FINAL REPORT

LCI SUMMARY FOR PLA AND PET 12-OUNCE WATER BOTTLES

Prepared for

PET RESIN ASSOCIATION

by

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INTRODUCTION

This summary evaluates the life cycle inventory results for 10,000 12-ounce water bottles. The common polyethylene terephthalate (PET) 12-ounce water bottle is compared to one produced from polylactide (PLA). A virgin PET water bottle has been analyzed for comparison, as well as a PET water bottle scenario including the gross PET bottle recycling rate (23.5 percent) for the PET water bottle.[1] The same methodology utilized in the report, Life Cycle Inventory of Five Products Produced from PLA and Petroleum-Based Resins, is also used in this analysis.[2] The goal, scope, and boundaries of this summary are consistent with those of the LCI cited as well.

The weights of the water bottles are shown in Table 1. In order to express the results on an equivalent basis, a functional unit of equivalent consumer use (10,000 bottles) was chosen for this analysis. Figures 1 and 2 display flow diagrams of the production of the two resins analyzed in this analysis. Figure 3 shows the three systems analyzed in this analysis from the transportation of the resin to fabrication through their end-of-life. The common end-of-life scenario for municipal solid waste (MSW) in the United States estimates that 20 percent of mixed MSW is combusted in a waste-to-energy facility, whereas the remaining 80 percent of the mixed MSW is landfilled. This scenario is considered for both PLA and PET water bottles. A separate end-of-life scenario was included for PET water bottles to include the 23.5 percent recycling rate reported by NAPCOR.

Table 1

WEIGHTS FOR 10,000 PLA AND PET 12-OUNCE WATER BOTTLES
(Basis: 10,000 water bottles)

	Weight per unit		Weight per functional unit	
	(oz)	**(g)**	**(lb)**	**(kg)**
12-ounce Water Bottles				
PLA	0.74	21.0	463	210
PET	0.72	20.3	448	203

Source: Franklin Associates, a Division of ERG

[1] 2006 Report on Post Consumer PET Container Recycling Activity. Final Report. Prepared by NAPCOR and APR. October, 2007. Available at http://www.napcor.com

[2] Available at http://www.athenasmi.ca/projects/docs/Plastic_Products_LCA_Technical_Rpt.pdf.

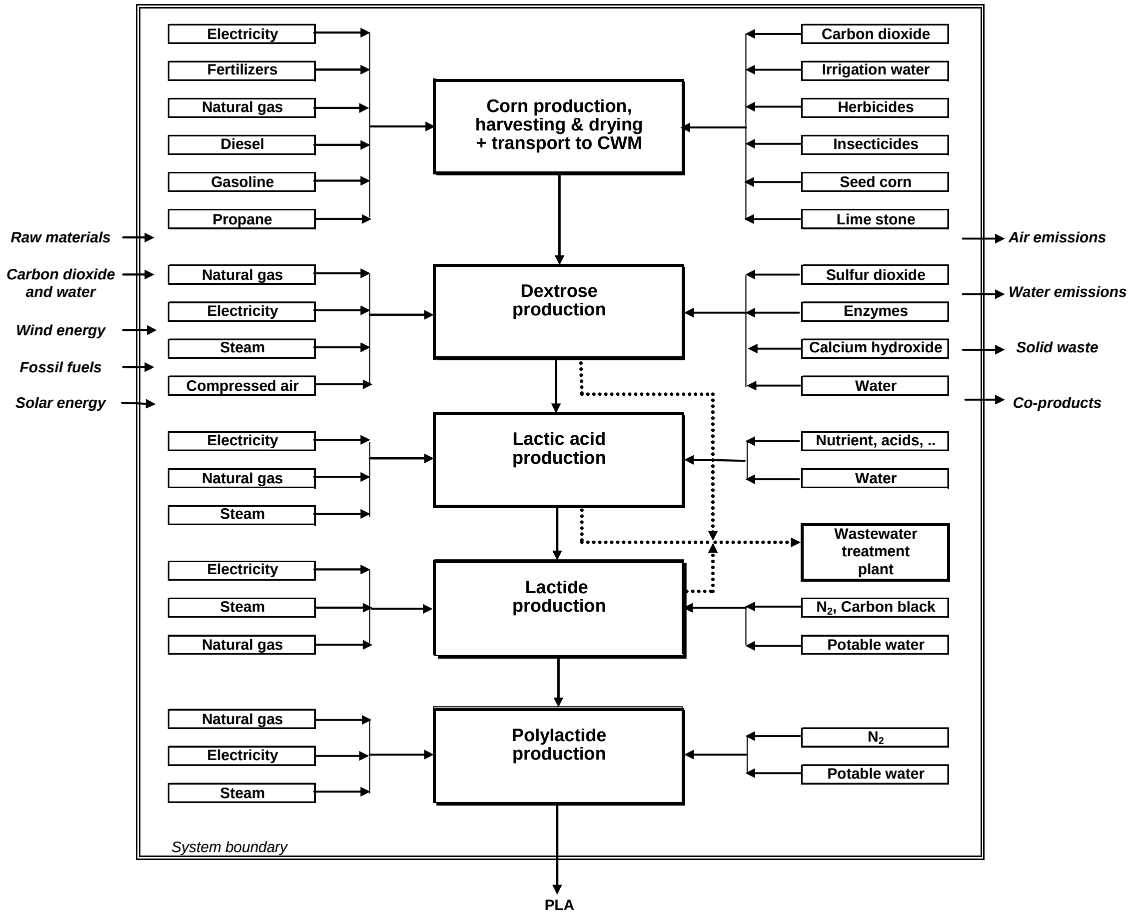

Figure 1. Simplified flow diagram and system boundary for the NatureWorks PLA resin production system. This flow diagram was taken from the 2006 draft journal paper provided by Mr. Erwin Vink of NatureWorks, LLC.

LIMITATIONS AND ASSUMPTIONS

Key assumptions of the LCI of water bottles are as follows:

- The caps and labels for each of the bottles are assumed to be equivalent and are, therefore, not included in the analysis.
- The PET resin data is taken from the U.S. LCI database (http://www.nrel.gov/lci/). The stretch blow molding fabrication process data comes from the PlasticsEurope database. This process data was placed in Franklin Associates models so that U.S. fuel pollutants could be calculated.
- The PLA LCI data was taken from the 2006 analysis, Life Cycle Inventory of Five Products Produced from PLA and Petroleum-Based Resins. In that study, Erwin Vink of NatureWorks provided a journal paper, which was under peer review, that included the NatureWorks 2005 PLA data used in this report. Mr. Vink's LCA of PLA uses the Boustead Model, which does include a U.S. fuels database.

