and Bouwman comprises virtually all available N_2O studies, any bias in this database will necessarily be reflected in these and other Europe-wide bottom-up model estimates. This confirms that high quality and representativeness of experimental data are the crucial prerequisites for meaningful upscaling to the regional scale.

Comparison of model estimates at country level

Recently, several Europe-wide models have been used to estimate greenhouse gas fluxes or fluxes of reactive nitrogen from agricultural soils. De Vries *et al.* [49•,50] provide a detailed comparison of input data and model results for four main models: INTEGRATOR [51], MITERRA [52], IDEAg [53], and IMAGE [54]. De Vries *et al.* [50] include in their comparison also results from GAINS [55], and data from EDGAR v4.0 [56] and UNFCCC [57]. The authors conclude that while data on N-input to agricultural soils are quite comparable, due to similar data sources, fluxes of reactive nitrogen show a relatively large scatter, in particular for N_2O and NO_x emissions and for N-leaching. Here we broaden the comparison of model results for N_2O emissions further and include in the comparison, next to EDGAR v.4.0, IDEAg, INTEGRATOR and UNFCCC, results from the DNDC-EUROPE model [31•], a new implementation of the Stehfest and Bouwman model (SuB) [8],[b] and estimates done with FISE, a model based on a fuzzy interference scheme [48].[a] Also, country-specific fertilizer-induced emission (FIE) factors have been calculated from the SuB and

[a] See also Dechow R, Freibauer A: **Assessment of German nitrous oxide emissions using empirical model approaches**. Nutr Cycl Agroecosyst, submitted for publication.

[b] Background information on the datasets used to calculate N_2O emissions with the Stehfest and Bouwman model are available at http://afoludata.jrc.ec.europa.eu/index.php/dataset/files/221.

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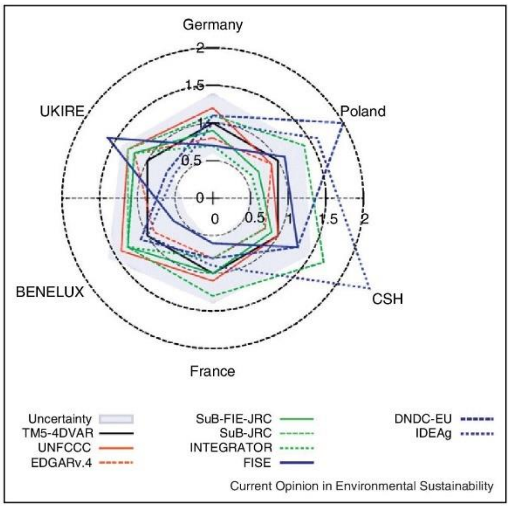
applied in the IPCC approach (SuB-FIE), substituting the general default direct emission factor for N input from mineral fertilizer, manure, and crop residues. The models cover different approaches, from simple emission-factor models (UNFCCC, EDGAR, SuB-FIE) over various empirical models (SuB-JRC, INTEGRATOR, FISE) to process-based approached (DNDC, IDEAg).

The top-down estimates are from the recent inverse modeling study of Corazza *et al.* [27•], based on the TM5-4DVAR inverse modeling system, using a horizontal resolution of $1 \times 1°$ over Europe. This study used 15 European monitoring stations (including several continuous monitoring stations) complemented by the global NOAA air sampling network. In general, the inverse modeling derives total emissions, but different groups of source categories can be separated in the inversion, if characterized by different spatial and/or temporal correlation properties. The *a posteriori* emissions presented here are optimized for four groups of emission sources, that is, soils, biomass burning, oceans and remaining emissions. Within these groups, emissions are assigned to more specific source categories using their partitioning in the *a priori* inventories. On the European scale, soil emissions are dominated by agricultural soils, while natural soil emissions are assumed to play a minor role. Based on a recent comparison of five independent models in the NitroEurope project, the overall uncertainties of total annual emissions are estimated to be in the order of 30–40% [27•]. However, in the context of this paper, an additional uncertainty is introduced for the estimate of the contribution of soil emissions on total emission (with emissions from agricultural soils estimated to contributing between 34% and 72% of total emissions).

In Figure 2, total N_2O emissions are compared for six countries or country-groups in Europe for which the inverse model was able to reduce the uncertainty. Estimates of total N_2O emissions by the different models are plotted relative to the estimate of TM5-4DVAR. An indicative uncertainty range has been drawn around the results from TM5-4DVAR, based on the share of emissions from agricultural soils and assuming that this source is more uncertain than the others. The plot shows three individual countries (Germany, France, and Poland) and three country groups, that is, Belgium, Netherlands and Luxembourg (BENELUX), United Kingdom and Ireland (UK_IRE), and Czech Republic, Slovakia and Hungary (CSH). The countries are aligned around a regular hexagon approximately according to their orientation in Europe. The underlying data to Figure 2 are reported in Table 2.

IDEAg and DNDC-EUROPE appear to over-estimate fluxes in Eastern Europe, while underestimating fluxes from UK and Ireland. Speculatively, this could be caused by the bias of available field measurements used for the

Figure 2



Plot of relative estimates of direct N_2O fluxes [Gg N_2O-N year^{-1}] from agricultural soils from eight bottom-up models as compared with data from the TM5-model in inverse mode [TM5-4DVAR, 27]. An indicative uncertainty range around the TM5-4DVAR estimate is indicated in light grey, derived from a total uncertainty of 40% and assuming that the uncertainty of non-agricultural emissions is half of that from agricultural sources. The plot shows data for three individual countries (Germany, France, and Poland) and three country groups, that is, Belgium, Netherlands and Luxembourg (BENELUX), United Kingdom and Ireland (UKIRE), and Czech Republic, Slovakia and Hungary (CSH). The bottom-up models are IPCC [UNFCCC [57]] and EDGARv4.0 [56], IPCC-approach using a factor for fertilizer-induced emissions from the Stehfest and Bouwman model (Sub-FIE-JRC, R Koeble *et al.*, unpublished), Stehfest and Bouwman [8] as implemented by JRC (SuB-JRC, R Koeble *et al.*, unpublished), INTEGRATOR [51], FISE [48], DNDC-EUROPE [DNDC-EU [31]], and IDEAg [53].

parameterization of the model toward Central European conditions with the consequence that co-varying factors, such as the content of soil organic carbon and climatic parameters, were not sufficiently resolved. As a consequence, the models perform very well in Central Europe across the north-south axis, but fail to estimate N_2O fluxes outside this axis in Eastern Europe and UK and Ireland. This is consistent with the findings of the comparison with regional studies shown above [37,40].

Deviations between *a posteriori* emissions in TM5-4DVAR and the FISE estimates (mean of 1990–1999) are most pronounced for the regions Germany, France and Benelux. Albeit German emissions are underestimated, the spatial emission pattern within this region characterized by higher emissions in West and South West of Germany is similar for both approaches. The spatial pattern is influenced by the pattern of precipitation and soil frost. Rewetting of dry soil

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Table 2

Estimates of direct N_2O fluxes from agricultural soils [Gg N_2O-N year^{-1}] by the different model approaches for the six countries or country-groups considered. See caption of Figure 2 and text for details.

	Germany	Poland	CSH	France	BENELUX	UK_IRE	Total
TM5-4DVAR *a priori*	43	27	13	49	12	40	184
TM5-4DVAR *a posteriori*	55	29	16	61	13	37	211
INTEGRATOR	39	18	13	51	18	43	182
EDGARv4.0	46	27	12	50	12	43	190
FISE	40	32	20	37	8	57	194
SuB-FIE-JRC	49	20	14	64	17	45	208
DNDC	59	59	21	46	15	26	227
IDEAg	55	48	38	53	14	22	229
UNFCCC	63	26	16	66	18	47	236
SuB-JRC	59	42	26	77	17	49	270

and freeze–thaw events might initiate emission peaks, contributing thus considerably to total annual emissions.

Both DNDC-EUROPE and FISE estimates rely on the meteorology of the time period 1990–1999 while the TM5-4DVAR emission are based on the meteorology of the year 2006. The variation of weather conditions between both periods might be partly responsible for the observed deviations. The good explanatatory power of N sources could be caused by a compensation of changing direct N_2O emissions with indirect N_2O emissions; a change in the share of direct/indirect emissions could be detected in the empirical and process-based estimates, but not with the emission-factor models or with TM5-4DVAR.

Most approaches tend to estimate lower direct N_2O fluxes in France which might be caused by the low density of plot scale measurements in France available for model calibration, but also the estimate of TM5-4DVAR for France is poorly constrained due to the lack of monitoring stations. Both UNFCCC and SuB-FIE-JRC results both very close to the TM5-4DVAR value. It needs to be stressed however, that there are differences in various parameters (e.g. direct emission factors, N input from crop residues) of the calculation chain of the two factor based approaches but they average out in the final value.

Looking at the whole area considered, we found that SuB-FIE-JRC, DNDC-EUROPE, IDEAg and FISE are closest to the *a posteriori* estimate of TM5, while UNFCCC and SuB-JRC tend to estimate higher, and EDGARv4.0 and INTEGRATOR tend to estimate lower overall fluxes (Table 2).

Conclusions

Estimation of N_2O fluxes at (sub-country) regional scale remains a challenge. We analyzed 17 studies that give regional estimates in Europe and found that only three provided a validation that could be considered as robust. Others lack data required for a good validation and discuss their results exemplarily on individual observations or just highlight the general performance of their model.

Complex empirical or process-based models do not appear to have a better agreement with inverse model results in estimating N_2O emissions from agricultural soils for countries or country-groups than simple ones. The reason is unclear and might be a combination of insufficient or not representative experimental observations to calibrate the models, the over-estimation of the sensitivity to certain important parameters or poor spatial input datasets. Process-based models have been used only recently for large scale simulations, but more progress is still required to improve their performance. On the other hand, also the inverse models need to be further validated, and their uncertainties better quantified.

It is of high importance to obtain regional independent estimates of N_2O fluxes at landscape level or above, for example using regional inverse approaches. Only with these data it will be possible to understand the factors that drive N_2O fluxes at the scale at which action is possible and mitigation measures can be designed and implemented. Large-scale independent estimates need to cover additional areas, such as Scandinavia and Southern Europe.

Because of a lack of a sufficient number of flux observations from agricultural soils, inverse approaches remain currently the only method able to verify national GHG inventories. Detailed bottom-up models however will remain essential in advancing our understanding in the current interaction between agricultural soils and climate, anticipating future challenges or assessing mitigation or adaptation measures. Research needs to focus on developing tools that inherit the advantages of both methods.

Acknowledgements

The authors gratefully acknowledge support from the European Commission for the NitroEurope Integrated Project and the CC-TAME project, underpinning the present study.

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- • of special interest

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Please cite this article in press as: Leip A, et al. Estimation of N_2O fluxes at the regional scale: data, models, challenges, Curr Opin Environ Sustain (2011), doi:10.1016/j.cosust.2011.07.002

ARTICLE IN PRESS

Waste Management xxx (2011) xxx–xxx

Contents lists available at SciVerse ScienceDirect

Waste Management

journal homepage: www.elsevier.com/locate/wasman

Eco-efficient waste glass recycling: Integrated waste management and green product development through LCA

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ARTICLE INFO

Article history:
Received 8 July 2011
Accepted 19 October 2011
Available online xxxx

Keywords:
Eco-efficiency
Recycling
Life cycle assessment
Green products
Industrial symbiosis
Waste glass
Foam glass

ABSTRACT

As part of the EU Life + NOVEDI project, a new eco-efficient recycling route has been implemented to maximise resources and energy recovery from post-consumer waste glass, through integrated waste management and industrial production. Life cycle assessment (LCA) has been used to identify engineering solutions to sustainability during the development of green building products. The new process and the related LCA are framed within a meaningful case of industrial symbiosis, where multiple waste streams are utilised in a multi-output industrial process. The input is a mix of rejected waste glass from conventional container glass recycling and waste special glass such as monitor glass, bulbs and glass fibres. The green building product is a recycled foam glass (RFG) to be used in high efficiency thermally insulating and lightweight concrete. The environmental gains have been contrasted against induced impacts and improvements have been proposed. Recovered co-products, such as glass fragments/powders, plastics and metals, correspond to environmental gains that are higher than those related to landfill avoidance, whereas the latter is cancelled due to increased transportation distances. In accordance to an eco-efficiency principle, it has been highlighted that recourse to highly energy intensive recycling should be limited to waste that cannot be closed-loop recycled.

