6. BIODEGRADABILITY OF PACKAGING WASTE

6.1 Literature review

6.1.1 Introduction

Over 67 million t of packaging waste are generated annually in the EU, comprising about one third of total MSW (Klingbeil, 2000). In the UK, 5.1 million t, or approximately 20 %, of total annual household waste arisings was packaging (DEFRA, 2002; Wasteonline, 2002). Due to strict food packaging regulations and also the drive to enhance the appearance and increase product sales, food packaging is a major component of packaging waste, representing 60% of all packaging produced in developed countries (Northwood and Oakley-Hill, 1999). When food packaging enters the municipal waste stream, it becomes a major source of household refuse which is disposed to landfill sites. The European Directive on Packaging and Packaging Waste (94/62/EEC) and the Packaging (Essential Requirements) Regulations (2003) have been introduced and adopted in the UK to discourage the formation of packaging waste, promote reuse and recycling of packaging materials.

Paper and cardboard or pulp-based packaging including, for example, wrapping paper, carton boxes, disposable plates and cups and corrugate cardboard, make up 6.4 % by weight of the residual waste bin (Wasteonline, 2006). Currently, a large proportion of the packaging waste is recycled within industry, particularly since the introduction of the UK Packaging Waste Regulations. In 2001, obligated UK industries had a packaging recycling rate of 42 % and the minimum material specific target of 15 % set within the regulations was exceeded for all materials (IEMA, 2003). However, when recycling or re-processing of these materials is not environmentally sound, composting of suitable packaging waste may be a viable alternative. Packaging waste can be collected directly from households with kitchen and garden waste and composted in municipal composting facilities, or could potentially be composted in home compost bins, provided that they are compostable under such conditions.

Whilst significant improvement has been achieved in recycling pulp-based packaging materials, little success has been achieved in reducing plastic packaging waste collected for landfill disposal. The difficulties associated with collection, identification, sorting, transportation, cleaning and re-processing of plastic packaging materials render the recycling process for these materials uneconomic and landfilling is often the main disposal method. Only 23 % of packaging plastic waste was recycled in the UK in 2001 (Wasteonline, 2002). Plastic materials for packaging have increased significantly in the last two decades and over the last 50 years synthetic polymers have been replacing more traditional materials in packaging applications because of their low cost, low density, resistance to corrosion, desirable physical and mechanical properties and ease of processing (Davis, 2006). Plastics that are used in packaging can be thermoplastic or thermosetting and are made almost entirely from chemicals derived from crude oil (McCarthy, 1993). They vary in composition and characteristics, are mixed with additives such as fillers, plasticizers, colorants and antioxidants to improve the polymer's physical or chemical properties, and usually are coated, printed or laminated with other polymers. Material additions and different processing procedures add to the variation of plastic materials, which poses a complication in sorting the materials for recycling purposes. Due to the high volume and low weight to volume ratio of plastics, the collection of these materials in recycling kerbside schemes is not an economic option, and therefore, many Local Authorities in the UK are not willing to include them in their recycling scheme. Moreover, plastic packaging materials are often soiled with food leftovers or other organic substances, making recycling of these materials impractical and problematic. Low recycling rates and the high volume of non-degradable plastics have

shortened dramatically the life expectancy of current commercial landfills (McCarthy, 1993) and increased the demand of biodegradable plastic packaging materials.