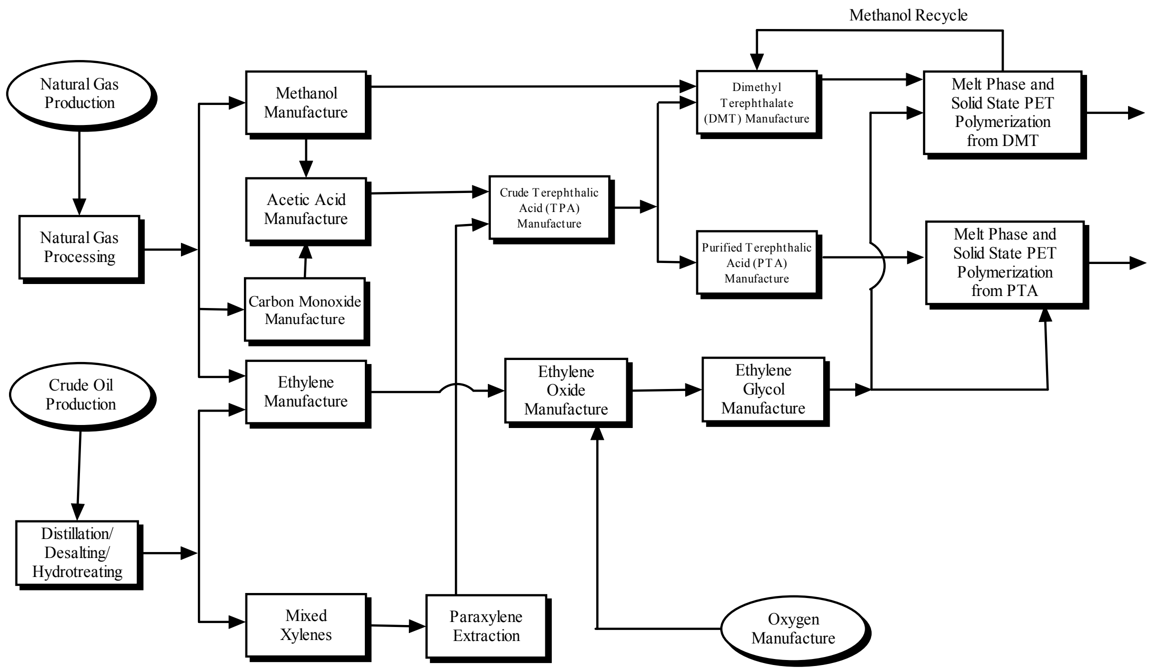

Figure 2. Flow diagram for the manufacture of virgin polyethylene terephthalate(PET) resin.

- Only the 2005 PLA dataset was used in this analysis. The choice not to present the 2006 PLA data which includes purchased wind energy was made based on the fact that any manufacturer of resin could buy those same credits. The PET dataset used in this report is based on industry averages while the PLA data represents one company, NatureWorks.
- Franklin Associates staff estimated the energy for the drying of PLA resin, a hygroscopic resin, from specifications found on ConAir's website for the dehumidifying dryer, CD1600. The kW provided on the specifications sheet represent maximum power expended for the dryer.
- Transportation from the resin producer to the product fabrication site was estimated using locations of actual U.S. resin plants and bottle fabrication plants. In this case, the following distances and modes were used for each resin type:
  - PLA—425 ton-miles by combination truck
  - PET—96 ton-miles by combination truck, 96 ton-miles by rail.
- Transportation to user and the use phase are considered equivalent for the water bottle systems and not included in this analysis.
- The disposal of the PLA and PET bottles include landfilling of postconsumer products, as well as a 20 percent waste-to-energy (WTE) combustion energy credit for the incineration of postconsumer products in mixed municipal solid waste. The energy credit given is based on gross higher heating values (HHV). Due to the fact that PET has an existing recycling infrastructure, a recycling rate of 23.5 percent[1] has been included for an additional PET water bottle system. The end-of-life

scenario used for that PET water bottle is 23.5% recycling, 15.3% WTE, and 61.2% landfilling.

- The higher heating values used for the resins analyzed in this summary report are PLA—19 MJ/kg and PET—26 MJ/kg.