1. Introduction

Due to the fact that many environmental and resource management issues faced by developed and developing nations alike are becoming highly uncertain, urgent, complex, and interconnected, we can no longer afford to address individual environmental and social problems in a convenient isolation of their context, or their spatial or temporal scale (Funtowicz and Ravetz, 2001).

Sustainable development (Bruntland, 1987) is both comprehensive and flexible, thus providing a framework for addressing complex problems through shared roles and responsibilities among the society as a whole and socially responsible companies (European Commission, 2001b; Shields et al., 2002). In such a framework, a growing number of companies are incorporating environmental sustainability in their business strategies, in order to integrate environmental concerns in their business operations and in their interactions with stakeholders (Albino et al., 2009; WBCSD, 1998).

Albino et al. (2009) have discussed the reasons that push firms to go "green", classifying them into three categories: legitimacy, competitiveness, and social responsibility. The authors also pointed out the strategic role of green products (i.e. goods or services that minimise their environmental impact over the whole life cycle) for sustainability-driven companies.

Due to the fact that environmental policies, especially at the EU level, are increasingly focusing on products, the attention of corporate environmental management has been shifting from processes (e.g. clean technologies) to products. The key role of green products in moving towards a 'new growth paradigm and a higher quality of life, through wealth creation and competitiveness' is clearly emphasised in the Green Paper on Integrated Product Policy (European Commission, 2001a). Product-oriented environmental policies offer at least two advantages: (1) they raise awareness that production is not the only source of environmental burdens, but rather production, consumption and post consumption play equally important and inter-dependent roles; (2) they foster shared responsibilities and roles between producers and their customers.

A general definition of a sustainable product could be: a product designed, manufactured, used and disposed of according to criteria

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doi:10.1016/j.wasman.2011.10.018

Please cite this article in press as: Blengini, G.A., et al. Eco-efficient waste glass recycling: Integrated waste management and green product development through LCA. Waste Management (2011), doi:10.1016/j.wasman.2011.10.018

of economic, environmental and social efficiency, which maximise net benefits across generations. However, it should be mentioned that there is still much confusion about what can be considered a sustainable product and what should not. Moreover, the terms "green", "eco" and "sustainable" are often used inter-changeably.

While a comprehensive classification of green products and the definition of their role within environmental policies, strategies and goals are out of the scope of this paper, it should be remarked that the following key issues are currently becoming commonly accepted (Blengini and Shields, 2010):

(1) Companies are increasingly using green product innovation in order to fulfil the environmental quality expectations expressed by their eco-responsible customers.
(2) Companies need reliable tools to make their environmental claims credible and distinguish themselves from firms which merely pursue market targets with green-wash packaging or advertising.
(3) Environmental burdens should be assessed and subsequently minimised throughout the whole product life cycle.
(4) Life cycle assessment (LCA) is one of the most important analytical tools to provide the scientific background for engineering solutions to sustainability, both during the design phase (eco-design) and during life cycle management.

In the above described context of green product development, a new recycling route has been implemented with the goal of maximising resources and energy recovery from post-consumer waste glass through integrated waste management and industrial production.

Life cycle assessment (LCA) has been used to highlight and quantify the eco-efficiency of such an innovative waste-to-production chain, with the objective of identifying engineering solutions to sustainability during the development of new building products to be used in energy efficient buildings (Blengini and Di Carlo, 2010). The purpose of applying LCA in this instance is threefold: quantifying environmental and energy savings and impacts, improving eco-efficiency and, finally, increasing the credibility of sustainability claims.

Both the new process and the related LCA have meaningful aspects that deserve discussion, as they are framed within a case of industrial symbiosis, where multiple waste streams are utilised as input in a multi-output industrial process. In other terms, the waste-to-recycling system under analysis can be considered a hybrid waste management – production system.

The green building product under development is a recycled foam glass (RFG). However, unlike current foam glass products, such as those described in Hurley (2003) and Scarinci et al. (2005), where waste glass is recycled in an open-loop fashion through a energy intensive process, the new waste-to-production route is based on a general eco-efficiency principle according to which RFG should be produced from the part of waste glass that cannot be closed-loop recycled. Hurley (2003) argued that cullet from glass packaging waste should be used in foam glass production only if it is heavily contaminated and not suitable for containers. Accordingly, in the recycling route that will be described in the next chapter, most of post-consumer container glass is not converted into RFG, but rather recovered, purified and sent back to the industries from which the waste originated, increasing eco-efficiency.

2. Materials and methods

The principal output from the innovative recycling route implemented under the EU Life + NOVEDI project by the Italian company SASIL (2009) is RFG, which is an artificial aggregate manufactured from waste glass (Fig. 1).

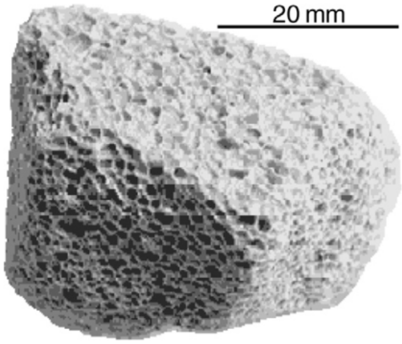

Fig. 1. Recycled foam glass (RFG).

However, the focus of the present paper is not RFG as material, a topic that has been deeply investigated and reported on in the literature (Bernardo and Albertini, 2006; Bernardo et al., 2010, 2005; Fernandes et al., 2009; Herat, 2008; Hurley, 2003; Lebullenger et al., 2010; Méar et al., 2005; Scarinci et al., 2005; Yot and Méar, 2011). Rather the focus is on gaining a better understanding of the eco-efficiency of the proposed recycling route, and the way LCA can be used to measure eco-efficiency and support green product development.

Table 1 reports the main physico-mechanical properties of the RFG that SASIL currently produces as a loose aggregate, which is graded in two ranges of particle size (0–8 mm and 8–16 mm) and shows a good mechanical strength (crushing test 0.62–5.2 N/mm^2 according to the Standard UNI EN 13055-1). Table 1 also shows a selection of mineral based insulating materials for which LCAs are publicly available (Ecoinvent, 2007; Lavagna, 2008; Pittsburgh Corning Europe, 2007). It must be said that, due to commercial confidentiality and limited information in the public domain, there is little availability of detailed and transparent LCAs of foam glass products.

This RFG is intended to be used for several applications in the building sector, where energy saving and resource efficiency are regarded as key issues. Thanks to the combination of low density, low thermal conductivity and good mechanical strength, RFG can be employed in light-weight concrete with good thermal-insulating properties. These RFG-based concrete products, which are presently under testing in co-operation with SASIL SpA, Italcementi Group and the Politecnico di Torino, are expected to open the way towards new engineering solutions for energy efficient buildings. One of these end-uses is represented by mono-material building envelopes that, beyond enhancing energy saving during the operational phase of buildings, are expected to increase recyclability of the building as a whole.

The production of foam glass dates back to the 1930s. It was originally manufactured from a specially formulated glass composition using virgin glass only. Since then, foam glass producers have steadily increased the quantity of post consumer waste glass in their product up to 98% (Hurley, 2003; Scarinci et al., 2005). Some of these RFGs are currently traded as green building products and their environmental claims are often self-declarations based on their status of recycled materials, which avoid waste landfilling and save non-renewable resources.

A granular RFG similar to the one presented in this paper was produced in Switzerland in the 1980s by Misapor AG (www.misapor.ch) in order to provide an alternative market outlet for

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Table 1
Characteristics of the RFG and other mineral-based insulating materials.

		RFG	Expanded clay	Expanded perlite	Expanded vermiculite	Foam glass
Type of product		Granulate	Granulate	Granulate	Granulate	Slabs
Density (slab)	kg/m^3	210–240	–	–	–	110–165
Density (bulk)	kg/m^3	160–180	260–500	80–130	70–140	–
Thermal conductivity	W/(Km)	0.07–0.09	0.08–0.13	0.04–0.07	0.06–0.08	0.04–0.06
Temperature of expansion	°C	900	1200	870–1090	1250–1500	850–1250

separately collected waste glass. Although the foaming process adopted by SASIL is similar to the one developed by Misapor AG (i.e. continuous production of sheets of foam glass that are then broken into loose foam glass aggregate and sized), the principal difference is the sorting process. Unlike most of RFGs, where recycling into building products represents an alternative to closed-loop recycling, the new RFG is produced from a by-product of conventional closed loop recycling, i.e. the waste glass that is rejected by container glass recyclers after their sorting process (between 6% and 15% in Italy), and which is enriched in contaminants (mainly metallic, ceramics and plastic scraps). The principal route for post-consumer soda-lime glass from municipal solid waste separate collection therefore remains recycling into container glass.

Since such a rejected glass is made mostly of glass and ceramic fragments/powders and also contains metals and plastic scraps, SASIL has developed an innovative recycling process that allows the separation of waste streams and makes it possible to sell purified materials back to the industries from which they were generated. RFG is therefore produced from the waste that has previously been rejected by container glass recyclers and, after the sorting step, is not used in glass, ceramic, brick or metal works, or recovered in waste-to-energy facilities.

In a mix with soda-lime glass, SASIL also uses special glass, mostly composed of monitor glass, glass fibres, and glass containing heavy metals that are presently landfilled. Due to the present shift of technologies in the TV/PC monitor industry (plasma, LCD), the problem of CRT (cathode-ray tube) glass disposal is in fact of particular concern. Alternatives for recycling have been reported in the literature (Dondi et al., 2009; Méar et al., 2006; Menad, 1999), including use of CRT glass in RFG production (Bernardo and Albertini, 2006; Bernardo et al., 2005; Méar et al., 2005). Particular attention has been paid to the resistance of RFG to possible leaching of barium and lead, typically contained in the front (Ba) and in the back (Pb) of the screen (Musson et al., 2000; Yamashita et al., 2010; Yot and Méar, 2011). Leaching tests run according to the standards UNI 10802:2004 and UNI EN 12457-2:2004 showed that the potential releases are below the thresholds, being 3–15 μg/l in case of lead and 20–40 μg/l for barium, which is in accordance with the findings of Bernardo et al. (2005).

As far as the environmental management of the recycling route and the role of LCA are concerned, the first environmental gain is the avoidance of waste landfilling, which leads to a saving in terms of waste dump space: a scarce resource nowadays in a country such as Italy. At the same time, sorting of waste glass prior to RFG production allows for the separation and recovery of glass, ceramic, metal and plastic, which improve the overall eco-efficiency.

All the above listed environmental gains are clearly perceived by SASIL, who intends to use them as the basis for environmental claims and green marketing. But, the open question is how the sustainability performance of the new green product can be quantified and communicated in a credible way?

Beyond the environmental gains, recycling is responsible for environmental impacts due to use of additives and fossil fuel in an energy intensive thermal process and it might increase transport-related impacts. Moreover, selection and material recovery efficiencies can lead to lower the overall efficiency of the process (Rigamonti et al., 2009). Consequently, induced impacts might outweigh environmental gains, thus rendering self-declared environmental claims less credible, or even false (Blengini and Garbarino, 2010).

From the previous discussion, it clearly emerges that the environmental profile (eco-profile) of RFG is the final result of a complex and inter-dependent waste-to-production system. LCA is therefore used to outline the eco-profile of RFG, because it is an analytical tool able of capturing complexity and inter-dependencies.

