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Bio-Based Versus Conventional Plastics for Electronics Housings: LCA Literature Review
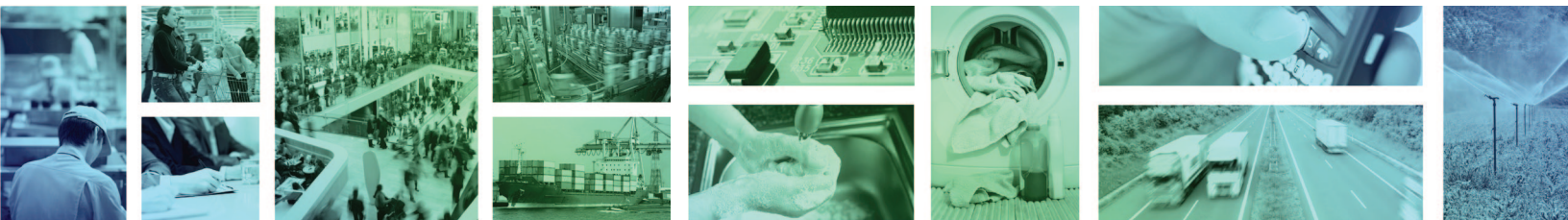
Dr. Sarah B. Boyd
University of California, Berkeley
The Sustainability Consortium

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Bio-based versus conventional plastics for electronics housings: LCA literature review (White Paper)

Sarah B. Boyd[1]

The use of bio-based housing materials is included as an optional criterion for EPEAT with the acknowledgement that further research was needed before the use of bio-based materials could be endorsed by the standard. In order to evaluate the benefit of the use of bio-based materials, we compare the impacts associated with production of currently dominant plastic blends with those of the most promising bio-based material (polylactic acid or PLA), as either the principal constituent or supplementary fraction of housing plastic.

While environmental impact assessment data indicates that PLA is nominally environmentally advantageous over the existing plastics used for laptop housings when considering all life-cycle stages but the end-of-life (EOL), the fate and chemical behavior of PLA at EOL are unknown at this point, and thus the potential impacts at EOL are highly variable.

Current housing materials: ABS and PC

The most popular housing material for laptop computers at this time, based on responses from manufacturers and review of the literature, is a mixture of polycarbonate (PC) and acrylonitrile butadiene styrene (ABS), with the typical blend composed of 60-70% PC and 20-30% ABS with the remaining fraction (typically about 10%) composed of additives. High-impact polystyrene, while still used commonly in printers and other electronic equipment, is no longer commonly used in laptop casing.

Four sources of LCA data which report total cradle-to-gate global warming potential (GWP) for ABS have been identified (Boustead 2005 and 1997, Levy 2007, Ecoinvent v1.1). The initial publication describing the LCI of ABS (Boustead 1997) from the Association of Plastics Manufacturers in Europe (APME, now Plastics Europe) has been superseded by a more recent 2005 Plastics Europe study, by the same lead author (Boustead 2005), reducing the number of unique, relevant studies to three.

Though a complete critical review of the 2005 Plastics Europe study (Boustead 2005) is not included in this report, some notes are made of the methodology. The methodological overview document for the most recent Plastics Europe LCI studies (Plastics Europe 2009) describes the boundary as including over 98% of total mass inputs, 98% of all energy inputs, and all materials of known environmental significance

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(those with significant impacts associated with extraction, manufacture, use or disposal, or known toxicity), and describes the use of quantitative sensitivity analysis for boundary checking. Interestingly, the Plastics Europe data does not include impacts associated with infrastructure. The main drawback to this data source is that the electricity emissions factor represents an average across Europe and inventory data is not reported in a way which allows application of differing geographical electricity emissions factors.

The Ecoinvent dataset for plastics was developed by EMPA (the Swiss Federal Laboratories for Materials Testing and Research) and is based primarily on LCI data from APME (Plastics Europe). Quality control and boundary checking is described in the Ecoinvent supporting documentation (Frischknecht 2005, 2007). In an overview methodology paper, which describes the general approach across all Ecoinvent datasets (Frischknecht 2005), the boundary is said to be determined by "expert knowledge." The description of boundary definition implies that quantification and sensitivity analysis is not used in every case but may be used by some Ecoinvent analysts, if data is available. In the case of plastics, the LCA boundary is inherited from the Plastics Europe LCIs, with review by Ecoinvent. The impact values reported in the Ecoinvent dataset for PC and ABS do not vary considerably from the Plastics Europe results, indicating that EMPA found no considerable gaps, errors in boundary definition in the original Plastics Europe inventories, nor error in impact evaluation.