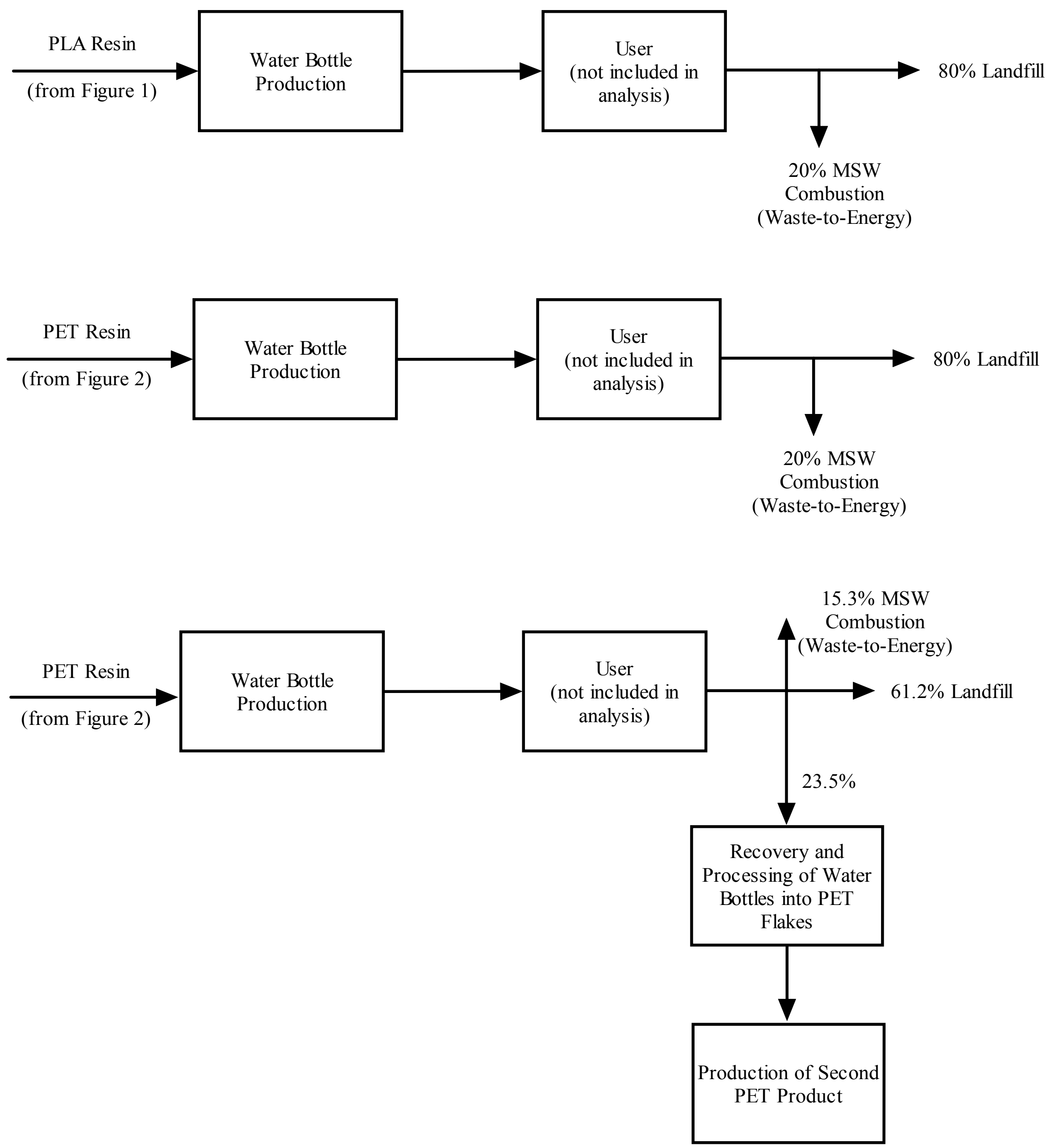

Figure 3. Flow diagram of the life cycle of the PLA and PET 12-ounce water bottles from transportation of resin to fabrication through end-of-life. Transportation to user and use phase are not included in this analysis.

- The global warming potentials used in this study were developed in 2001 by the International Panel of Climate Change (IPCC). The 100 year GWP used are as follows: fossil carbon dioxide—1, methane—23, nitrous oxide—296, CFC/HCFCs—1700, methylene chloride—10, HCFC22—1700.
- The Franklin Associates LCI models were used to calculate fuel production and delivery energy and emissions for drying, PLA resin transportation, and disposal steps. There may be some differences between the Franklin Associates model and the Boustead model. Appendix A addresses these differences at the end of this summary report.
- PET bottle recycling includes recovery and processing of waste PET into flakes. No chemical recycling of PET is considered in this study.

ADDITIONAL METHODOLOGY

As stated earlier in this report, the methodology for this report is the same as found in the report, "Life Cycle Inventory of Five Products Produced from PLA and Petroleum-Based Resins." However, recycling was not considered in that report and so a discussion of the recycling methodology follows.

In this study, open-loop recycling was evaluated for PET water bottles at the national average PET bottle recycling rate of 23.5 percent. In an open-loop system, a product made from virgin material is manufactured, recovered for recycling, and manufactured into a new product which is generally not recycled. This extends the life of the initial material, but only for a limited time. Thus, for open-loop recycling, the energy and emissions of virgin material manufacture, recycling, and eventual disposal of the recycled material are divided evenly between the first (the water bottle), and second product (a subsequent application). This analysis inherently assumes that the recycled material replaces virgin material when producing the second product.

LCI RESULTS SUMMARY

Based on the uncertainty in the data used for energy, solid waste, and emissions modeling, differences between systems are not considered meaningful unless the percent difference between systems is greater than the following:

- 10 percent for energy and postconsumer solid waste
- 25 percent for industrial solid wastes and for emissions data.

Percent difference between systems is defined as the difference between energy totals divided by the average of the two system totals. The minimum percent difference criteria were developed based on the experience and professional judgment of the analysts and are supported by sample statistical calculations (see Appendix B).

The complete LCI results include energy consumption, solid waste generation, and environmental emissions to air and water. A summary of the total energy, postconsumer solid waste, and total greenhouse gas (GHG) emissions results for the three water bottles is displayed in Table 2.

The PLA water bottles require significantly more net energy than either of the PET water bottle systems analyzed. This energy difference is a combination of many factors, including PLA's feedstock energy. The postconsumer solid waste is higher for the PLA water bottle compared to a PET water bottle including recycling. The weights of these two bottles are similar (see Table 1) and so the lower postconsumer solid waste for PET is due to the recycling of PET. When recycling is included the postconsumer solid waste for the PET water bottles is approximately 14 percent less than for the PLA water bottles. The postconsumer solid waste for the PET water bottles with no recycling is considered equivalent to the PLA water bottles. Although the total GHG emissions for the PET water bottles including recycling are a lower amount than those of the PLA water bottles, the totals are not considered significantly different (5 percent difference). The GHG totals of the PET water bottles with no recycling and the PLA water bottles are also not considered significantly different.

Table 2

TOTAL ENERGY, POSTCONSUMER SOLID WASTE, AND GREENHOUSE GASES FOR PLA AND PET 12-OUNCE WATER BOTTLES
(Basis: 10,000 water bottles)

	Net Energy (GJ)	Postconsumer Solid Waste (kg)	Greenhouse Gases (kg of CO2 equivalents)
12-ounce water bottles			
PLA (1)	19.0	168	744
PET (1)	16.6	163	757
PET (2)	15.2	144	710

(1) Disposal of solid waste is modeled with 80% going to a landfill and 20% combusted with energy recovery.
(2) Disposal of solid waste is modeled with 23.5% recovered for recycling, 61.2% going to a landfill, and 15.3% combusted with energy recovery.

Source: Franklin Associates, a Division of ERG calculations using original data from LCI/LCA by NatureWorks, LLC and the U.S. LCI Database

COMPLETE LCI RESULTS

Tables 3 through 8 display the complete LCI results for this analysis. The energy results are shown in Tables 3 and 4; the solid waste results are shown in Table 5; and the atmospheric and waterborne emissions are shown in Tables 6 through 8.

Energy

Due to the differences in the definitions of energy categories used in the Franklin Associates model and the Boustead model, only total and net energy are shown in Table 3. The combustion energy credit, which is the credit for the recovered energy from combustion of the final product in an incinerator, is shown separately in Table 3; however, the recovered energy from processes within the production of the resin are already included in the total energy. Also shown in Table 3 is a breakout of the cradle-to-resin and fabrication-to-grave total energy.