2.1. Definition of a LCA methodology to support environmental claims of RFG

The LCA methodology according to ISO 14040 (2006) and ISO 14044 (2006) has been used in order to capture the multiple environmental gains and the environmental impacts of the waste-to-recycling system under analysis.

The main advantage of using LCA is the possibility of assessing the environmental performance of products throughout their life cycle with a comprehensive perspective. However, there are some specific aspects that must be considered when dealing with waste management (Ekvall et al., 2007; Martínez-Blanco et al., 2010). In fact, when applying LCA to waste management systems, the from-cradle-to-grave and from-cradle-to-gate philosophies, typically adopted when dealing with production systems, must be turned into from-gate-to-grave or sometimes from-gate-to-cradle. In fact, as far as waste management is concerned, the input material is waste, which can either be sent to landfill or re-enter further life cycles in substitution of virgin materials.

Substitution means avoidance of products manufactured from primary resources through secondary materials gathered from recovery and recycling. In other terms, the production of a recycled material that re-enters further life cycles represents a potential credit for avoiding the production of an equivalent quantity of virgin products. The system that recycles the waste into a valuable product is credited with the environmental burdens of the corresponding primary production, but is charged with energy and ancillary materials used for the recycling process. This is currently called system expansion (Finnveden, 1999).

In this research, system expansion has been used in order to associate the multiple benefits of the new recycling process to RFG. Thus, net environmental gains relevant to glass and ceramic fragments/powders recovery, metal scrap recycling, plastic scrap energy recovery and landfill avoidance were allocated to RFG. This is a way to make visible the benefits achieved through industrial symbiosis: one system transfers environmental gains to the other.

2.2. Selection of environmental indicators

Bearing in mind that the choice of indicators and methodologies to express the results of an LCA is a subjective step, the research partners acknowledged the importance of supporting the environmental sustainability claims with sound, objective, understandable and internationally recognised LCA indicators.

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A combined top-down and bottom-up approach has been used in order to adopt a meaningful set of indicators. A top-down approach can be described as one that selects indicators that are representative of broadly recognised areas of environmental concern, as well as based on various international conventions, agreements and guidelines. This type of approach is indeed consistent with the recommendations of ISO (2006). In contrast, a bottom-up approach can be defined as one that identifies indicators based on the interests of the industry, public administrators or stakeholders and/or data availability.

This said, a first set of five mid-point indicators has been identified according to the above mentioned top-down approach (Boustead and Hancock, 1979; SEMC, 2000): Gross Energy Requirement (GER), Global Warming Potential (GWP), Acidification Potential (AP), Eutrophication Potential (EP) and Photochemical Ozone Creation Potential (POCP).

According to the bottom-up approach, in order to further assist decision makers with a simplified overall judgement across areas of environmental concern, the above mid-point indicators were complemented with the Eco-Indicator 99 H/A (Goedkoop and Spriensma, 1999). The latter is based on the so called damage-oriented (end-point) approach, and is aimed at evaluating the environmental implications for human health, ecosystem quality and depletion of non-renewable resources. It must be remarked that, although worldwide used, Eco-Indicator 99 involves both physical and social aspects and introduces subjective value choices and uncertainties that render it not fully consistent with the recommendations of ISO.

2.3. Functional unit, system boundaries and data sources

According to ISO 14040, the functional unit (FU) is a quantified description of product systems' performances. In the case of RFG, the selected FU should provide a reference to which the inputs and outputs can be related, thus allowing comparison among final products. However, RFG-based products are still under development and thus, at this stage, the analysis must be restricted to the environmental profile of RFG as granular material. Consequently, the adopted FU is 1 tonne of RFG aggregate.

The selected FU allows outlining the eco-profile of RFG as material, which will be the necessary background knowledge for subsequent comparative LCAs of building products or engineering solutions to energy saving in the built-environment. Nevertheless, a rough comparison can be based on thermal insulation properties of RFG against other building materials. An example is reported in Ardente et al. (2008) where the chosen functional unit was the mass unit of insulating board with a given thermal resistance R (measured in m^2K/W). It must be remarked that, in a full life cycle perspective, the selected FU would likely emphasise the use phase of buildings insulated with RFG, while it might underestimate some important RFG peculiarities such as lightness, fast assembly and recyclability.

With reference to the insulating materials reported in Table 1, it should also be said that a direct comparison with RFG granulate might be partially (or totally) misleading. In fact, RFG, expanded clay, expanded perlite and expanded vermiculite are granular insulating materials that might be used inter-changeably for some, but definitely not for all, possible end-uses. For instance, as far as concrete is concerned, expanded perlite and expanded vermiculite, which show good thermal insulating performance, have unacceptably low mechanical strength. On the contrary, expanded clay, which shows a mechanical strength similar to RFG, has higher thermal conductivity. Finally, the foam glass (last column of Table 1) that is described in the Ecoinvent database (2007) shows some important analogies with RFG, the most evident of which is that it is produced using nearly 70% of post-consumer glass. However, it is produced with a different process and traded in slabs and pre-cut shapes (Pittsburgh Corning Europe, 2007), therefore intended for different end-uses. With these limitations in mind, a comparison in terms of environmental indicators will be given in the next chapter.

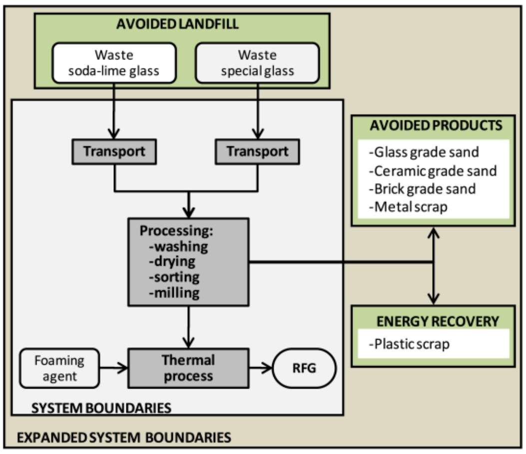

Fig. 2. System boundaries of the RFG waste-to-production chain.

As system boundaries are concerned, these encompass (Fig. 2): waste glass collection and transportation, processing (i.e. washing, sorting, drying and milling), production of additives and the thermal process.

As will be described in the inventory analysis, in the LCA model waste glass collection corresponds to landfill avoidance, while sorting corresponds to the recovery of commercial granulate materials, which in turn displace primary production of silica sand (glass grade sand and ceramic grade sand) and clay (brick grade sand). Moreover, sorting corresponds to avoided production of primary metals and fossil fuels.

As far as data sources are concerned, primary data were supplied by SASIL SpA, while secondary data were retrieved from the database Ecoinvent 2 (2007). The LCA model has been implemented with the software SimaPro (PRé Consultants, 2006).

2.4. Inventory analysis

RFG production can be schematically divided into the macro-processes shown in Fig. 2. Waste soda-lime glass and waste special glass are collected and transported to the plant (waste glass collection) where glass is separated from other components and transformed into a powder through a sequence of processes (washing, sorting, drying and milling). RFG grade powder is then mixed with a foaming agent prior to undergoing a thermal process where the foam glass expands.

The sequence of industrial processes (or avoided processes) in the waste-to-production chain is briefly described in the following paragraphs, with emphasis on input data used for LCA modelling. Unless otherwise specified, data refer to 1 t of input glass waste.

2.4.1. Glass landfilling (avoided)

Environmental gains related to landfill avoidance depend on the type and origin of waste glass, which can be grouped into two main categories:

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Table 2
Classification and average composition of waste glass used for RFG production.

EWC code	Glass type	Description	%
19 12 05	Soda-lime	Glass (waste glass rejected by post-consumer glass recyclers)	90.1
10 11 03		Waste glass-based fibrous materials	3.8
10 11 12		Waste glass other than those mentioned in 10 11 11	3.3
10 11 10		Waste preparation mixture before thermal processing, other than those mentioned in 10 11 09	0.8
20 01 02	Special glass	Glass (post-consumer special glass)	0.5
6 05 03		Sludge from on-site effluent treatment other than those mentioned in 06 05 02	0.3
10 09 08		Casting cores and moulds which have undergone pouring other than those mentioned in 10 09 07	0.4
10 09 12		Other particulates other than those mentioned in 10 09 11	0.8

– Soda-lime glass; including flat glass, windshield glass, light bulbs, tableware and containers.
– Special glass; glass with different compositions used for special applications such as coloured glass, tempered glass, hard glass, laminated glass, UV glass, fibre glass, optical fibres and various physical, chemical and industrial applications.

Table 2 shows the origin and the percentage of waste glass collected by SASIL according to the European Waste Catalogue (EWC). Soda-lime glass (EWC 19 12 05) refers to waste glass that is rejected by container glass recyclers after their sorting process. According to company measures, although quite variable over time, the average composition of rejected cullet is: glass (94%), plastics (2%), paper (1%), ceramics (2%), metals (0.5%) and organic compounds (0.5%). Special glass (all other EWC codes in Table 2) usually contains little non-glass material and its composition is rather constant over time.

According to Italian legislation (Ministero dell'Ambiente, 2005), unless recycled or reused, waste glass must be disposed of in an inert waste or non-hazardous waste landfill.

As far as the LCA model is concerned, it has been assumed that EWC 10 11 10, 10 11 12 and 06 05 03 would partially be disposed of in an inert waste landfill (50%) and partially (50%) in a non-hazardous waste landfill, while other EWC codes would be sent to inert waste landfills. Inventory data necessary to quantify the avoided environmental impacts were retrieved from the Ecoinvent database: "Disposal, glass, 0% water, to inert material landfill" and "Disposal, inert material, 0% water, to sanitary landfill".

2.4.2. Waste glass collection

Waste materials are transported from waste management facilities to the RFG production site. Company data were analysed in order to establish average routes and calculate average distances. Glass with EWC code 19 12 05 is transported on 30-tonne payload trucks for 246 km (weighted average). However, this distance must be reduced according to the avoided average transportation to the landfill facility (80 km). Thus, the net collection distance is 166 km. The same procedure has been used to estimate transportation of all other EWC codes, obtaining an average distance of 179 km. Road transportation by trucks has been modelled using the Ecoinvent datasets.

2.4.3. Washing

With respect to water consumption, input waste glass washing is a quasi-closed loop and the quantity of fresh water collected from a well is 0.83 m^3/t. Water is injected with oxygen to prevent anaerobic fermentation and it is then treated with a physico-chemical process; it enters with a chemical oxygen demand (COD) of 600 mg/l and leaves with a COD of 300 mg/l. Although the treatment is not sufficient to allow discharge to surface waters (160 mg/l being the maximum allowed COD according to Italian laws) it is enough to permit its re-use.

Washing 1 tonne of input material requires 1.4 kWh of electricity for running hydraulic pumps and treating waste water. Moreover, 2.2 kg of aluminium sulphate are used as coagulant, while 0.49 kg of liquid oxygen is used to prevent anaerobic fermentation and speed up the natural treatment by aerobic bacteria. In the washing process, 65 kg of wet sludge per tonne of washed glass are also produced. This sludge is added to the brick grade sand produced by the sorting process and sold to brick manufacturers.

2.4.4. Sorting, drying and milling

According to direct measures, sorting and milling of soda-lime glass require 25 kWh of electricity per tonne of input material. Bulk solid handling consumes 0.5 l of diesel, while drying consumes 103.59 MJ of natural gas. Natural gas drying has been modelled according to Ecoinvent data relevant to the unit "heat, natural gas, at industrial furnace >100 kW".

As previously mentioned, the composition of soda-lime and special glasses can vary over time; consequently, the sorting process must be flexible with respect to possible changes in input material composition, as well as with respect to the variation of market demand for RFG co-products (see Table 3).