For PC, two sources of LCA data are similarly available from Plastics Europe and Ecoinvent, and these values also compare closely, while data from the ACC is not available for PC. Values for both plastics from all sources are given in Table 1.

	Primary energy (MJ/kg)	GWP (kg CO_2eq./kg)	Source	Location
ABS	93.3	3.15	ACC (Levy 2007)	US/Canada
ABS	95.3	3.76	Plastics Europe (Boustead 2005)	Europe
ABS	95.2	3.16	Ecoinvent v1.1 (2004)	Europe
PC	113	7.62	Plastics Europe (Boustead 2005)	Europe
PC	104	4.98	Ecoinvent v1.1 (2004)	Europe

Table 1: Primary energy demands and GWP impacts for ABS and PC

PLA

Polylactide or polylactic acid (PLA) is identified through review of the literature and discussion with working group members as the leading contender for a bio-based material for use in electronics housing plastic. Although the material has not been adopted widely, it shows potential in this application area. PLA has been used by Fujitsu in the FMV-BIBLIO NB80-K notebook model, released in mid-2007 in Japan, and worldwide in 2008. In the FMV-BIBLIO, PLA is used in a blend with other plastics, which are not identified to the public, and additives.

The only PLA LCI or LCA data available has been reported in a set of studies from NatureWorks (Vink 2003, 2007), the latter of which was reviewed by Ian Boustead (known for his work on the LCAs from APME and PlasticsEurope) and published in a peer reviewed publication. While several other life-cycle analyses of PLA-containing products have been performed, these other studies all reference the original data provided in the Vink reports. The scarcity of LCA data for PLA makes the task of validating the studies difficult. However, although there has been only one LCA of PLA, there has been a great deal of work done on LCA of corn grown in the US, due to the interest in corn-derived ethanol as a competitive source of renewable fuel. Many of the difficult questions related to bio-geochemical activities and related emissions and impacts from corn farming have thus been addressed in work done at in LCA of corn ethanol.   Argonne National Lab, UC Davis, UC Berkeley, and several other academic institutions have contributed to the study of the environmental impacts of corn production in the US (Wang 1999, Wang 2007, Shapouri 2002, Kendall 2009). As a result, the methodology and assumptions used in the NatureWorks study can be checked against those used in LCA of corn (and corn-derived ethanol).

In Table 2, factors which are included in existing LCA of US-grown corn are listed. The factors which may contribute a significant fraction to total GWP, but are often left out of corn ethanol LCA, are soil N_2O emissions, co-product allocation, and accounting of carbon release due to land use change (Kendall 2009). In the Vink study, soil N_2O emissions are apparently included, as well as co-product allocation, while land use change is not considered. A recent review article asserts that land use change is a small contributor, when applicable (Liska 2009).

If the corn used in production of PLA is produced from active land (rather than from land converted from forest to agricultural use) direct land use change is not applicable. Given that the total fraction of the corn market which could be used in production of PLA for electronic housings is very small, the omission of land use change is deemed reasonable in this case. Indirect land use change, or land use change which would result from additional land being converted to agricultural use to replace the corn production used in the PLA, is included in some studies of bio-ethanol, but is also not considered in the Vink study. Consideration of indirect land use change, which is a consequential LCA method, is not necessarily applicable in the case of the use of PLA for laptop housings, as this application represents a demand for a much smaller fraction of total corn production than, for example, ethanol for fuel. In the scenario that one million laptops were produced in a given year, and used a housing composed entirely of PLA (0.8 kg PLA/laptop), roughly 55,000 bushels of corn or 370 acres of corn production would be needed as inputs to production, representing 5 millionths of the total annual production of corn in the US. Given that a great deal of corn is wasted before or after conversion into food, this fraction is below a threshold that would merit fair consideration of the consequential LCA. In addition, use of consequential methods to quantify indirect land use change requires the equivalent use of consequential LCA for indirect fuel or fossil hydrocarbon production for the case of traditional plastics. Given that both options (corn-derived PLA and fossil fuel-based plastic) incur impacts indirectly and may largely cancel one another out, and because there are so many uncertainties for each consequential LCA scenario, indirect impacts are not considered significant enough to warrant inclusion in this particular comparison.