From Table 3, the PLA resin (cradle-to-resin) requires 80 percent of the total energy needed to make the water bottles; whereas, the resin transportation, drying, blow molding, and disposal require 20 percent of the total energy. This is also true for the PET water bottle with no recycling. In the case of the PET water bottles including recycling, the split is comparable at 77 percent of the total energy used for the cradle-to-resin, while 23 percent is used for resin transportation, blow molding, recycling, and disposal. When recycling is considered for PET, the greatest reduction in energy is for the feedstock energy, which is partially allocated to the second product. The PLA resin has also been given a feedstock energy in the NatureWorks report—most of this feedstock energy represents the corn used as raw material. It is true that the use of corn as a fuel (ethanol) has been increasing over the past few years. Franklin Associates does not commonly assign a fuel-energy equivalent to combustible biomass materials, such as corn, that are not major fuel sources in the U.S. However, the corn feedstock energy was included to follow NatureWorks' basic approach and methodology.

The total energy for the PLA water bottle is considered significantly higher than both of the PET water bottle systems total energy. Due to the similarities in weight and higher heating values, the combustion energy credit given to all bottle systems are comparable. The combustion energy credit for all systems is 6 percent or less of the total energy.

In Table 4, over 60 percent of the total energy required for the PLA water bottle system comes from fossil fuels, while fossil fuels are utilized for more than 90 percent of the total energy for the PET water bottles. The total energy from fossil fuels is 20 to 30 percent higher for the PET water bottles; most of this difference comes from the cradle-to-resin production. The PET resin fossil fuel energy includes the feedstock energy, which makes up approximately 45 percent of the total energy from fossil fuels. The feedstock energy is the energy sequestered in the petroleum and natural gas used as a material within the PET. The PLA resin was also given a feedstock energy as discussed previously. This feedstock energy is shown in the non-fossil fuel for PLA; it makes up over 70 percent of the total energy from non-fossil fuel for PLA water bottles. The non-fossil fuel energy shown for PET water bottles comes from the use of non-fossil fuels to produce electricity in the average U.S. grid.

Table 3

Energy by Process for 12-ounce Water Bottles
(MJ per 10,000 12-ounce water bottles)

	Total	Combustion Energy Credit (1)	Net Energy
12-ounce water bottle			
PLA (2005)			
Cradle-to-material	15,836		
Fabrication-to-Grave	3,955		
Total	19,791	798	18,993
PET (0% recycling)			
Cradle-to-material	14,087		
Fabrication-to-Grave	3,478		
Total	17,565	1,002	16,563
PET (23.5% recycling)			
Cradle-to-material	12,432		
Fabrication-to-Grave	3,686		
Total	16,118	884	15,234

	Total	Combustion Energy Credit (1)
12-ounce water bottle		
PLA (2005)		
Cradle-to-material	80%	
Fabrication-to-Grave	20%	
Total	100%	4%
PET (0% recycling)		
Cradle-to-material	80%	
Fabrication-to-Grave	20%	
Total	100%	6%
PET (23.5% recycling)		
Cradle-to-material	77%	
Fabrication-to-Grave	23%	
Total	100%	5%

(1) The combustion energy credit includes a credit for the recovered energy from combustion of the final product at an incinerator. Any recovered energy from the material produciton processes are subtracted out of the cradle-to-material total.

Source: Franklin Associates, a Division of ERG

Table 4

Energy by Fuel Type for 12-ounce Water Bottles
(MJ per 10,000 12-ounce water bottles)

	Fuel Type			Fuel Type (percent)		
	Fossil Fuel	Non-fossil Fuel	Total	Fossil Fuel	Non-fossil Fuel	Total
12-ounce water bottle						
PLA (2005)						
Cradle-to-material	9,603	6,233	15,836	49%	31%	80%
Fabrication-to-Grave	3,038	917	3,955	15%	5%	20%
Total	12,641	7,150	19,791	64%	36%	100%
PET (0% recycling)						
Cradle-to-material	13,683	404	14,087	78%	2%	80%
Fabrication-to-Grave	2,880	598	3,478	16%	3%	20%
Total	16,563	1,002	17,565	94%	6%	100%
PET (23.5% recycling)						
Cradle-to-material	12,076	356	12,432	75%	2%	77%
Fabrication-to-Grave	3,059	628	3,686	19%	4%	23%
Total	15,134	984	16,118	94%	6%	100%

Source: Franklin Associates, a Division of ERG

Solid Waste

Solid waste, shown in Table 5, is categorized into empirical categories as shown in the Boustead model used by NatureWorks. Also included in the solid waste table are the common Franklin Associates solid waste categories—process, fuel-related, and postconsumer wastes, which are the wastes discarded by the end users of the product.

No solid waste data were provided in Erwin Vink's journal paper for the PLA(2005) resin. The solid waste data shown for the PLA resin in Table 5 are estimated from the PLA (2006) dataset. Due to the differences in the way Franklin Associates and the Boustead model handle industrial solid wastes, no analysis has been attempted to group the two models' solid waste results into corresponding detailed waste categories.

Based on the U.S. average combustion of mixed municipal solid waste, 20 percent of the mixed waste sent to landfill is combusted in waste-to-energy facilities and therefore subtracted out of the total postconsumer wastes. The weight of postconsumer wastes is directly related to the weight of a product. In this case, the water bottles' weights are very similar. The postconsumer solid waste for the PLA water bottle system and PET water bottle with no recycling system directly correspond to the bottle weights. The difference in the amounts of postconsumer solid waste when comparing the PLA water bottle and PET water bottle with recycling is due to the recycling of the PET water bottles. This recycling was included only for PET because a well-established national recycling infrastructure is in place and 23.1 percent of PET soft drink/water bottles are being recycled. It is possible that a recycling infrastructure may be developed for PLA; however, at this time, there is none. There is a 16 percent difference between the PLA and PET (with recycling) water bottle systems. The PET water bottle with recycling system produces significantly less waste than the PLA water bottle system. The postconsumer solid waste for the PLA water bottles and PET water bottles with no recycling are not considered significantly different.