Some co-products are internally re-used and some sold to third parties. The composition and market value of outputs are given in Table 3. "Ceramic grade sand" is the commercial name for a granular material made of glass and ceramic fragments/powders with characteristics suitable for ceramic tiles industries. Ceramic grade sand can either be sold, or used in mix with special glass, as the input materials in the foaming process. "Glass grade sand", whose chemical composition is reported in Bernardo et al. (2010) as "glassy sand", is the commercial granular materials sold to the glass industry. "Brick grade sand" is suitable for the production of bricks. A nil price means that the products are delivered free of charge, while R stands for internal reuse.

As previously stated, in the LCA model glass grade sand and ceramic grade sand displace primary production of silica sand, while brick grade sand displaces extraction of brick clay. Plastic scrap is internally used to produce electrical and thermal energy in a waste-to-energy facility. However, since the data related to the SASIL waste-to-energy system are not yet available, and given that post consumer plastic scrap in the study area is often co-incinerated in cement kilns in partial substitution of petcoke (Genon and Brizio, 2008), plastic scrap recovery has been modelled as an

Table 3
Composition and selling price of outputs (R = internal reuse; NA = not applicable).

Products	Output (%)	Destination	Selling price (€/t)
Ceramic grade sand	15	Ceramic industry/ RFG	22
Glass grade sand	80	Glass industry	36
Brick grade sand	3	Brick factories	3
Plastic scrap	1	Waste-to-energy	R
Metal scrap	0.5	Recycling	0
Organic materials	0.5	Water treatment	NA

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avoided production of petcoke. In the LCA model, a low heat value for plastic scrap (30 MJ/kg) and a 10% process loss were utilised. Also in the case of metals, an avoided production of the corresponding virgin materials has been assumed, according to Rigamonti et al. (2010).

Processing of waste special glass requires less electricity (12 kWh/t), as it contains less contaminants. This is likely a consequence of the different route for separate collection, in comparison to waste container glass. However, given the major difficulties related to closed-loop recycling, especially in case of waste CRT (Bernardo et al., 2005; Méar et al., 2005), and considering that no co-products can be recovered from waste special glasses presently treated by SASIL, no avoided impacts can be accounted for in the LCA model.

2.4.5. Thermal process

Glass/ceramic powder from soda-lime and special glass (97%) is mixed with silicon carbide (2%), calcium sulphate (0.5%) and calcium carbonate (0.5%) and is subsequently sent to a furnace in which the raw mix undergoes preheating, foaming ($T = 900$ °C), cooling and subsequent sizing. Although a detailed description of the foaming process is out of the scope of this paper, as it has been extensively described elsewhere (Hurley, 2003; Scarinci et al., 2005), some remarks will help clarify why the industrial partner has selected this specific mix, which is an evolution of a process presented in a previous paper (Bernardo et al., 2010). In fact, other chemical products and industrial minerals such as graphite, coal, MnO_2 and sulphates were also tested, but with less satisfying results or major costs. In rough terms, during the thermal process, silicon carbide (SiC) and calcium carbonate ($CaCO_3$) both generate CO_2 bubbles, which are the foaming agent. While the carbonate provides CO_2 by decomposition (Scarinci et al., 2005), SiC provides CO_2 by oxidation; such oxidation is operated by the oxygen available in the foaming furnace and also by the oxygen provided via the reduction of sulphates ($CaSO_4$) into sulphites and sulfides. The silica released as by-product of SiC oxidation is incorporated by the glass.

The structure of the foam glass traps a large part of the emitted CO_2, the rest being released into the atmosphere. Based on the total carbon in SiC and assuming a 50% entrapment, a direct emission of 11 kg of CO_2 per tonne of RFG has been entered in the LCA model. During the reaction, SO_4 is reduced to SO_3 and remains in the RFG matrix, therefore no SO_x air emissions needed to be accounted for. As far as heavy metals contained in the special glass are concerned, a set of laboratory tests showed that their volatility starts to significantly increase at temperatures above 1200 °C; this is also confirmed in the literature (Bernardo and Albertini, 2006; Bernardo et al., 2005). Consequently, no significant lead and barium air emissions are released from the furnace ($T = 900$ °C).

The required thermal energy of 1800 MJ/t is supplied by an electric furnace. A scenario with a natural gas furnace and a scenario with an electric furnace fuelled with electricity from natural gas co-generation were considered for comparison. Inventory data related to the Italian electricity mix, electricity from co-generation, use of a natural gas furnace and silicon carbide production were retrieved from the Ecoinvent database.

In order to take into account the variability of input materials, three possible mixes of soda-lime and special glass were considered (Table 4). As far as density and insulating properties of the RFG are concerned, these were found to remain within the range given in Table 1.

Table 4
Composition of the raw mix for RFG production.

	Soda-lime glass (%)	Special glass (%)
Mix 1	50	50
Mix 2	80	20
Mix 3	20	80

3. Results and discussion

RFG produced from Mix 1 (50% of soda-lime glass and 50% of special glass) has been chosen as the baseline scenario.

Table 5 displays mid-point indicators and the single score Eco-Indicator 99 relevant to both electric heating (E), natural gas heating (NG) and electric heating from natural gas co-generation (C). The differences in terms of environmental performance are remarkable: −52% in the case of AP and −54% in the case of POCP.

3.1. Contribution analysis

Impacts are due to transportation, processing and firing, while savings come from avoided landfill and recovery of co-products (Fig. 3).

It can be observed that the environmental gains related to the avoided landfill are cancelled by the transport-related impacts. Thus, it is not sufficient to base environmental claims on the statement that RFG is sustainable because it avoids landfilling, as the related gains are lower than the induced impacts.

An important contribution to improve the environmental profile of RFG is represented by recovered plastic, metals and glass fragments/powders, whose environmental gains are higher than those corresponding to landfill avoidance. This suggests that, in order to improve the RFG eco-profile, the raw mix should preferably be made of soda-lime glass rather than special glass, which does not contain recoverable metals and plastics. This finding highlights that industrial symbiosis can play a key role in eco-efficient glass recycling and further supports the recommendation of Hurley (2003) according to which closed-loop container glass recycling remains a preferable option.

Production of SiC and RFG firing represent the highest induced impacts. In spite of the small amount used, SiC is an important contributor to the overall impacts. Consequently, although SiC proved to be an excellent foaming agent (Bernardo et al., 2007), a more environmentally friendly additive is preferable. A possible solution to cutting SiC-related impacts might be replacing primary SiC with waste SiC, which is an option already investigated in the literature (Bernardo et al., 2007; Fernandes et al., 2009). Another source for waste SiC, which is likely to be pursued by SASIL SpA is the use of waste SiC from end-of-life roll-conveyors currently used in ceramic and sanitaryware production. Waste SiC would correspond to zero impacts related to SiC production and limited impacts for transportation and grinding.

3.2. Sensitivity and improvement analysis

The influence of the raw mix composition (Table 4) has been investigated, as reported in Fig. 4.

Table 5
Eco-profile of RFG made from Mix 1 according to different thermal processes.

Indicator		RFG-E	RFG-NG	RFG-C
GER	MJ/t	7761	5118 (−34%)	5405 (−30%)
GWP	kg CO_{2eq}/t	513	349 (−32%)	386 (−25%)
AP	mol H^+/t	77	37 (−51%)	37 (−52%)
EP	g O_{2eq}/t	7907	4338 (−45%)	5583 (−29%)
POCP	g C_2H_{4eq}/t	12.6	8.6 (−31%)	5.8 (−54%)
EI-99	Pt/t	23	15 (−35%)	17 (−25%)

E = electric heating using electricity from the Italian mix; NG = natural gas heating; C = electric heating using electricity from natural gas co-generation.

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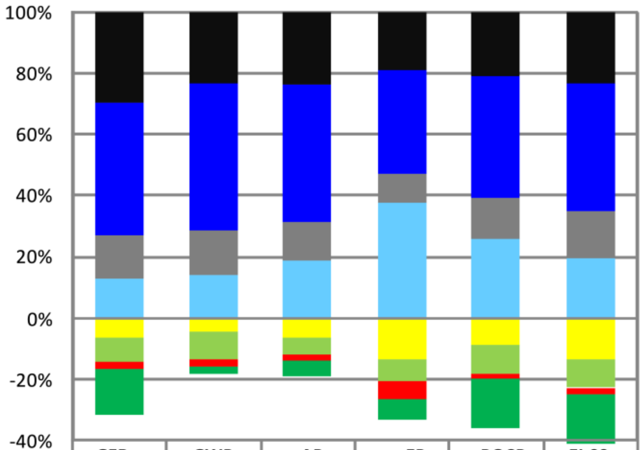

	GER (MJ)	GWP (kg CO_2)	AP (mol H+)	EP (g O_2)	POCP (g C_2H_4)	EI-99 (Pt)
Foaming agent	3382	144.6	22.5	2237	4.1	9.1
RFG firing	4928	304.2	42.8	4078	7.9	16.4
Processing	1595	92.7	12.2	1113	2.6	5.8
Transport	1491	86.1	17.8	4458	5.1	7.8
Avoided landfill	-773	-27.7	-6.4	-1605	-1.7	-5.3
Glass recovery	-876	-55.3	-5.0	-881	-1.9	-3.6
Metal recovery	-213	-18.1	-1.8	-663	-0.3	-0.9
Energy recovery	-1772	-13.1	-5.1	-827	-3.1	-6.2

Fig. 3. Induced and avoided impacts in the RFG waste-to-production chain (Mix 1, electric heating) – (100% = sum of induced impacts).

Mix 2 shows the best performance according to GER, POCP and EI-99. Mix 3 has the lowest impact according to GWP, AP and EP.

Since the electricity from the Italian grid showed a heavy contribution to the environmental impacts of RFG, an improvement scenario could be switching from an electric to a natural gas fuelled kiln. Such a proposed change of technology was regarded as too costly and thus a second improvement scenario was proposed, namely using electricity from a natural gas co-generator instead of drawing electricity from the grid.

The comparison between the eco-profiles of RFG produced using electricity from the grid, RFG produced with a natural gas fuelled kiln, and with an electric kiln powered with a co-generator are shown in Table 5. The solution adopted by the industrial partner, i.e. electric kiln plus natural gas co-generator, shows reasonably good environmental performance at reasonable costs. New energy-saving foaming processes are described in the literature (Hurley, 2003; Scarinci et al., 2005), but are not presently used for commercial purposes.

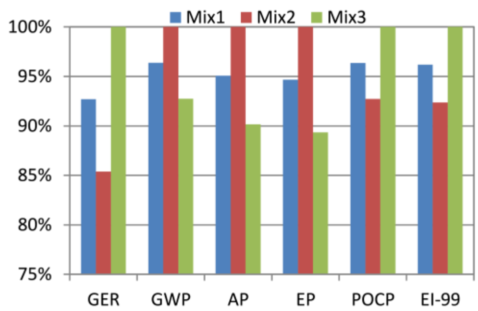

Fig. 4. Comparison among RFGs produced from different raw mixes (electric heating) – (results are normalised to the case with the highest impact).

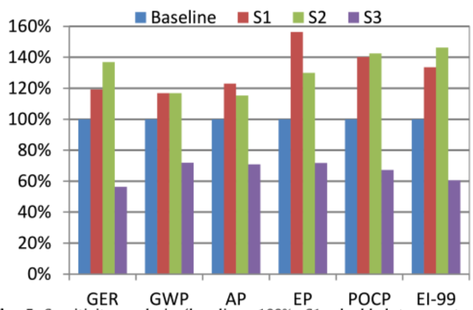

Fig. 5. Sensitivity analysis (baseline = 100%; S1 = doubled transport distances; S2 = exclusion of avoided products; S3 = use of waste SiC).