LCA-relevant stages of corn production	Included in Vink PLA LCA?
Direct land use change	No
Indirect land use change	No
Equipment used in planting and harvesting	No
Pest control and fertilizers	Yes
Soil carbon emissions (from lime)	Possibly, not explicitly
Vegetation GHG emissions	Possibly, not explicitly
N_2O emissions due to fertilizers	Yes
Fuel used in corn growing, harvesting, drying	Yes
Electricity used in corn growing, harvesting, drying	Yes
Co-product allocation	Yes
Transport of corn	Yes

Table 2: Life-cycle inventory stages in LCA of corn

Both corn ethanol LCA methodology review papers (Kendall 2009, Liska 2009) agree that the most important factors to consider in evaluating energy use and GWP in production of corn for technical products (such as ethanol or PLA) are: direct N_2O and CO_2 emissions from soil due to nitrogen fertilizers and lime, energy used in production of fertilizers, pesticides and herbicides, and fuels and electricity used in harvesting and grain transportation. All of these important contributions appear to be included in the Vink PLA LCA (Vink 2007).

	Primary energy (MJ/kg)	GWP (kg CO2eq./kg)	Source	Location
PLA		1.8	Vink 2003	US
PLA		0.97	Vink 2007	US
PLA		3.2	Modified Vink 2003	US

Table 3: LCA results for PLA

A speculative worst case scenario, in which 10% of the PLA is methylated in a landfill environment, is included given the complete lack of data on PLA behavior in landfills. PLA is composed of a repeating structure of CH_3O_2, as shown in Figure 1. Fully methylated, 1 kg of PLA would produce 0.34 kg methane. Thus, if 10% of a given mass of PLA is converted to methane, 0.034 kg methane is emitted, resulting in 0.71 kg CO_2eq GWP impact per kg PLA.
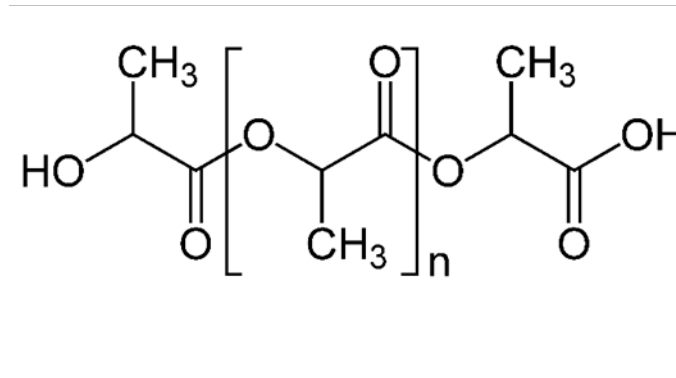

Figure 1: Chemical structure of polylactide

Some benefits popularly ascribed to PLA as a material for electronics housings may not be actualized in all cases. For example, in a press statement from Fujitsu and in media articles about the BIBLIO laptop, it is stated that PLA is preferable to traditional fossil fuel plastics because at end of life if its incineration will produce no toxic environmental emissions (Fujitsu 2008, Chapa 2008). This is a speculative statement, particularly because PLA is alloyed with other plastics or impregnated with additives such as fibers or flame retardants, and depends on the conditions of incineration. In practice, PLA could produce carbon monoxide or other toxic emissions if combusted in an improper incineration system, just as other plastics may. Also, while articles in the media have described the Fujitsu BIBLIO laptop housing as "biodegradable," (Kanellos 2007) Fujitsu communications do not claim that the laptop housing is biodegradable, and rather state only that the material has a carbon footprint 14% or 15% lower than it would if composed of standard plastics (Kimura 2005, Fujitsu 2008). Benefit of the use of PLA as a vessel of carbon sequestration depends on the material's end-of-life (EOL) treatment, as will be discussed in a following section.

Importance of end-of-life treatment for bio-based materials

End-of-life treatment can make a critical difference in the impacts associated with PLA. The Vink PLA LCA studies assume that the PLA will be recycled at end of life. Currently, widespread recycling facilities

for PLA, particularly for PLA blends (as the material would most likely be used in electronics housings), do not exist.

By assuming a recycling case at end-of-life, these studies sidestep an important potential contributor to GWP impacts: methane released during bio-degradation in landfill. At the stage of this study, it is not clear whether PLA will likely hydrolyze to a biodegradable molecular weight in landfill, and whether a larger fraction would consequently methylate than in the case of fossil-fuel plastics, which also have the potential for the same behavior.