6.1.2 Paper and cardboard packaging biodegradation

Although considerable research has focused on the development and biodegradability testing of biodegradable plastics (Gu *et al*., 1994; Ho *et al*., 1999; Ohtaki, 2000; Grima *et al*., 2001; Tokiwa and Jarerat, 2004; Kaluss and Bidlingmaier, 2004), relatively little attention has been given to assessing the biodegradation and compostability of paper-based packaging materials. Paper is made up of lignocellulose, which consists of three types of polymers: cellulose, hemicellulose and lignin. The latter is a persistent macromolecule, which protects cellulose and hemicellulose against microbial attack by hydrolytic enzymes. Due to the protective lignin barrier, lignocellulose disintegration is an intricate, complex process, which requires the synergistic action of several enzymes, such as laccases and peroxidases, in addition to cellulases and hemicellulases (Hatakka, 2001). To understand paper and cardboard packaging biodegradation in a compost environment, information about lignocellulose and in particular lignin degradation is important because paper may contain up to 20% of lignin (Biermann, 1993). Only few groups of microorganisms are capable of attacking the complex lignin molecule, among which the white-rot fungi (basidiomycetes) are the most efficient degraders, causing a substantial mineralisation of lignin (Hatakka, 2001; Tuomela *et al*., 2000). White-rot fungi degrade lignin by means of oxidative enzymes (Hatakka, 1994). Because of the nature and size of the lignin molecule, the enzymes responsible for the initial attack must be extracellular and nonspecific (Kirk and Farrell, 1987; Hattaka 1994). The best studied extracellular enzymes of white-rot fungi are lignin peroxidases (LiPs), manganese peroxidases (MnPs) and laccase. Brown-rot fungi extensively degrade cellulose and hemicellulose molecules, but lignin degradation is limited. Lignin is chemically modified by demethylation of its phenolic and nonphenolic units (Kerk and Farrell, 1987; Eriksson et al., 1990), and limited aromatic hydroxylation and ring cleavage of lignin also occurs (Kerk and Farrell, 1987). Brown-rot fungi are able to mineralise the methoxyl of lignin, but the mineralization of other parts is much lower (Buswell and Odier, 1987; Kirk and Farrell, 1987). Soft-rot fungi, Ascomycotina or Deuteromycotina, degrade lignin in both hardwood and softwood, but hardwoods are degraded to a greater extent than softwoods (Kuhad et al., 1997). Although all wood parts are degraded, the rate of degradation is minimal compared to that of white-rot or brown-rot fungi (Eriksson *et al*., 1990). Soft-rot fungi can degrade wood in conditions that are unsuitable for white- or brown-rot fungi, for example, in wet environments (Blanchette, 1995). Little is known about the enzyme system of soft-rot fungi or their lignin degradation capacity as litter decomposing organisms (Haider and Trojanowski, 1980; Kirk and Farrell, 1987). There are also many genera of actinomycetes and eubacteria which can degrade extracted lignin (Buswell and Odier, 1987). Many bacterial strains, especially actinomycetes, can solubilise and modify the lignin structure, but their ability to mineralise lignin is limited (Buswell and Odier, 1987; Ball et al., 1989; Eriksson et al., 1990; Godden et al., 1992). Actinomycetes degrade lignin as their primary metabolic activity and at higher nitrogen concentrations compared to white-rot fungi, most of which degrade lignin via secondary metabolism. The lignin-degrading eubacteria can be divided into erosion, cavitation and tunnelling bacteria (Eriksson *et al*., 1990; Blanchette, 1995). Wood is degraded by bacteria under certain extreme environmental conditions, e.g. wood saturated with water, anaerobic conditions or wood with a high extractive content. However, the rate of degradation is very slow (Eriksson *et al*., 1990; Blanchette, 1995).

Table 6.1 includes a number of thermophilic fungi that occur in composts and have a lignocellulose-degrading capability. The occurrence of fungi has been studied in large-scale composting (von Klopotek, 1962; Thambirajah and Kuthubutheen, 1989; Nusbaumer *et al*., 1996; Thambirajah *et al*., 1995). Waksman *et al*. (1939 a, b) studied the microbial dynamics of composting processes on a laboratory scale at temperatures of 28 °C, 50 °C, 65 °C and 75 °C. At 28 °C the population was heterogeneous with bacteria being dominant throughout

Table 6.1 Thermophilic fungi occurring in composts with a lignocellulose-degrading capability

Fungus	Subdivision	Rot type	T_{opt} (°C)	T_{max} (°C)	Lignocellulose degradation	C/MC[a]	Reference[b]
Aspergillus fumigatus	Deuteromycotina		35-43	52-55	Wood degradation, cellulose degradation, found in piles of hay and bark	C	3,5,7,8,9,10,11,16,19,21,22,23,24
Chaetomium thermophilum Var. *coprophile* Var. *dissitum*	Ascomycotina		40-52.5	54-61	Very active cellulose degradation, found in piles of wood chips	C, CM	3,5,6,7,8,18,19,21,24
Coprinus sp. *Coprinus cinereus* *Coprinus delicatulus*	Basidiomycotina	Brown-rot	45 35	55 45	Lignin modification	C, CM	3,6,15,25 4,7,21,22 6
Ganoderma colossum	Basidiomycotina	White-rot	40	>45	Effective lignin degradation		1,2
Malbranchea cinnamomea (=*Malbranchea sulfureum*) (=*Malbranchea pulchella*) (=*Thermoideum sulfureum*)	Deuteromycotina		45-47.5	53-57	Cellulose degradation, found in piles of hay	C, CM	3,4,5,6,14,19,20,22,24
Melanocarpus albomyces (=*Myriococcum albomyces*)	Ascomycotina		37-50	55-57	Cellulose and hemicellulose degradation	C, CM	3,4,7,11,16,17
Myceliophthora thermophila (=*Sporotrichum thermophile*) (teleomorph: *Corynascus heterothallicus*)	Ascomycotina		36-50	52-65	Very active cellulose degradation, wood degradation, found in piles of wood chips		3,5,6,8,9,16,18,19
Paecilomyces spp. *Paecilomyces varioti*	Deuteromycotina	Soft-rot	45-50	55-60	Cellulose and some lignin degradation		3,6,7,9,22,24,

Table 6.1 (continued)