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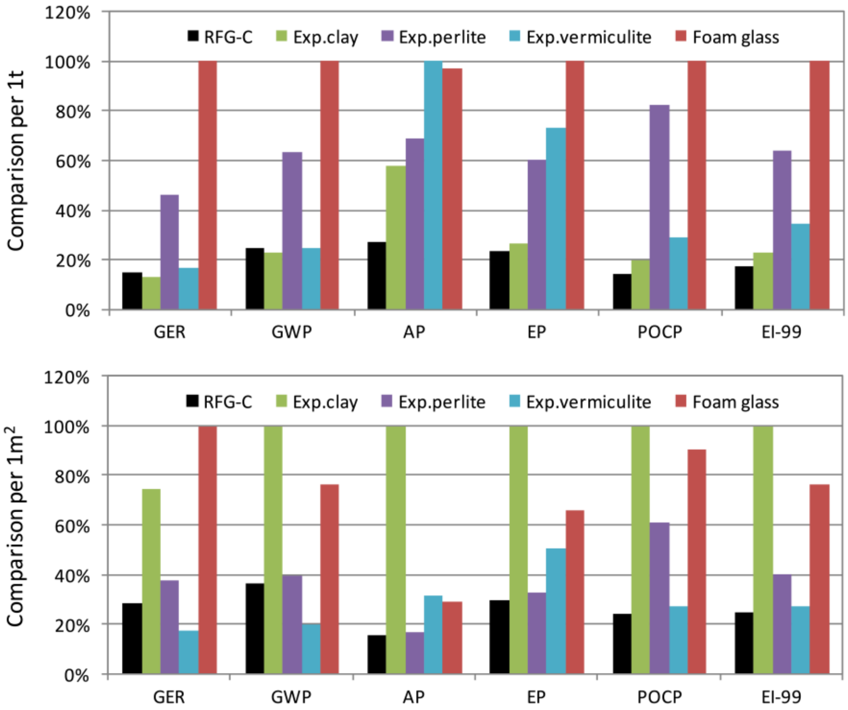

Fig. 6. Comparison among RFG (RFG-C = electric heating from gas cogeneration) and some selected mineral-based insulating materials – (results are normalised to the case with the highest impact).

Given the remarkable influence of transport-related impacts, recovery of co-products, and SiC production, a sensitivity analysis is presented in Fig. 5, where, one at a time, the parameters are changed and compared to the baseline scenario: transport distances are doubled; avoided products are excluded; SiC is substituted by waste SiC.

Bearing in mind that a meaningful comparison among insulating materials should be carried out only once their end-use and the function of the system under study have been identified, Fig. 6 shows energy and environmental indicators relevant to the materials shown in Table 1. The RFG eco-profile is that for electric heating and natural gas co-generation (RFG-C). Inventory data for the remaining insulating materials are taken from the Ecoinvent. The comparison is given per unit of mass (1 tonne) and per unit of area of an insulating board with the same thermal resistance, using average densities and thermal conductivities from Table 1. No further comments are provided because the aim of this paper has been to describe the recycling route under study and stimulate interest in more transparent LCAs of other building products made from recycled materials.

4. Conclusions

LCA led to a deeper understanding of the RFG eco-profile, and improved environmental management of the waste-to-production chain. Moreover, the results provided scientific background supporting the producer's environmental claims and information essential for conducting LCAs of future end-uses of RFG in energy efficient buildings.

The main environmental strengths and weaknesses of the waste-to-production chain can be summarised as follows:

- The environmental gains related to landfill avoidance are offset by increased transport-related impacts.
- The environmental gains related to recovery of co-products are higher than those from avoided landfilling. This emphasises the role of an eco-efficient waste glass recycling chain, in which an innovative multi-output process made it possible to re-process low quality glass rejects and sell most of them back to their original industries (closed-loop recycling), while only a small fraction of post-consumer glass is used in RFG production. This is a synergy obtained from integrated waste management and production in the context of industrial symbiosis and eco-efficient recycling.
- As energy use for the thermal process is a hot spot, LCA results suggested switching to a natural gas powered kiln or an electric kiln powered with a natural gas co-generator, the latter being the solution adopted by the industrial partner.
- A further important improvement could be obtained through the substitution of silicon carbide for a more environmentally friendly additive, or the use of waste silicon carbide from end-of-life roll-conveyors currently used in ceramic and sanitaryware production.
- For a more comprehensive comparison between RFG and other building materials it will be necessary to better define the end-uses. Otherwise, it will not be possible to fully understand the direct and indirect environmental gains that RFG will transfer to the final product (i.e. the building as a whole). Nevertheless, the eco-profile of RFG has been contrasted against those of

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other insulating materials. This should be helpful in order to discuss the relative importance of single subsystems that transfer environmental gains and/or impacts to green products made from waste, for instance: landfill avoidance, change in transportation distance, recovery of co-products and, finally, energy/material saving in highly energy intensive thermal processes.

Acknowledgements

The research was carried out in co-operation with SASIL SpA within the EU Project NOVEDI (www.sasil-life.com/), financed under the call LIFE+ 2007 (contract ENV/IT/00361). The authors would like to thank Dr. Enrico Bernardo (University of Padua, Italy) for the support and for the fruitful and stimulating discussion.

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Please cite this article in press as: Blengini, G.A., et al. Eco-efficient waste glass recycling: Integrated waste management and green product development through LCA. Waste Management (2011), doi:10.1016/j.wasman.2011.10.018

11. ANNEX D

Waste Management 32 (2012) 1712–1721

Contents lists available at SciVerse ScienceDirect

Waste Management

journal homepage: www.elsevier.com/locate/wasman

Participatory approach, acceptability and transparency of waste management LCAs: Case studies of Torino and Cuneo

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ARTICLE INFO

Article history:
Received 23 November 2011
Accepted 13 April 2012
Available online 12 May 2012

Keywords:
Participatory approach
Stakeholders engagement
Waste management
Municipal solid waste
Attributional life cycle assessment
LCA assumptions

ABSTRACT

The paper summarises the main results obtained from two extensive applications of Life Cycle Assessment (LCA) to the integrated municipal solid waste management systems of Torino and Cuneo Districts in northern Italy. Scenarios with substantial differences in terms of amount of waste, percentage of separate collection and options for the disposal of residual waste are used to discuss the credibility and acceptability of the LCA results, which are adversely affected by the large influence of methodological assumptions and the local socio-economic constraints. The use of site-specific data on full scale waste treatment facilities and the adoption of a participatory approach for the definition of the most sensible LCA assumptions are used to assist local public administrators and stakeholders showing them that LCA can be operational to waste management at local scale.

1. Introduction

Life Cycle Assessment (LCA) applied to sustainable municipal solid waste management has rapidly expanded over the last few years as a tool that is able to capture and handle complexities and interdependencies typically characterising modern integrated waste management systems (I-WMS).

A recent, fairly comprehensive and extensive literature review by Pires et al. (2011b) pointed out to what extent a system approach is becoming strategic in order to take into account many technical and non-technical aspects of solid waste management systems. In fact, I-WMSs should be analysed as a whole, since they are inter-related with one another and developments in one area frequently affect practices or activities in another area. The same authors (Pires et al., 2011b) classified nine system assessment tools commonly used in waste management (WM), among which LCA clearly emerged as the most popular, scientifically sound and worldwide appreciated.

The context, objectives and operational conditions that characterise the growing number of recently published LCA applications to WM are quite variable.

With reference to the European countries and the most recent legislation published by the European Commission (EC), i.e. the Waste Directive 2008/98/EC (EU, 2008), in the geographical areas where WM is closer to the EU targets of sustainability, LCA is mostly used to rationalise technological choices and management strategies, while in less advanced regions LCA is used to develop measures to implement more integrated solid waste management and reach EU directives.

As reported in Rigamonti et al. (2010), several LCA studies deal with the I-WMS as a whole (i.e. from a system perspective), while other studies are focused on single subsystems (or groups of subsystems taken individually) devoted to the treatment of single waste fractions.

Although the LCA methodology and the WM related tools are rapidly expanding, there are still uncertainties and open issues, which are challenging the scientific community and that, are limiting the diffusion among end-users. Thus, the question "What life-cycle assessment does and does not do in assessments of waste management" raised by Ekvall et al. (2007) is still partially unanswered and still of great interest.

One of the key issues is understanding what LCA can do for local waste authorities and operators. Moreover, it is still unclear to what extent these subjects are aware of the potential of WM LCAs and/or are willing to put into practice the results.

LCA can supply objective and comprehensive information, but, in Italy and elsewhere, the final decision lies mostly with public

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0956-053X/$ - see front matter © 2012 Elsevier Ltd. All rights reserved.
http://dx.doi.org/10.1016/j.wasman.2012.04.010

administrators seldom aware of the great potential of LCA. Such public administrators often set up priorities and take decisions more on financial constraints rather than on environmental optimisation issues (Blengini, 2008).

Beyond the great advancements of the scientific community, the central question is therefore: "is WM LCA fully operational to business?" In other words: "is LCA accepted, used, understood and put into practice by all the stakeholders?".

LCAs of complex and inter-dependent systems such as WMs necessarily reflect complexity, which is also influenced by non-technical factors, site-specific aspects and local socio-economic constraints. Therefore, LCAs are difficult to be handled and/or developed by non-experts. Moreover, the results of a LCA applied to an I-WMS are unique and should never be generalised, though a lesson can be learned. Consequently, the most important message from a WM LCA should not be the final results, but rather a combination of the results and the way LCA was conducted.

In order to develop meaningful LCAs of I-WMS: (1) goal and scope must be clearly identified, justified and outlined; (2) both input data and inventory results must be fully made available and it should be possible to mathematically manipulate them. Numerical results unsupported by assumptions and the full dataset might be of limited importance.

Two examples taken from the literature might help clarify the context. Rigamonti et al. (2009, 2010) have shown to what extent the selection efficiencies, the adopted technologies and the methodological assumptions related to avoided products might drastically change the overall environmental performance of WM subsystems. A similar picture is presented by Merrild et al. (2008) in case of wastepaper recycling vs. incineration, where the overall energy and environmental indicators can change from positive (net impact) to negative (net gain) depending on the combination of the adopted technologies.

The consequence of the large variability of the environmental performance of subsystems, which heavily depends on assumptions, becomes exponential when dealing with an I-WMS. Moreover, when also socio-economic constraints are taken into account, including, for instance, the preferences of stakeholders relevant to different areas of environmental concern, it is very possible that LCA results become subjective to a large extent, with consequent increased scepticism and loss of credibility and acceptance. This is a very important area of concern that represents an obstacle to the diffusion of WM LCAs, and is also the central point of the present article, where two extensive LCAs run by the Politecnico di Torino in the years 2008 and 2009 (Blengini et al., 2008, 2009) are used in order to discuss on strategies to boost adoption of LCA in WM in northern Italy, and elsewhere, and increase the credibility and acceptability of results.

The original contribution of the present paper can be summarised as follows:

- Use of site-specific data on full scale waste treatment facilities in the study area in order to cover all the WM activities in the I-WMS and the full life cycle of waste;
- use of the participatory approach in order to address the most sensible LCA assumptions and propose solutions in order to enhance the acceptability of results;
- assist the local public administrators in order to verify and quantify the effectiveness of EU strategies on WM using site-specific data and taking into account the local socio-economic constraints, emphasising that LCA application is both useful and feasible.

2. Model and data development

The paper presents a synthesis and the main results of two research programmes focused on the application of LCA to a set of WM scenarios in Torino and Cuneo Districts in northern Italy (Blengini et al., 2008, 2009). The study area covers a population of nearly 2800,000 inhabitants with an annual generation of nearly 1500,000 tons of municipal solid waste (Fig. 1). In both cases, the overall objective was identifying scenarios with best energy and environmental performance. A detailed energy and environmental analysis was carried out for the main components of the I-WMS and for the I-WMS as a whole in order to support public administrators towards sustainable waste management.

The above research programmes were developed by the Politecnico di Torino and funded by the WM Authorities of Torino and Cuneo Districts. LCAs were implemented using the SimaPro 7 software (SimaPro7, 2006).