Option 1: Landfilling

If the PLA is land-filled, the material will most likely resist biodegradation and behave similarly to other plastics, and this possibility is supported by some preliminary research (Pullammanappallil 2003) in which PLA cups were observed to resist biodegradation over 12 months in composting facilities. However, conflicting results have been produced in a study of PLA in municipal yard waste composting sites in the US (Greene, 2007). The subject of both of these studies is the behavior of PLA in compost facilities, rather than in a landfill, however. There is a possibility that PLA will eventually biodegrade in the anaerobic environment of a landfill and be converted to methane, though this scenario is only theoretical. Based on the chemical composition of PLA, assuming 100% methylation, 1 kg PLA could, in this worst case scenario, produce 7 kg CO_2eq in methane emissions. The timescale of biodegradation or methylation of PLA is unknown and may require anywhere from 10 to 10,000 years. Basic research on the chemical behavior of PLA in landfill is needed to address this area of gross uncertainty.

Option 2: Composting

Composting systems produce a waste stream which is sold or passed on to end users for a variety of applications (e.g., agriculture, horticulture, silviculture, erosion control) as a soil amendment (EPA 1994). While some local waste management systems are exploring the acceptance and processing of PLA food containers and plastic bags within the organic waste stream, and pure PLA has been observed to biodegrade or disintegrate sufficiently in some composting facilities (Greene 2007, Kale 2006), PLA-based computer housings have a variety of special concerns which differentiate them from pure PLA-based products such as food containers. The only laptop housing containing PLA thus far has been a PLA/PC mixture. Composting is not a realistic end of life scenario for computer housings composed of a PLA blend as these mixtures cannot be expected to biodegrade in the same way that pure PLA would. Additives for color, anti-flammability and other properties also may impede biodegradation, or make the material inappropriate for inclusion in the composting waste stream due to safety requirements or reduced value as a soil amendment.

Even in cases in which the bio-plastic were biodegradable after a pre-treatment, a composting infrastructure is not yet widely available for PLA or other bio-plastics from consumer waste, despite a considerable supply of PLA waste from food packaging. There are 113 industrial composting facilities nationwide which are running a process sufficient to biodegrade Natureworks PLA (140°F for 10 days). In the locations where composting of organic waste is available, the laptop could be dismantled and shredded in a traditional electronic waste facility, and any pure PLA waste sent to compost, but without financial incentive to drive special handling of the material stream, as with recyclable plastics, composting of the PLA from e-waste seem less likely.

Options 3 and 4: Energy recovery through combustion or anaerobic digestion

Other waste treatment options besides landfilling and composting can yield energy, and thus provide a financial incentive, making PLA recovery from e-waste more likely. Energy may be recovered from PLA through combustion or anaerobic digestion to valuable byproducts such as methane. In both of these cases, the material could be considered to have additional environmental benefit, in that the energy generated at end-of-life would offset impacts associated with other energy sources. The environmental benefit of this energy would depend on the efficiency of conversion to electricity or fuel, as well as the emissions factor for electricity in the location where generation occurs.

Energy recovery through combustion is a realistic EOL option for e-waste PLA. Although most housing plastics contain flame retardants, combustion is achieved for plastics containing typical[2] flame retardants (APME 1997, OECD 1998, EC 1995), at operating temperatures required in European energy recovery plants. Although empirical research indicates that the presence of brominated flame retardants does not correlate with dioxin emissions from municipal energy recovery systems running at high enough temperature (Vehlow 2001), laboratory tests have shown dioxin formation from combustion of brominated flame retardants at lower temperatures (OECD 1998). Formation of brominated dioxins and furans during combustion remains a concern, whether or not plastic is conducted towards energy recovery, due to the use of open-pit burning of plastics (and smelting of PWB) in informal or illegal recycling situations (BAN 2002), as high concentrations of dioxins have been found at locations of informal recycling (Wong 2007).