Table 5

Solid Wastes for 12-ounce Water Bottles
(g per 10,000 12-ounce water bottles)

	PLA (2005)			PET (0% recycling)			PET (23.5% recycling)		
	Cradle-to-PLA resin	Fab-to-Grave	Total (1)	Cradle-to-PET resin	Fab-to-Grave	Total	Cradle-to-PET resin	Fab-to-Grave	Total
Solid Waste Categories									
Plastics	210	0	210	0	0	0	0	0	0
Unspecified refuse	221	14.7	235	0	0	0	0	0	0
Mineral waste	3,869	7.35	3,876	0	0	0	0	0	0
Slags & ash	95.3	5,691	5,787	0	0	0	0	0	0
Mixed industrial	474	14.7	489	0	0	0	0	0	0
Regulated chemicals	928	29.4	958	0	0	0	0	0	0
Unregulated chemicals	239	14.7	253	0	0	0	0	0	0
Construction waste	0.42	0	0.42	0	0	0	0	0	0
Inert chemical	0.21	0	0.21	0	0	0	0	0	0
Waste to recycling	0.21	0	0.21	0	0	0	0	0	0
Waste returned to mine	2.73	7,791	7,794	0	0	0	0	0	0
Tailings	2,043	0.29	2,043	0	0	0	0	0	0
Municipal solid waste	0	1,485	1,485	0	0	0	0	0	0
Process solid waste	0	0	0	6,698	0	6,698	5,911	1,791	7,702
Fuel-related solid waste	0	0	0	21,706	25,231	46,937	19,155	26,529	45,684
Postconsumer solid waste	0	168,000	168,000	0	162,600	162,600	0	143,500	143,500

(1) No solid waste data were provided in Mr. Vink's journal paper for the PLA(2005) resin. The data shown for the resin is estimated from the PLA(2006) dataset and does not include the solid waste reduction associated with the purchase of wind energy credits

Source: Franklin Associates, a Division of ERG

Environmental Emissions

Atmospheric and waterborne emissions for each system include emissions from processes and those associated with the combustion of fuels. Table 6 presents atmospheric emissions results and Table 8 shows waterborne emissions for 10,000 12-ounce water bottles. Table 7 gives a greenhouse gas summary for each of the water bottle systems analyzed.

It is important to realize that interpretation of air and water emission data requires great care. The effects of the various emissions on humans and on the environment are not fully known. The degree of potential environmental disruption due to environmental releases is not related to the weight of the releases in a simple way. No firm conclusions can be made from the various atmospheric or waterborne emissions that result from the product systems. The atmospheric and waterborne emissions shown here represent systems totals and are not broken out by life cycle stage or process and fuel-related emissions.

Table 6

Atmospheric Emissions for 12-ounce Water Bottles
(g per 10,000 12-ounce water bottles)

	PLA (2005)	PET (0% recycling)	PET (23.5% recycling)
Atmospheric Emissions			
dust (PM10)	2,128	64.8	59.3
CO	2,269	3,701	3,294
CO2	628,996	682,021	641,096
SOX as SO2	2,806	5,328	4,959
H2S	0.22	0	0
mercaptan	4.5E-04	0.80	0.71
NOX as NO2	3,721	1,936	1,811
NH3	1.27	7.54	6.68
Cl2	0.033	0.0052	0.0047
HCl	104	80.4	78.7
F2	2.1E-05	0	0
HF	4.19	9.88	9.68
hydrocarbons not specified elsewhere	433	1,442	1,275
aldehyde (-CHO)	0.48	40.0	35.4
organics	15.8	227	200
Pb+compounds as Pb	0.0020	0.045	0.043
Hg+compounds as Hg	3.9E-04	0.011	0.010
metals not specified elsewhere	0.18	0.062	0.058
H2SO4	0.0028	0	0
N2O	78.1	14.9	14.1
H2	63.8	0	0
dichloroethane (DCE) C2H4Cl2	4.2E-05	1.5E-04	1.3E-04
vinyl chloride monomer (VCM)	6.9E-04	0	0
CFC/HCFC/HFC not specified elsewhere	1.3E-06	1.8E-05	1.6E-05
organo-chlorine not specified elsewhere	2.10	0.0083	0.0076
CH4	4,015	3,059	2,793
aromatic HC not specified elsewhere	0.22	0.0091	0.0083
polycyclic hydrocarbons (PAH)	7.7E-05	0.0014	0.0013
NMVOC	77.4	144	130
methylene chloride CH2Cl2	0.0016	0.035	0.033
Cu+compounds as Cu	9.2E-05	4.1E-04	3.7E-04
As+compounds as As	0.0018	0.031	0.030
Cd+compounds as Cd	2.9E-04	0.0067	0.0063
Zn+compounds as Zn	4.7E-04	2.7E-04	2.5E-04
Cr+compounds as Cr	0.0012	0.022	0.021
Se+compounds as Se	0.0051	0.090	0.088
Ni+compounds as Ni	0.0076	0.24	0.22
Sb+compounds as Sb	1.3E-04	0.0012	0.0012

Table 6 (cont'd)

Atmospheric Emissions for 12-ounce Water Bottles
(g per 10,000 12-ounce water bottles)

	PLA (2005)	PET (0% recycling)	PET (23.5% recycling)
Atmospheric Emissions			
dioxin/furan as Teq	3.6E-07	2.8E-06	2.7E-06
benzene C6H6	0.17	8.24	7.57
toluene C7H8	0.24	12.0	11.1
xylenes C8H10	0.14	15.3	13.8
ethylbenzene C8H10	0.018	0.93	0.86
styrene	4.6E-08	9.2E-05	8.2E-05
propylene	0.040	0.052	0.047
Fe+compounds as Fe	4.8E-04	0	0
Co+compounds as Co	8.3E-04	0.022	0.021
V+compounds as V	0.0025	0	0
Al+compounds as Al	-0.87	0	0
B+compounds as B	0.0011	0	0
Manganese	0.0025	0.044	0.042
Molybdenum	2.1E-05	0	0
Corn dust	15.8	0	0
Tin	1.1E-04	0	0
Titanium	2.1E-05	0	0
Barium	0.074	0	0
Beryllium	8.6E-05	0.0017	0.0016
Bromine	8.8E-04	0	0
Cyanide (unspecified)	1.9E-04	0.0092	0.0082
Fluoride (unspecified)	3.6E-04	0.17	0.15
Helium	0.080	0	0
VOC (volatile organic compou	0.051	0	0
Dust (PM 2.5)	3.14	0.041	0.036
Dust (unspecified)	29.1	290	277
Ethanol	95.6	0	0
Lactic acid	0.18	0	0
Particles (< 2.5 um)	-4.39	0	0
Particles (> 10 um)	-53.66	0	0
Particles (<10 and > 2.5 um)	-48.03	0	0

Source: Franklin Associates, a Division of ERG

The atmospheric emissions shown in Table 6 and the waterborne emissions shown in Table 8 are at times magnitudes apart for the two systems. As two different models were used, there are bound to be differences in the results. Some of the reasons for this are differences in methodology, data sources, and actual differences in the emissions from diverse processes. A more in-depth discussion of potential differences can be found in Appendix A.