All the subsystems included in the I-WMS were considered and analysed paying attention to energy and environmental implications and inter-dependencies. Separate collection (SC) and its downstream recycling chains were investigated in terms of environmental benefits and impacts, in order to quantify advantages and drawbacks that can be ascribed to the new objectives of SC (65% by the year 2012) introduced by the law presently in force in Italy (Dlgs.152/06). At the same time, the role and environmental implications of energy recovery from residual waste were analysed, paying attention to the consequences of possible pre-treatment options of the residual waste, and considering both incineration and co-incineration.

2.1. Definition of goal and scope through a participatory approach

The LCA methodology according to ISO 14040 (2006) is worldwide accepted and appreciated because it allows an objective evaluation of the environmental performances of products and processes (Guinée, 2002).

However when applying LCA to WM, there are some sector-specific aspects that must be considered and assumptions to be undertaken that might affect the results to a large extent (Ekvall et al., 2007; Finnveden, 1999; Merrild et al., 2008; Rigamonti et al., 2010).

In order to keep under control the negative influence that assumptions might have in terms of acceptability of the results, a participatory approach was adopted since an early stage of the research. When applied from the very beginning, a participatory process may be of help in reducing possible conflicts among opposite interest groups, which is typical in waste management, and contribute towards defining acceptable solutions for all involved parties (Salhofer et al., 2007b). As it was observed in other case studies, where a structured participative approach was applied to waste management, different stakeholders have different objectives (Pires et al., 2011a) and some of them might try to influence the results by changing the criteria in a late stage (Salhofer et al., 2007b). Setting up clear and shared rules and preferences since the beginning is therefore a key issue (De Marchi et al., 2000).

A panel of stakeholders and experts, including participants from Politecnico di Torino, WM authorities of Torino and Cuneo Districts and Environmentalist NGOs, was set up. The trans-disciplinary nature of the panel was similar to those presented in De Marchi et al. (2000) and in Salhofer et al. (2007b), where attention was paid to include all local actors and give them equal opportunities to express their opinion.

An initial brainstorming and subsequent structured meetings were used in order to reach a shared definition of the following aspects that, as the participants revealed, can highly increase the acceptability of the LCA results:

- Identification and description of the scenarios to be compared: amount of waste, composition, percentage of SC, definition of technologies/strategies not yet defined in the local WM policies/plans;

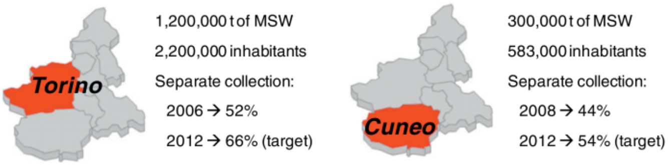


Fig. 1. Municipal solid waste production and population in the Torino and Cuneo districts.

- definition of the case-specific LCA methodological assumptions: system boundaries, avoided/substituted production, cut-off criteria;
- data collection, sources and responsibilities: mass balances, energy consumption and emissions for all the WM subsystems.
- selection of meaningful energy and environmental indicators;

while the detailed description of input data, assumptions and subsystems are reported as Supplementary content, and are fully available in the research reports quoted in the references (Blengini et al., 2008, 2009), in the following paragraphs the main aspects relevant to the discussion within the panel of stakeholders are summarised.

The following sources of data were also used and quoted in the Supplementary content: (AEA, 2001; ANPA, 2000; Blengini et al., 2007; Bovea and Powell, 2006; Cernuschi et al., 2003; Ecoinvent, 2007; Favoino and Hogg, 2008; Giugliano, 2007; Rigamonti et al., 2009).

2.2. Identification and description of scenarios to be compared

With reference to the Torino District, four scenarios relevant to the whole I-WMS were identified: 1A, 1B, 2A, 2B. Scenarios 1 (A/B) depict the status of SC in the year 2006, which was 52.1%, whereas scenarios 2 (A/B) represents the target of SC for the year 2012, which was set at 65.7%.

It must be said that, at the time of the study, the construction of a new incinerator had already been launched, but some important aspects related to the residual waste management chain were still undefined. Thus, inventory data to model residual waste management had to be retrieved from design documents provided by the main contractor, or assumed via consultation with the panel of stakeholders. Therefore, scenarios 1 (A/B) depict a past situation where management of separately collected waste was modelled on real data, while residual waste management is modelled as if incineration was already operational.

The discussion among the panel of stakeholders about adoption or exclusion of a mechanical–biological treatment (MBT) of the residual waste prior to incineration was particularly controversial. An agreement was reached that two out of four scenarios include MBT (1B/2B) and the remaining exclude MBT (1A/2A). However, the panel was not able to reach a complete agreement and left the description of the MBT characteristics and operating conditions vague to some extent.

The destination of biowaste out of MBT was felt of particular importance. It was agreed that the baseline scenarios would consider landfilling of biowaste out of MBT, while possible improvement scenarios were left for a sensitivity analysis that will be presented in Section 5, and for future more detailed investigations. A graphical description of the four scenarios under analysis is available in the Supplementary content.

As Cuneo District is concerned, it must be remarked that the size, socio-economic constraints and organisation of the WM facilities and infrastructures are quite different in comparison to Torino District. In that case, a scenario analysis was carried out to better understand the environmental and energy implications of the I-WMS re-organization in progress at the time of the study (2009).

However, to the aim of the present paper, none of the results relevant to the I-WMS as a whole are provided, but rather a comparison among four residual waste management alternatives (chains) is carried out:

- chain 1: residual waste to MBT, dry waste to residue derived fuel (RDF) production and co-incineration in an existing cement kiln;
- chain 2: residual waste to bio-drying, dry waste to RDF production and co-incineration in an existing cement kiln;
- chain 3: residual waste to a dedicated incinerator;
- chain 4: residual waste to MBT and dry waste to a dedicated incinerator.

Chains 1–4 encompass the whole sequence of activities in the life cycle of 1 ton of residual waste, from collection to the final disposal of residues, including substitution of primary energy and/or recovery of secondary materials. Data on quantities of waste and composition are presented in Table 1. Detailed data are supplied as Supplementary content.

2.3. Definition of the case-specific LCA methodological assumptions

It is well known that identifying and clearly describing system boundaries is a very important step that can heavily influence LCA results. Although the expansion of the principal system boundaries in order to avoid allocation is warmly recommended since long (Finnveden, 1999; ISO 14040, 2006), it must be noticed that case-specific choices related to expanded system boundaries are still controversial (Ekvall et al., 2007).

System expansion avoids allocation, but introduces subjective choices relevant to the substituted primary production, for instance the electricity produced from primary sources that is displaced by energy recovery from waste. Moreover, this introduces a crediting system (negative figures) that is sometimes source of confusion and/or misinterpretation among non-LCA experts. In such a context, an important step is the choice between attributional or consequential LCA modelling (EC et al., 2010). In rough terms, attributional modelling means taking a picture of the present operating conditions, while consequential modelling implies that the LCA model is made representative of an evolution of the present situation towards a given target.

This said, although consequential modelling is certainly more relevant for long-term decision making, this introduces value choices that were again source of conflict among the panel. Some citizen's groups feared that the bargaining power of public authorities could drive the choice of primary energy to be substituted and therefore distort the results, thus increasing scepticism. On the other hand, everybody was interested to know the environmental performance of the full-scale local WM activities and inter-dependencies. It was therefore decided to adopt an attributional principle and create a LCA model that (where possible) represents the actual fate of waste in the study area and the actual displacement

Table 1
Total waste, waste composition and separate collection in Torino and Cuneo Districts.

Waste type	Torino District		Cuneo District	
	2006; SC = 52.1%; ton × 1000	2012; SC = 65.6%; ton × 1000	2008; SC = 43.7%; ton	2012; SC = 54.4%; ton
Organic	190	219.7	6247	13,299
Green	51.7	51.8	16,866	19,319
Plastic	35.1	47.4	10,806	17,264
Paper	193.4	260	51,745	61,930
Wood	54	52.3	6526	7399
Glass	67.5	74.2	23,566	27,635
Metals	20.2	24.7	6200	7029
Other	45	63.2	12,178	16,585
Residual waste	603	414	172,605	142,820
Total	1260	1203	306,738	313,279

of primary energy/materials. Moreover, the panel of stakeholders recommended to include in the LCA model also those waste flows (and the related waste treatment) that go outside the administrative territory, according to actual data.

2.4. Criteria for missing inventory data

An important topic that was discussed among the panel of stakeholders at the initial stage was that relevant to the influence of transport-related impacts. Public administrators didn't have comprehensive and quantitative data on collection systems (from the place where waste is generated to the collection centres or transfer stations), but tended to put much emphasis on this aspect, saying that SC increases transportation distances and consequently impacts might outweigh savings from SC and downstream recycling. On the opposite side, citizen's groups, which are often more ecologically oriented than public administrators (Salhofer et al., 2007b), pushed more towards recycling. According to them, although transport distances might increase to some extent, SC is carried out with smaller and more efficient vehicles, which should compensate impacts.

Given the absence of comprehensive and reliable data on collection distances, the two opposite parties agreed to assume a transportation distance of 50 km for all types of collected waste. According to the field experience of public administrators, such a distance would be an overestimate of the real conditions. All the participants where interested to obtain a proxy quantitative estimate of transport-related impacts and contrast it against the environmental gains of recycling. However, they didn't want such a rough estimate influence the environmental comparison among WM scenarios.

For the sake of clarity, it must be said that transportation distances from transfer stations to waste treatment facilities were all available, and therefore were included in the LCA model (see Supplementary content).

2.5. Selection of meaningful energy and environmental indicators

As the selection of impact indicators is concerned, these were chosen through a consultation with the panel of stakeholders.

As suggested by Kruse et al. (2009) a combined top-down and bottom-up approach was adopted in order to develop a meaningful suite of indicators. A top-down approach can roughly be described as one that selects indicators that are representative of broadly recognised areas of environmental concern, as well as based on various international conventions, agreements, and guidelines. This approach is consistent with the International Standards Organization's (ISO) recommendations for LCIA methods (ISO 14040, 2006). In contrast, a bottom-up approach can be defined as one that identifies indicators based on industry, public administrators or stakeholder interests and/or data availability (Kruse et al., 2009).

This said, a first set of two impact categories was initially proposed according to the above mentioned top-down approach: emission of greenhouse gases and use of non-renewable energy.

Based on the bottom-up approach, the panel confirmed that energy and climate change are meaningful impact categories and expressed an interest to include some indicators relevant to human health. However, after a discussion, it clearly emerged that a careful modelling of human health impacts would have been much more complicated and not compatible with the time frame and data availability. There are in fact some important methodological aspects to be taken into account when incorporating local-scale environmental impacts in LCA (Kruse et al., 2009).

A comprehensive discussion about local-scale environmental impacts in LCA is beyond the scope of this article, however some remarks can help the reader. LCA is in fact mostly relying on additive indicators, i.e. that can be measured quantitatively and that are additive through the chain. Such indicators are calculated through characterization factors that can capture regional and global burdens, but which mostly neglect local and site specific aspects. This is obviously a problem that deserves to be accurately addressed in WM LCAs when human health is concerned. It is in fact very possible that some emissions from the foreground system and the background system are treated as additive, while they are not.

Selected indicators are therefore: GER (Gross Energy Requirement) expressing the total primary energy resource consumption (Boustead and Hancock, 1979); NER (Non-renewable Energy Resources) as the non-renewable part of GER; GWP_{total} (Global Warming Potential – 100 years) as an indicator of the greenhouse effect, including biogenic carbon dioxide (IPCC, 2006); GWP_{fossil} as part of GWP_{total} with exclusion of biogenic carbon dioxide.