Anaerobic digestion, similarly, may be possible, but would require separation within the e-waste recycling system and is thus subject to the same hurdles as described for PLA composting. Anaerobic digesters typically accept liquid polymers such as starch and cellulose (CalRecycle 2008), not solid particles of plastic. There has been some research conducted on the topic of anaerobic digestion of PLA at the National Institute of Advanced Industrial Science and Technology in Japan. Preliminary studies indicate that after pulverization, PLA in powder form will biodegrade 90% at 55°C over 70 days (Yagi 2009). Over this period, 10 g PLA yielded 0.79 liters of methane and about 0.4 liters CO_2 in one test. Anaerobic digestion of separated and shredded PLA seems promising, as valorization of the plastic can

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[2] polybrominated diphenyl ether (PBDE) and polybrominated biphenyl (PBB)

drive separation from e-waste and because inputs to these digesters are less restricted than to composting systems. The technical and cost feasibility of this option is a valuable topic for further research.

Option 5: Recycling

No studies concerning the recyclability of PLA-containing plastic blends, or PLA materials from e-waste sources have been identified in the literature at this time. A report on the potential for negative impact of biopolymers on the existing commercial recycling systems for beverage containers indicates that biopolymers are unlikely to contaminate HDPE recyclate but more likely to foul PET materials streams, due to the separation processes used for each plastic (Cornell 2007), and the relative inseparability of PLA from PET. Given that PLA will most likely be used in a blend with other plastics, as well as additives and, possibly, fibers, the potential for recycling the material back to a virgin-equivalent application is lower than separation and recycling of existing electronics housing blends, which has been proven to be technically feasible but is not commonly occurring due to either to cost or to the momentum of the existing market.

Feedstock recycling, the cracking of a plastic blend into its component plastics and additives, may also be possible as a potential approach, depending on the ability to separate the PLA/polymer blend from additives through physical or chemical means.

ABS and PC, like PLA, have a density greater than water and are hygroscopic. Due to these common properties, PLA would be difficult to separate from ABS/PC blends using common recycling separation methods: density-based separation by water, or olefin separation. Depending on how the PLA is blended, newer PLA-containing e-waste plastics may not need to be separated from non-PLA containing e-waste, however, if the second generation material properties of PLA blends are similar enough to existing blends. The potential for PLA blend recyclate for downcycling applications (road fill, etc.) is good, given that these materials will be designed to meet the mechanical standards required for their application as an electronics housing plastic. Recycling of PLA-containing blends back into housing materials will face the same technical and policy challenges that hinder recycling of traditional electronics plastics back into virgin applications. More research is required to understand the technical limitations of and policy barriers to recycling plastic material streams from e-waste, as well as the chemical compatibility of PLA in potential mechanical performance of recycled PLA blends in virgin-equivalent applications.

LCA comparison of PLA with PC/ABS blends

Ideally the functional unit used for comparison would be the plastic content of a single laptop. However, because the material properties of PC/PLA are not known to be precisely equivalent to PC/ABS, different types and quantities of additives would likely be used for each of the two plastic blends. The quantity of PC/PLA may be higher, if a thicker wall and larger amount of material is used to achieve the same strength and durability, or the mass may be smaller, if a larger fraction of additives is used in a PC/PLA blend to achieve the same characteristics within the same volume as a PC/PLA housing. The composition of additives used in existing laptop housings is not disclosed in most cases; for example, those used in the Fujitsu BIBLIO laptop are described only as a flame retardant (Fujitsu 2005). Particularly in the case of PLA, additive formulations are continuously changing (Grande 2007). Thus, including the life-cycle inventory contributions of a hypothetical combination of additives for each material scenario is not yet possible and is not considered within this report. In order to make a fair comparison between realistic PC/ABS and PC/PLA blends, on the basis of a laptop housing mass, the additives must be known and included in the analysis. For the purposes of this report, the scope of the LCA comparison is limited to the polymer, before addition of flame retardants, coloring, strengthening fibers or other agents.

Because the content of potential PLA blends is unknown at this time, the life-cycle results for pure PLA are presented for comparison with common PC/ABS blends with the understanding that PLA would most likely not be used in a pure form in computer housings. Four blends of PLA and PC are also included for comparison. These hypothetical blends are based on preliminary technical performance data from unpublished industry reports (Natureworks 2007), and include the blend used by Fujitsu in the BIBILO laptop, 50% PC/50% PLA (Fujitsu 2005). GWP impacts for pure materials as well as these blends are presented in Table 4.