This analysis is not an LCIA (life cycle impact assessment) and thus the impacts of various environmental emissions are not evaluated. However, due to the scientifically accepted relationship between greenhouse gases and global warming, it is reasonable to develop conclusions based on the quantity of greenhouse gases generated by a system. Greenhouse gas emissions are expressed as carbon dioxide equivalents, which use global warming potentials developed by the International Panel on Climate Change (IPCC) to normalize the various greenhouse gases to an equivalent weight of carbon dioxide. The 100-year time horizon Global Warming Potentials for GHG was used for this analysis.

Table 7

Greenhouse Gas Summary for 12-ounce Water Bottles
(g carbon dioxide equivalents per 10,000 12-ounce water bottles)

	PLA (2005)	PET (0% recycling)	PET (23.5% recycling)
CO2	628,996	682,021	641,096
N2O	23,128	4,421	4,186
CFC/HCFC/HFC not specified elsewhere	0.0022	0.031	0.028
CH4	92,351	70,353	64,231
methylene chloride CH2Cl2	0.016	0.35	0.33
Total	744,475	756,796	709,514

Note: The 100 year global warming potentials used in this table are as follows: fossil carbon dioxide--1, nitrous oxide--296, CFC/HCFCs--1700, methane--23, methylene chloride--10.

Source: Franklin Associates, a Division of ERG

Although the PET water bottle system with no recycling produces more grams of CO_2 equivalents than the PLA water bottle system, there is actually only a 2 percent difference, and so the amounts cannot be considered significantly different. The same is true of the CO_2 equivalents released for the PLA water bottle system and the PET water bottle system with recycling. Much of the fossil fuel used in the PET water bottles is from feedstock energy, which is bound within the product and therefore does not produce greenhouse gases.

Fossil carbon dioxide makes up 84 percent of the total grams of CO_2 equivalents for the PLA water bottle and 90 percent of the total grams of CO_2 equivalents for both of the PET water bottle systems. Methane also comprises a significant portion of the total grams of CO_2 equivalents for all three systems.

Table 8

Waterborne Emissions for 12-ounce Water Bottles
(g per 10,000 12-ounce water bottles)

	PLA (2005)	PET (0% recycling)	PET (23.5% recycling)
Waterborne Wastes			
COD	1,245	619	550
BOD	228	356	331
Pb+compounds as Pb	0.027	0.51	0.46
Fe+compounds as Fe	10.6	135	121
Na+compounds as Na	646	11,408	10,242
acid as H+	0.28	8.29	7.37
NO3-	253	0.097	0.096
Hg+compounds as Hg	4.3E-05	6.8E-04	6.1E-04
ammonium compounds as NH4+	0.34	0.30	0.27
Cl-	2,087	17,707	16,246
CN-	1.1E-04	8.1E-05	7.3E-05
F-	0.85	0.63	0.62
S+sulphides as S	0.0021	0.029	0.026
dissolved organics (non-hydrocarbon)	0.11	0	0
suspended solids	810	1,932	1,751
detergent/oil	1.11	24.3	21.8
hydrocarbons not specified elsewhere	0.34	0.099	0.091
organo-chlorine not specified elsewhere	4.2E-04	4.5E-05	4.1E-05
dissolved chlorine	3.8E-04	0	0
phenols	0.026	0.54	0.49
dissolved solids not specified elsewhere	2,154	49,905	44,806
P+compounds as P	2.52	0	0
other nitrogen as N	17.7	0	0
other organics not specified elsewhere	0.20	7.81	6.95
SO4--	36.5	145	136
vinyl chloride monomer (VCM)	2.1E-05	0	0
K+compounds as K	0.26	0	0
Ca+compounds as Ca	182	3,599	3,231
Mg+compounds as Mg	30.6	704	632
Cr+compounds as Cr	0.11	3.01	2.68
ClO3--	0.013	0	0
BrO3--	6.3E-05	0	0

Table 8 (cont'd)

Waterborne Emissions for 12-ounce Water Bottles
(g per 10,000 12-ounce water bottles)

	PLA (2005)	PET (0% recycling)	PET (23.5% recycling)
Waterborne Wastes			
TOC	329	10.8	9.60
AOX	4.2E-05	0	0
Al+compounds as Al	3.87	19.0	17.4
Zn+compounds as Zn	0.090	3.09	2.75
Cu+compounds as Cu	0.014	0.25	0.23
Ni+compounds as Ni	0.013	0.25	0.23
CO3--	0.055	0	0
As+compounds as As	0.013	0.28	0.25
Cd+compounds as Cd	0.0020	0.042	0.037
Mn+compounds as Mn	0.10	2.02	1.88
Ag+compounds as Ag	0.10	2.35	2.11
Ba+compounds Ba	52.7	850	759
Sr+compounds as Sr	2.64	61.1	54.9
V+compounds as V	0.0013	0.030	0.027
benzene	0.081	1.88	1.69
dioxin/furan as Teq	9.2E-06	3.1E-04	2.8E-04
Mo+compounds as Mo	0.0011	0.026	0.023
Ca++	52.0	0	0
PO4(-3)	0.051	0	0
Chromium +III	0.0016	0	0
Chromium +IV	1.1E-04	0	0
Heavy metals unspecified	7.60	598	542
Selenium	0.0013	0.021	0.020
Titanium	0.037	0.58	0.52
Chlorine dissolved	0.0011	0	0
Fluorine	2.5E-04	0	0
Neutral salts	0.0054	0	0
halogenated organics	0.016	0.0020	0.0018

Source: Franklin Associates, a Division of ERG

SUMMARY OF KEY FINDINGS

The following points summarize the key findings in this analysis.

- The PLA water bottles require significantly more total and net energy than the PET water bottle systems (with or without the inclusion of recycling).
- The total energy from fossil fuels is 20 to 30 percent higher for the PET water bottles; most of this difference comes from the cradle-to-resin production.
- The postconsumer solid waste is higher for the PLA water bottle compared to a PET water bottle including recycling. The postconsumer solid waste for the PET water bottles with no recycling is considered equivalent to that of the PLA water bottles.
- The carbon dioxide equivalent totals (GHG) for all systems are within 5 percent of each other and so are not considered significantly different.

APPENDIX A

POTENTIAL DIFFERENCES BETWEEN THE FRANKLIN ASSOCIATES AND BOUSTEAD MODEL/METHODOLOGY

INTRODUCTION

This appendix presents some potential differences between the models and methodology used for the PET water bottles and the PLA water bottles. The purpose of this comparison was to highlight the differences between the two separate models and methodology used in this analysis. The analysis focuses on differences in energy, since fuel-related emissions results are dependent on energy results.