Given the data reported as Supplementary content, these allow calculation of environmental indicators typically included in WM LCAs, including human toxicity. However, for the above consideration, although Eco-Indicator 99 (Goedkoop and Spriensma, 1999) was used in order to provide a rough estimate to the panel, human health and ecosystem quality issues were left outside the final panel discussion and are therefore not reported in this paper.

2.6. Methodology for accounting of biogenic carbon dioxide emissions

Accounting and reporting of biogenic carbon emissions and/or sequestration was another important topic under panel discussion. It was agreed that all biogenic carbon emissions were to be accounted and included in GWP_{total}, whereas GWP_{fossil} excludes biogenic carbon dioxide.

Since the greenhouse effect is determined either by fossil and biogenic carbon dioxide (Blengini, 2008; Hogg et al., 2008), in LCAs applied to WM the assumption excluding the biogenic carbon dioxide is not scientifically correct. In fact, the simplification whereby the biogenic carbon dioxide cycle is neutral (the amount of CO_2 absorbed during the plant growth is the same released in its end of life) is an oversimplification when comparing disposal scenarios with different potentials of biogenic carbon dioxide generation.

Here it must be said that there are different accounting methodologies to handle carbon uptake, carbon sequestration in landfill, sequestration associated to the use of compost in agriculture and biogenic emissions. It is therefore necessary to consistently handle C-uptake and emissions throughout the whole life cycle (Christensen et al., 2009; Rabl et al., 2007).

According to Christensen et al. (2009), a simple and transparent model for the calculation of C-balances is recommended. Beyond that, in the authors of this paper opinion, it is also important to report in a transparent way, i.e. separating the biogenic carbon dioxide contribution.

As the present study is concerned, the following accounting rules for biogenic carbon were adopted:

- Generated waste holds no carbon credits (and no environmental burdens) according to the so-called "zero burden assumption" (Ekvall et al., 2007). This assumption can also be supported by the statement of Vergara et al. (2011): "if waste carries with it no environmental burdens, then it should not carry with it any environmental benefits either".
- All biogenic carbon dioxide emissions associated with WM are accounted for (GWP + 1).
- Biomass recycling corresponds to permanent locking of carbon according to the mass flow that is permanently re-circulating in the loop. In practical terms, recycling corresponds to locking of carbon dioxide absorbed during the growth of the biomass. This is automatically obtained in the present LCA according to the following example: if the biomass is incinerated, a CO_2 emission is recorded (GWP + 1), while recycling has no direct emissions (GWP 0).
- Biomass landfill and use of compost in agriculture is assigned a carbon sequestration potentials (GWP − 1).

Detailed data and assumptions are reported in Supplementary content.

3. Results

The following results reflect field data from the study area (Piedmont, Italy) and reflect the full-scale performance of existing (or under construction) WM plants. All the activities in the I-WMS were considered and analysed with focus on their energy and environmental implications and inter-dependencies.

SC and its downstream recycling/treatment are investigated first, paying attention to the net environmental gains, i.e. the eco-balance between recovered materials/energy and the environmental impacts of WM activities. The analysis turns then on the I-WMS as a whole, in order to better understand the role (and weight) of WM subsystems in a more holistic perspective. The environmental implications of energy recovery from residual waste are then presented, with focus on the consequences of possible pre-treatment options of the residual waste, and considering both incineration and co-incineration.

3.1. Energy and carbon balance of recycling/treatment of single materials from SC

With reference to 1 ton of separately collected waste, Fig. 2 shows the energy and carbon balances of SC and subsequent recycling/treatment in a life cycle perspective. The sequence of activities starts after collection (not included) and encompasses transportation, selection, recycling/treatment and substitution (avoided products/energy). Both the main waste flows and residues were included in the analysis, whereas residues are either landfilled of sent to energy recovery. Negative indicators mean that environmental gains are higher than induced impacts. Biowaste refers to a mix of composting and anaerobic digestion (AD), which reflects the current situation in Torino (33% AD and 67% composting), while metals refer to a mix of ferrous and non-ferrous metals (detailed data are reported as Supplementary content).

With reference to Fig. 2, GER savings were found to be substantially similar to non-renewable energy savings, except paper (NER = −13770 MJ/t) and wood (NER = −3559 MJ/t) where most of the GER is renewable energy from biomass. As carbon emissions are concerned, GWP_{total} and GWP_{fossil} were found to be substantially the same, except for biowaste where GWP_{fossil} shows a net saving of −163 $kgCO_{2eq}$/t.

3.2. Analysis of whole I-WMS: comparison of the four scenarios in Torino District

With reference to 1 ton of total waste, Table 2 shows the energy and carbon balances related to the four scenarios. According to both energy and climate change indicators, scenarios with 65.6% of separated collection appear to be more eco-efficient than those with 52.1%. Scenarios which include MBT (1B and 2B) show a more favourable carbon balance, but perform worse in terms of energy balance.

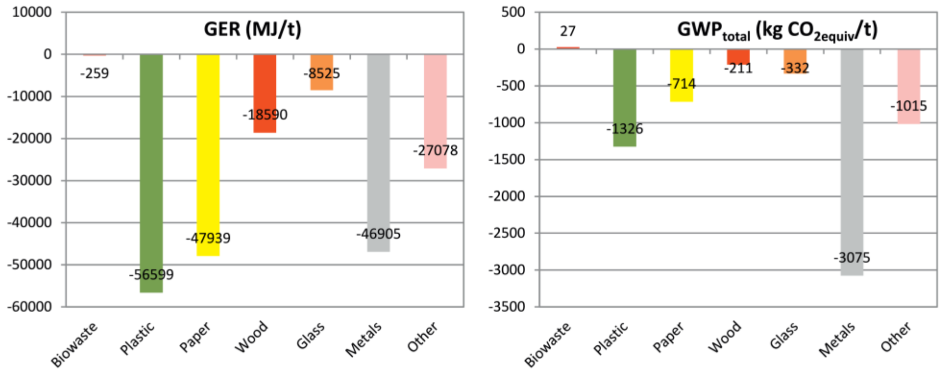
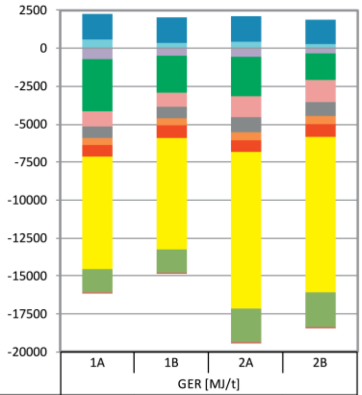


Fig. 2. Energy and carbon balance of separately collected waste materials.

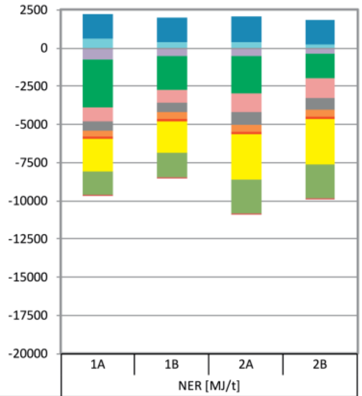
Table 2
Energy and carbon balance of the four scenarios under comparison (Torino District).

Impact category	Unit	Scenario 1A	Scenario 1B	Scenario 2A	Scenario 2B
GER	MJ/t	−13,898	−12,858	−17,362	−16,497
NER	MJ/t	−7476	−6499	−8811	−8001
$GWP_{100total}$	kg CO_2eq/t	233	142	26	−46
$GWP_{100fossil}$	kg CO_2eq/t	−156	−160	−230	−241



	1A	1B	2A	2B
Collection	1633	1633	1633	1633
Biowaste	-50	-50	-58	-58
Plastic	-1575	-1575	-2219	-2219
Paper	-7356	-7356	-10299	-10299
Wood	-797	-797	-804	-804
Glass	-456	-456	-523	-523
Metals	-753	-753	-959	-959
Others	-967	-967	-1411	-1411
Residual Waste	607	368	436	254
Electricity (WtE)	-3458	-2400	-2606	-1743
Heat (WtE)	-726	-504	-550	-367

Fig. 3. Contribution of subsystems to the energy balance of the I-WMS of Torino District (GER).



	1A	1B	2A	2B
Collection	1613	1613	1613	1613
Biowaste	-50	-50	-59	-59
Plastic	-1561	-1561	-2199	-2199
Paper	-2113	-2113	-2958	-2958
Wood	-152	-152	-154	-154
Glass	-396	-396	-454	-454
Metals	-642	-642	-818	-818
Others	-869	-869	-1269	-1269
Residual Waste	593	378	426	261
Electricity (WtE)	-3174	-2203	-2392	-1600
Heat (WtE)	-724	-502	-549	-365

Fig. 4. Contribution of subsystems to the energy balance of the I-WMS of Torino District (NER).

A contribution analysis was conducted in order to quantify the relative importance of various subsystems over the environmental performance of the I-WMS as a whole (per 1 ton of total waste). Figs. 3–6 report the contributions of collection and WM activities related to single waste fractions. The contribution of separately collected wastes encompass the same activities already described in Section 3.1. Such a contribution depends on the overall recycling efficiency (Fig. 2) and the quantities shown in Table 1.

In the case of residual waste, recovered electricity and heat (WtE) are reported separately in the lower part of the tables embedded in Figs. 3–6, whereas "residual waste" encompass the impacts from pre-treatment (scenarios 1B/2B), incineration and landfill of residues. In order to provide more quantitative information on the impacts related to residual waste, in scenario 2B the contribution of pre-treatment to the GWP_{total} is 16 kg CO_{2eq}/t, that of incineration is 300 kg CO_{2eq}/t and that of stabilised organic fraction landfill is 23 kg CO_{2eq}/t.

3.3. Comparison among alternatives for energy recovery from residual waste: Torino and Cuneo districts

A comparison among alternative chains for energy recovery from residual waste was felt of strategic interest by the panel of stakeholders. The results are presented in Table 3 and are based on both Torino and Cuneo Districts LCAs (detailed data are reported as Supplementary content).

With reference to 1 ton of residual waste, Chains 1 and 2 are based on two alternative processes for the production of RDF and subsequent co-incineration in an existing cement kiln located in the surroundings of Cuneo town, which currently produces 1.6 Mt of clinker per year and hold a co-incineration capacity of 100 kt.

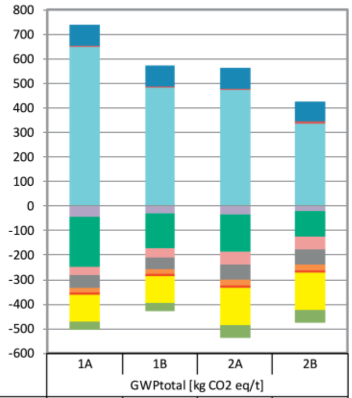
Chains 3 and 4 reflect the operating conditions of the above described incinerator under construction in the town of Torino. In particular, Chain 3 is based on inventory data retrieved from the scenario 1A of Torino District LCA (without MBT), while Chain 4 is based on inventory data retrieved from the scenario 1B of Torino District LCA (with MBT).

Table 3 shows that Chain 1 is the most efficient in terms of energy recovery and greenhouse emissions.

4. Discussion

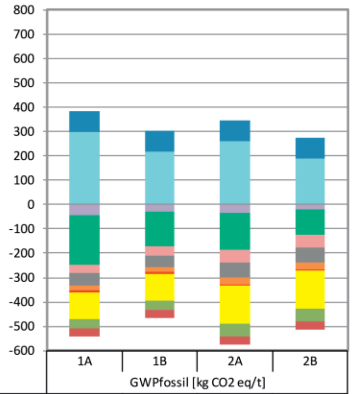
The results from the LCA were discussed with the panel of stakeholders.