GWP (kgCO$_2$eq/kg)	Baseline	Low	High
ABS	3.8	3.1	3.8
Polycarbonate	7.6	5.0	7.6
20%PC / 80% ABS	4.5	3.5	4.5
30%PC/70%ABS	4.9	3.7	4.9
PLA	2.0	1.8	2.7
50% PLA / 50% ABS	2.9	2.5	3.2
50% PLA / 50% PC	4.8	3.4	5.2
20% PC/ 80% PLA	3.0	2.4	3.7
30% PC/70% PLA	3.7	2.8	4.2

Table 4: GWP impacts for traditional housing plastics and PLA blends, per kg

To compare the impacts of the housing with other components and parts of a laptop, and to facilitate a future, more detailed study of the GWP impacts of each housing, the total range of volumes for each size category of laptop is given in the first column of Table 5. An average of these volumes, combined with an assumption for the volume fraction of plastic in the laptop and the density of the most common housing plastic blend (75% PC/25% ABS) of 0.039 lbs/in^3, results in an expected mass of plastic for each size category. The average among all categories is then taken, resulting in roughly 0.85 kg of plastic per laptop. This value compares well with a recent disassembly study which found a laptop chassis to have a mass of 780 g (Deng 2010).

	Average volume in^3	% plastic by vol.	Mass of plastic lb	Mass of plastic kg
Netbook/Subnotebook: <10.5" x <9.5" x <1.3" @ <3.5 lbs	117	25%	1.14	0.52
Ultraportable: 9-11" x 8-10" x <1.5" @ 2.0-5 lbs.	135	25%	1.31	0.60
Thin and light: 11-15" x <11" x 1-1.7" @ 5-7 lbs.	231	25%	2.25	1.0
Desktop Replacement: >15" x >11" x >1.5" @ >7 lbs.	248	20%	1.93	0.87
Luggables: >18" x >13" x >1.5" @ >12 lbs.	351	20%	2.73	1.2

Table 5: Mass per unit of typical laptop sizes

Because information concerning the expected mechanical properties is not available yet, differences in the required volume of material for each blend cannot yet be considered in this analysis. Impacts scale with mass, as shown in Table 6: GWP impact results for traditional housing plastics and PLA, per average laptop.

GWP (kgCO$_2$eq/laptop)	Baseline	Low	High
ABS	3.2	2.6	3.2
Polycarbonate	6.5	4.3	6.5
20%PC / 80% ABS	3.8	3.0	3.8
30%PC/70%ABS	4.2	3.1	4.2
PLA	1.7	1.53	2.3
50% PLA / 50% PC	2.5	2.1	2.8
50% PLA / 50% ABS	4.1	2.9	4.4
20% PC/ 80% PLA	2.5	2.1	3.2
30% PC/70% PLA	3.1	2.3	3.6

Table 6: GWP impact results for traditional housing plastics and PLA, per average laptop

Summary of Research Priorities

In order to allow a definitive comparative LCA of traditional plastic blends with blends of bio-based materials, data and further study are needed in several areas.

- *Additive mixtures and fractions required for bio-based housing blends.* Because the GWP benefits of PLA in the production stage over fossil-based plastics are not overwhelming, changes in the percentage fraction of additives can determine the relative benefit of bio-based blends over traditional plastics.

- *Life cycle inventory data for flame retardants and mechanical additives.* Without this data, the scope of comparison must be limited to the polymer alone. Because additives contribute a significant proportion of mass of the housing plastic (>5%), and vary depending on the blend of polymers, a better understanding of the impacts associated with production of additives (for color, strength, and flame resistance) would allow a more detailed and definitive comparison.

- *Wall thickness or mass requirements for a bio-based blend to meet equivalent mechanical performance of existing housing blends.* This information would allow comparison by functional unit of the polymer fraction of a laptop, in the absence of life-cycle inventory data for additives.

- *Extended studies of chemical behavior of PLA in a landfill* The likelihood and rate of PLA conversion to methane in the chemical conditions in a landfill can make or break the life-cycle GWP benefits of PLA.

- *Potential for PLA blend recycling*

  - *More research into the market limitations and policy barriers to recycling existing, fossil fuel-based plastic material streams from e-waste*

  - *Understanding of chemical compatibility of newly developed electronics blends containing bio-based polymers with existing e-waste plastics material streams, and evaluation of potential for fouling due to introduction of these blends in recycling or down-cycling e-waste plastics*

  - *Mechanical performance of recycled PLA blends in virgin-equivalent applications*

- *Technical, safety and cost feasibility of energy recovery through combustion*

- *Technical and cost feasibility of energy recovery through methane digestion*

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