Franklin Associates has had previous in-depth discussions with Ian Boustead of Boustead Consulting, Ltd. to discuss differences in ACC Plastics Division resin LCI results and PlasticsEurope LCI results for corresponding resins. The differences in modeling methodologies and data sources identified in these discussions are reflected here.

DATA SOURCES

All primary data for the U.S. plastics LCI database, from which the PET resin data was used, were collected between 2003 and 2005. These data represent the year 2003 for the most part. Franklin Associates and APC were diligent in finding at least three companies to participate in collecting data for each resin and precursor studied. However, in the case of PET, there was limited participation by producers due to confidentiality concerns and issues for U.S. companies. The data collection effort focused on the resin production step, as well as intermediate chemicals that the resin producers manufactured.

NatureWorks produced their own LCI using data collected from their one plant and using Boustead Consulting's LCI software. Boustead Consulting is a European LCI consulting firm known for collecting and updating various plastic Eco-Profiles for PlasticsEurope over the past 15 years.

One difference between the Franklin and Boustead models is the fuel data. The fuels information in Boustead's European database is based on statistics published by the International Energy Agency (IEA). It is unknown what specific data sources for fuel production and combustion are used by the Boustead Model to represent U.S. fuel production and combustion. Franklin Associates' fuel production and combustion data for the U.S. were based on Department of Energy national statistics and data. The national average U.S. electricity grid (from the U.S. LCI Database) was used.

SYSTEM BOUNDARIES

Another potential difference in the models may be in the system boundaries for the resins. In some of the emissions categories, negative amounts are displayed for the PLA water bottles, which appear to reflect emissions credits, particularly in the case of carbon dioxide, where carbon dioxide uptake during corn growing is subtracted from the carbon dioxide released during processing.

METHODOLOGICAL DIFFERENCES

Coproduct Allocation Method. Simple mass allocation is a common type of allocation used for coproducts in LCI/LCA. For the PET resin dataset, mass allocation was used for material coproducts. Recovered energy credit was given for energy coproducts. NatureWorks did not discuss in their draft Journal paper what coproducts are produced with the PLA (2005). A gypsum coproduct was shown for PLA(2006); in that case, NatureWorks stated that a credit was given equal to the avoided gypsum mining.

Fuel Infrastructure. For the PET resin data, an overall average U.S. electricity grid was used. As only one NatureWorks plant was considered for the PLA LCI dataset and the Boustead Model has available regional U.S. electricity grids, it is likely that a specific electricity grid was used to represent electricity used at the PLA production facility.

APPENDIX B

CONSIDERATIONS FOR INTERPRETATION OF DATA AND RESULTS

INTRODUCTION

An important issue with LCI results is whether two numbers are really different from one another. For example, if one product has a total system requirement of 100 energy units, is it really different from another product system that requires 110 energy units? If the error or variability in the data is sufficiently large, it cannot be concluded that the two numbers are actually different.

STATISTICAL CONSIDERATIONS

A statistical analysis that yields clear numerical answers would be ideal, but unfortunately LCI data are not amenable to this. The data are not (1) random samples from (2) large populations that result in (3) "normal curve" distributions. LCI data meet none of these requirements for statistical analysis. LCI data for a given sub-process (such as potato production, roundwood harvesting, or caustic soda manufacture, for example) are generally selected to be representative of a process or industry, and are typically calculated as an average of two to five data points. In statistical terminology, these are not random samples, but "judgment samples," selected so as to reduce the possible errors incurred by limited sampling or limited population sizes. Formal statistics cannot be applied to judgment samples; however, a hypothetical data framework can be constructed to help assess in a general sense the reliability of LCI results.

The first step in this assessment is reporting standard deviation values from LCI data, calculated by:

$$s = \sqrt{\frac{\sum (x_1 - x_{mean})^2}{n-1}},$$

where x_i is a measured value in the data set and x_{mean} is the average of n values. An analysis of sub-process data from Franklin Associates, Ltd. files shows that, for a typical sub-process with two to five different companies supplying information, the standard deviation of the sample is about 30 percent of the sample average.

In a typical LCI study, the total energy of a product system consists of the sum of many sub-processes. For the moment, consider an example of adding only two numbers. If both numbers are independent of each other and are an average of measurements which have a sample standard deviation, s, of 30, the standard deviation of the sum is obtained by adding the variances of each term to form the sum of the variances, then taking the square root. Variances are calculated by squaring the standard deviation, s^2, so the sum of the variances is $30^2 + 30^2 = 900 + 900 = 1800$. The new standard deviation of the sum is the square root of the sum of the variances, or $\sqrt{1800} = 42.4$. In this example, suppose both average values are 100, with a sum of 200. If reported as a percent of the sum, the new standard deviation is 42.4/200 = 21.3% of the sum. Another way of obtaining this value is to use the formula $s_{\%} = \frac{s/x_{mean}}{\sqrt{n}}$, where the term $s_{\%}$ is defined as the standard deviation of n data points, expressed as a % of the average, where each entry has approximately the same standard deviation, s. For the example, then, $s_{\%} = \frac{30\%}{\sqrt{2}} = 21.3\%$.

Going back to a hypothetical LCI example, consider a common product system consisting of a sum of approximately 40 subsystems. First, a special hypothetical case is examined where *all of the values are approximately the same size, and all have a standard deviation of 30%*. The standard deviation in the result is $s_{\%} = \frac{30\%}{\sqrt{40}} = 4.7\%$. The act of summing reduces the standard deviation of the result with respect to the standard deviation of each entry because of the assumption that errors are randomly distributed, and by combining values there is some cancellation of total error because some data values in each component system are higher than the true values and some are lower.

The point of this analysis, however, is to compare two results, e.g., the energy totals for two different product systems, and decide if the difference between them is significant or not. To test a hypothetical data set it will be assumed that two product systems consist of a sum of 40 values, and that the standard deviation, $s_{\%}$, is 4.7% for each product system.

If there is statistical knowledge of the sample only, and not of the populations from which they were drawn, "t" statistics can be used to find if the two product totals are different or not. The expression selected is:

$\mu 1 - \mu 2 = x1 - x2 +/- t_{.025}\, s' \sqrt{\frac{1}{n1} + \frac{1}{n2}}$, where $\mu 1 - \mu 2$. is the difference in population means, x_1-x_2 is the difference in sample means, and s' is a pooled standard deviation of the two samples. For the hypothetical case, where it is assumed that the standard deviation of the two samples is the same, the pooled value is simply replaced with the standard deviation of the samples.