A first important discussion was that relevant to the ecological relevance of transport. Public administrators in the panel were unaware of the quantitative impacts of collection and transporta-



	1A	1B	2A	2B
	GWPtotal [kg CO2 eq/t]			
Collection	84	84	84	84
Biowaste	5	5	6	6
Plastic	-37	-37	-52	-52
Paper	-110	-110	-153	-153
Wood	-9	-9	-9	-9
Glass	-18	-18	-20	-20
Metals	-49	-49	-63	-63
Others	-36	-36	-53	-53
Residual Waste	649	483	472	339
Electricity (WtE)	-204	-142	-154	-103
Heat (WtE)	-42	-29	-32	-21

Fig. 5. Contribution of subsystems to the carbon balance of the I-WMS of Torino District (GWP_{total}).



	1A	1B	2A	2B
	GWPfossil [kg CO2 eq/t]			
Collection	84	84	84	84
Biowaste	-31	-31	-37	-37
Plastic	-37	-37	-52	-52
Paper	-110	-110	-153	-153
Wood	-9	-9	-9	-9
Glass	-21	-21	-24	-24
Metals	-49	-49	-63	-63
Others	-36	-36	-53	-53
Residual Waste	300	220	262	190
Electricity (WtE)	-205	-142	-154	-103
Heat (WtE)	-42	-29	-32	-21

Fig. 6. Contribution of subsystems to the carbon balance of the I-WMS of Torino District (GWP_{fossil}).

tion and tended to overestimate them, or believed that impacts of collection and transportation might outweigh savings from SC and downstream recycling. The numerical results showed that the impacts of collection and transportation are much lower than environmental gains from recycling, which is in accordance with other studies (Merrild et al., 2012; Salhofer et al., 2007a). Moreover, it was pointed out that only part of the impact of collection can be ascribed to transportation (60% of energy and 75% of GHG), the remaining impact being associated with the manufacturing of waste bags and containers. Public administrators committed themselves to made data on collection available in future LCAs, but, at the same time, they agreed that having assigned the same transportation distance to all waste types in all scenarios would not substantially change the conclusions of the study.

A very recent paper by Merrild et al. (2012) confirmed that SC corresponds to a higher diesel use (4.1 l/t) than collection of residual waste (3.6 l/t). However, such a difference is relatively low. This substantially supports the position of environmentalist NGOs, according to which, although transport distances remarkably increase, SC is carried out with smaller and more efficient vehicles, which partially compensate impacts.

Another important discussion was that on ecological efficiency of recycling. The numerical results of Fig. 2 and the quantitative description of the recycling chains (reported as Supplementary content) where extremely helpful to show to the panel how recycling is modelled in LCA. Many were unfamiliar to basic concepts such as selection and recycling efficiencies and partially unaware that recycling avoid not only manufacturing, but also its upstream activities. Thus, while incineration can recovery part of the feedstock energy, recycling can recovery part of feedstock energy, but also direct and indirect energy (Boustead and Hancock, 1979). Such a discussion helped the panel to correctly interpret the LCA results and highly contributed to enhance their acceptability.

Table 3
Comparison among four alternative residual WM chains (Torino and Cuneo Districts).

Impact category	Unit	Chain 1	Chain 2	Chain 3	Chain 4
GER	MJ/t	−10,117	−8065	−6354	−3187
NER	MJ/t	−9933	−8080	−5805	−2902
$GWP_{100total}$	kg CO_2eq/t	90	149	906	686
$GWP_{100fossil}$	kg CO_2eq/t	−327	−229	178	180

At the end of the process, in fact, all participants expressed their satisfaction on how the LCA was conducted. Similarly to other cases reported in literature (Salhofer et al., 2007b), all the participants took the opportunity to express their opinion and support the LCA with their contribution, but none could influence the results to a large extent.

Bearing in mind that data were collected from plants and activities well representative of the study area, at the end of the discussion on SC and recycling, the panel agreed that the net environmental gains obtained in this study are not overestimated. On the contrary, the research highlighted that there is room for improving the eco-efficiency of the collection-recycling chain, which is not fully optimised (in the case of plastic, only 49.4% is effectively recycled, see Supplementary content). An ex post confirmation on the actual recycling efficiencies came from another LCA study in northern Italy (Rigamonti et al., 2009), where similar numerical results are reported.

4.1. Sensitivity analysis of Torino District LCA

A sensitivity analysis was used to address some of the most controversial issues raised by the panel of stakeholders. With reference to the pre-treatment of residual waste prior to incineration, the debate concentrated on the mass balance at the MBT plant,

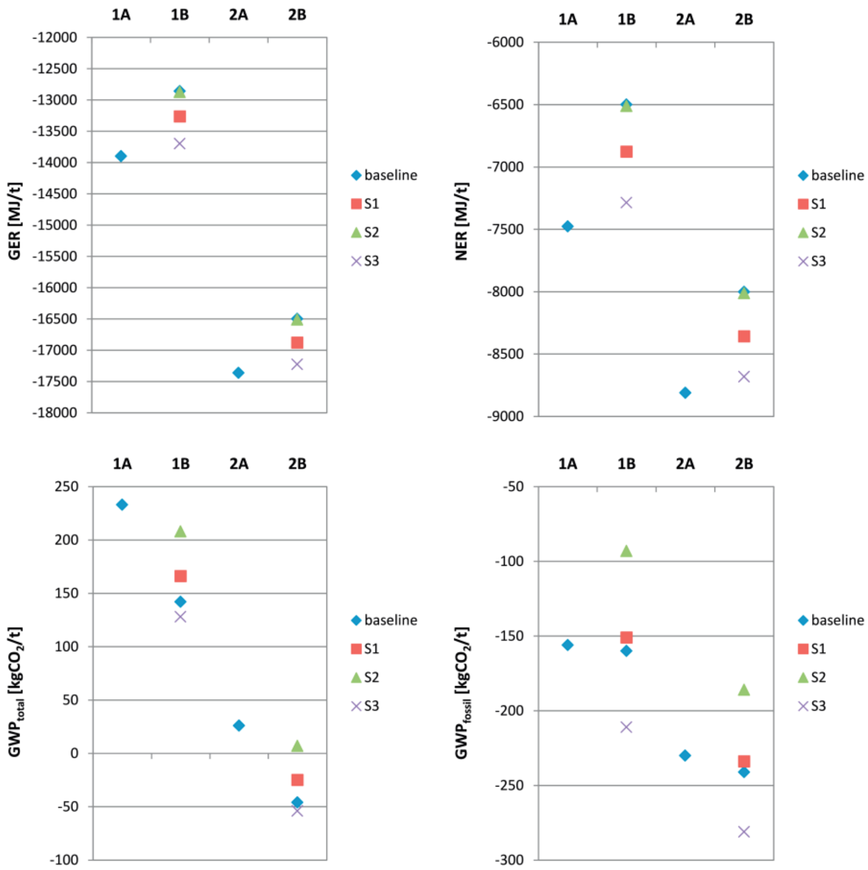


Fig. 7. Sensitivity analysis of Torino District LCA.

i.e. quantity and quality of light/dry waste and stabilised organic fraction, and the final destination of biowaste out of MBT.

Bearing that in mind, the baseline scenarios, i.e. the four scenarios (1A/B, 2A/B) summarised in Section 2.2 and described in the Supplementary content, were contrasted against the following sensitivity scenarios, where some of the key assumptions were changed.

- Sensitivity scenario S1: a different set of removal coefficients at the separation stage is considered (see Supplementary content). Consequently, mass balance and heat values of the light/dry fraction are recalculated, as well as carbon dioxide emissions;
- Sensitivity scenario S2: this is a pessimistic scenario where aerobic stabilisation of biowaste out of MBT is assumed to have a 50% reduced effectiveness and where biogas collection from landfill has a reduced efficiency (from 55% to 27.5%);
- Sensitivity scenario S3: this is an optimistic scenario were the organic fraction out of MBT is sent to AD, sludge from AD is sent to composting prior to landfill. Given the low quality of compost, no fertilizers substitution is accounted for.

The results of the sensitivity analysis (Fig. 7) helped understanding and interpreting the previously presented Tables and Figures. Note that in Fig. 7 scenarios that exclude MBT (1A/2A) remain unchanged.

With reference to the baseline scenarios, energy indicators highlighted that scenarios that includes pre-treatment plus incineration appear to be slightly less efficient (−5%) than scenarios with direct incineration. However, this can be ascribed to landfill without energy recovery of the biowaste fraction out of MBT. A possible improvement could be AD of biowaste out of MBT, which would bring scenarios A and B to a similar energy saving performance. In case of AD, also logistic, technical and economic aspects should be considered, which go outside the framework of the present study.

The carbon balance has emphasised that the pre-treatment of the residual waste sensibly improves the climate change impacts in comparison to incineration without pre-treatment. An important aspect is the dynamic of the carbon cycle of landfilled biowaste fraction after MBT and the actual efficiency of biogas collection. As GWP is concerned, the pessimistic scenario in the sensitivity analysis showed that scenarios A and B are nearly equivalent (scenarios B can be worse than A in case of GWP$_{fossil}$).

The sensitivity analysis highlighted therefore that the vagueness in the definition of the pre-treatment technologies and subsequent destination of the organic fraction are crucial factors that deserve further and more accurate investigation.

5. Conclusions

Detailed applications of LCA to integrated waste management systems are complex and the subsequent analysis necessarily reflects this complexity. Developing waste management strategies is a challenging task which encompasses several aspects that cannot be fully included in a LCA analysis. Moreover, the research programmes carried out for Torino and Cuneo Districts have once more confirmed that there are not preferable waste management solutions in terms of all the environmental and energy indicators.

The two LCAs summarised in this article confirmed that SC and downstream recycling is the most effective tool to improve energy efficiency and to lower environmental impacts. This conclusion was drawn after considering the whole sequence of activities, thus quantifying the eco-balance of collection, transportation, selection, recycling of the main waste flows and landfill/energy recovery from residues. This important site-specific conclusion confirmed that, under the local operational conditions, SC objectives according to the Italian national law in force (Dlgs.152/06) are consistent with an overall energy and environmental efficiency target. This was an important feedback for the public administrator involved in the research.

While the priority should be given to SC and subsequent recycling, energy recovery from residual waste also plays an important role. As residual waste is concerned, it must be remarked that Torino and Cuneo Districts have substantial differences in terms of existing/under construction WM infrastructures, which, at the time of the research, was a constraint limiting the possible WM scenarios to be compared. In fact, Cuneo District can take advantage of an existing cement factory, which is already authorised to co-incinerate RDF, whereas Torino District was going to build a new incinerator. In both cases, it clearly emerged that the eco-efficiency of energy recovery is much lower than the eco-efficiency of recycling. Based on the mass balances of the whole chain, and bearing in mind the site-specific data and local operational conditions, it appeared that co-incineration corresponds to better energy and carbon performances than dedicated incineration (with or without pre-treatment of residual waste). As far as energy and climate change issues are concerned, and according to the LCA results, an existing co-incineration plant should be preferred to a new incinerator. However, the research has also highlighted that the efficiency of the production of RDF plays an important role.

The research confirmed once more that the results of a LCA applied to an I-WMS are heavily influenced by site-specific aspects and local socio-economic constraints, and, therefore, should never be generalised. Consequently, the most important message from a WM LCA should not be the final results, but rather a combination of the results and the way LCA was conducted.

It was observed that stakeholders were extremely interested in actively contributing to the LCA, but under the condition to discuss the assumptions in details and agree upon the sources of data. Such a shared process highly contributed to the credibility and acceptability of the results. However, it was also observed that the vagueness in the definition of key elements in the I-WMS (in this case the pre-treatment of residual waste) can be an obstacle to the implementation of the LCA results.

Beyond site-specific conclusions, a general, more important, conclusion is that, without a deeper engagement of public administrators and stakeholders in the definition of case-specific methodological assumptions, LCA applied to WM will not easily become fully accepted and operational.

Acknowledgements

The Authors wish to thank all the public administrator of Torino and Cuneo Districts and all the members of the panel of stakeholders that actively participated in the research programmes.

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.wasman.2012.04.010.

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