The goal is to find an expression that compares our sample means to "true," or population, means. A new quantity is defined: $\Delta = (\mu 1 - \mu 2) - (x1 - x2)$, and the sample sizes are assumed to be the same (i.e., n_1=n_2). The result is $\Delta = t_{.025} s' \sqrt{\frac{2}{n}}$, where Δ is the minimum difference corresponding to a 95%

confidence level, s' is the standard deviation of the sum of n values, and $t_{.025}$ is a t statistic for 95% confidence levels. The values for t are a function of n and are found in tables. This expression can be converted to percent notation by dividing both sides by the average of the sample means, which results in $\Delta\% = t_{.025} s'\% \sqrt{\frac{2}{n}}$, where $\Delta\%$ is now the percent difference corresponding to a 95% confidence level, and s'% is the standard deviation expressed as a percent of the average of the sample means. This formula can be simplified for the example calculation by remembering that $s'\% = \frac{s\%}{\sqrt{n}}$, where s% is the standard deviation of each energy entry for a product system. Now the equation becomes $\Delta\% = t_{.025} s\% \frac{\sqrt{2}}{n}$. For the example, t = 2.0, s = 30%, and n = 40, so that $\Delta\%$ = 2.1%. This means that if the two product system energy totals differ by more than 2.1%, there is a 95% confidence level that the difference is significant. That is, if 100 independent studies were conducted (in which new data samples were drawn from the same population and the study was conducted in the identical manner), then 95 of these studies would find the energy values for the two product systems to differ by more than 2.1%.

The previous discussion applies only to a hypothetical and highly idealized framework to which statistical mathematics apply. LCI data differ from this in some important ways. One is that the 40 or so numbers that are added together for a final energy value of a product system are of widely varying size and have different variances. The importance of this is that large numbers contribute more to the total variance of the result. For example, if 20 energy units and 2,000 energy units are added, the sum is 2,020 energy units. If the standard deviation of the smaller value is 30% (or 6 units), the variance is 6^2 = 36. If the standard deviation of the larger number is 10% (or 200), the variance is 200^2 = 40,000. The total variance of the sum is 36 + 40,000 = 40,036, leading to a standard deviation in the sum of $\frac{\sqrt{(40036)}}{2020}$ = 9.9%. Clearly, the variance in the result is much more greatly influenced by larger numbers. In a set of LCI energy data, standard deviations may range from 10% to 60%. If a large number has a large percentage standard deviation, then the sum will also be more uncertain. If the variance of the large number is small, the answer will be more certain. To offset the potential problem of a large variance, Franklin Associates goes to great lengths to increase the reliability of the larger numbers, but there may simply be inherent variability in some numbers which is beyond the researchers' control.

If only a few numbers contribute most of the total energy in a system, the value of $\Delta\%$ goes up. This can be illustrated by going back to the formula for $\Delta\%$ and calculating examples for n = 5 and 10. From statistical tables, the values for $t_{.025}$ are 2.78 for n = 5, and 2.26 for n = 10. Referring back to the hypothetical two-product data set with s% = 30% *for each entry*, the corresponding values for $\Delta\%$ are 24% for n = 5 and 9.6% for n = 10. Thus, if only 5 numbers out of 40 contribute most of the energy, the percent *difference* in the two product system energy values must increase to 24% to achieve the 95% confidence level that the two values are different. The minimum difference decreases to 9.6% if there are 10 major contributors out of the 40 energy numbers in a product system.

CONCLUSIONS

The discussion above highlights the importance of sample size, and of the variability of the sample. However, once again it must be emphasized that the statistical analysis does not apply to LCI data. It only serves to illustrate the important issues. Valid standard deviations cannot be calculated because of the failure of the data to meet the required statistical formula assumptions. Nevertheless, it is important to achieve a maximum sample size with minimum variability in the data. Franklin Associates examines the data, identifies the large values contributing to a sum, then conducts more intensive analysis of those values. This has the effect of increasing the number of data points, and therefore decreasing the "standard deviation." Even though a calculated standard deviation of 30% may be typical for Franklin Associates' LCI data, the actual confidence level is much higher for the large values that control the variability of the data than for the small values. However, none of this can be quantified to the satisfaction of a statistician who draws conclusions based upon random sampling. In the case of LCI data, it comes down to a matter of professional judgment and experience. The increase in confidence level resulting from judgment and experience is not measurable.

It is the professional judgment of Franklin Associates, based upon over 25 years of experience in analyzing LCI data, that a 10% rule is a reasonable value for $\Delta\%$ for stating results of product system energy totals. That is, if the energy of one system is 10% different from another, it can be concluded that the difference is significant. It is clear that this convention is a matter of judgment. This is not claimed to be a highly accurate statement; however, the statistical arguments with hypothetical, but similar, data lend plausibility to this convention.

We also conclude that the weight of postconsumer solid waste data can be analyzed in a similar way. These data are at least as accurate as the energy data, perhaps with even less uncertainty in the results. Therefore, the 10% rule applies to postconsumer solid waste weight. However, we apply a 25% rule to the solid waste volume data because of greater potential variability in the volume conversion factors.

Air and water pollution and industrial solid waste data are not included in the 10% rule. Their variability is much higher. Data reported by similar plants may differ by a factor of two, or even a factor of ten or higher in some cases. Standard deviations may be as high as 150%, although 75% is typical. This translates to a hypothetical standard deviation in a final result of 12%, or a difference of at least 25% being required for a 95% confidence of two totals being different if 10 subsystems are major contributors to the final results. However, this rule applies only to single emission categories, and cannot be extended to general statements about environmental emissions resulting from a single product system. The interpretation of environmental emission data is further complicated by the fact that not all plants report the same emission categories, and that there is not an accepted method of evaluating the relative importance of various emissions.

It is the intent of this appendix to convey an explanation of Franklin Associates' 10% and 25% rules and establish their plausibility. **Franklin Associates' policy is to consider product system totals for energy and weight of postconsumer solid waste weight to be different if there is at least a 10% difference in the totals. Otherwise, the difference is considered to be insignificant. In the detailed tables of this report there are many specific pollutant categories that are variable between systems. For the air and waterborne emissions, industrial solid waste, and postconsumer solid waste volume, the 25% rule should be applied.** The formula used to calculate the difference between two systems is:

$$\% \text{ Diff} = \left(\frac{x-y}{\frac{x+y}{2}} \right) \text{ X } 100,$$

where x and y are the summed totals of energy or waste for two product systems. The denominator of this expression is the average of the two values.