Fungus	Subdivision	Rot type	T_{opt} (°C)	T_{max} (°C)	Lignocellulose degradation	C/MC[a]	Reference[b]
Phanerochaete chrysosporium (=*Sporotrichum pulverulentum*) (=*Chrysosporium pruinosum*)	Basidiomycotina	White-rot	36-45	46-55	Effective lignin degradation, newspaper degradation, found in piles of wood chips	C, CM	3,16,17,19,20
Scytalidium thermophilum (=*Torula thermophila*) (=*Humicola grisea*) (=*Humicola insolens*)	Deuteromycotina		35-47.5	55-58	Cellulose degradation	C, CM	3,5,6,9,14,22,24
Stibella thermophila	Deuteromycotina		35-50	55	Cellulose degradation	MC	3,6,10,14,19
Talaromyces emersonii (anamorph: *Penicillium emersonii*)	Ascomycotina		45-50	55-60	Wood degradation, found in piles of wood chips	C, CM	3,6,16,19,22
Talaromyces thermophilus (=*Talaromyces dupontii*) (anamorph: *Penicillum dupontii*)	Ascomycotina		45-50	57-60	Weakly ligninolytic, found in piles of wood chips	C, CM	3,5,6,7,9,10,14,22,24
Thermoascus aurantiacus	Ascomycotina	Soft-rot	45-52.5	55-62	Effective lignin degradation, found in piles of wood chips	C, CM	3,5,6,12,14,16,18,19,21,22,24
Thermomyces lanuginosus (=*Humicola lanuginose*)	Deuteromycotina		45-55	60	Found in piles of hay and wood chips, cellulose degradation (not for all isolates), some lignin degradation	C, CM	3,5,6,7,8,10,11,19,21,22,24,25
Thermomyces ibadensis	Deuteromycotina		42-47	60-61	Palm kernel degradation		3,6,14
Thielavia terrestris (=*Allescheria terrestris*)	Ascomycotina	Soft-rot	40-47.5	55	Wood and some lignin degradation		3,6,7,16,19,22

Notes: [a]C = found in compost, MC = found in mushroom compost
[b]References: (1) Adaskaveg *et al*. (1990); (2) Adaskaveg *et al.* (1995); (3) Brock (1978); (4) Yung-Chang (1967); (5) Cooney and Emerson (1964); (6) Crisan (1973); (7) Dix and Webster (1995); (8) Eggins and Malik (1969); (9) El-Naghy *et al.* (1991); (10) Fergus (1964); (11) Kane and Mullins (1973); (12) Machuca and Duran (1996); (13) Maheshwari and Kamalam (1985); (14) Mouchacca (1997); (15) Nusbaumer *et al.* (1966); (16) Ofosu-Asiedu and Smith (1973); (17) Rayner and Boddy (1988); (18) Romanelli *et al.* (1975); (19) Rosenberg (1975); (20) Rosenberg (1978); (21) Sharma (1989); (22) Straatsma *et al*. (1994); (23) Stutzenberger *et al.* (1970); (24) von Klopotek (1962), and (25) Walsman *et al.* (1939a).

the whole period, and fungi appearing later. Fungi, together with bacteria and actinomycetes, were represented in the microbial population in compost at 50°C. Active thermophilic fungi occurred initially followed by bacteria and actinomycetes, which also grew on the fungal mycelium. Compost contains about 10^6 microbial counts of mesophilic fungi g^{-1} and 10^3-10^6 g^{-1} of thermophilic fungi (von Klopotek, 1962; Thambirajah and Kuthubutheen, 1989; Thambirajah et al., 1995). The most abundant mesophilic fungus in composted residues are *Geotrichum* sp. (von Klopotek, 1962; Nusbaumer *et al*., 1996) and *Aspergillus fumigatus* is the main thermotolerant type (von Klopotek, 1962). Counts of fungi decrease as the temperature rises, and at 64 °C the thermophilic fungi disappear. However, a mesophilic fungus, *Cladosporium cladosporioides*, was able to grow at 64-65 °C, but no fungi were detected at 67 °C (von Klopotek, 1962). In the studies of Thambirajah *et al*. (1995) and Waksman *et al*. (1939 a, b), no fungi were detected in compost when the temperature exceeded 60 °C. In the study of Thambirajah and Kuthubutheen (1989), fungi survived at high temperatures probably due to the short duration of the exposure. When the temperature decreases below 60 °C, both mesophilic and thermophilic fungi reappear in compost (von Klopotek, 1962; Thambirajah *et al*., 1995). The dominating fungus after peak heating is *Aspergillus* sp. (Nusbaumer *et al*., 1996) or *Thermomyces lanuginosus* (von Klopotek, 1962), which was also found to be dominant at 50 °C. *T. lanuginosus* can decompose cellulose, hemicelluloses and even lignin, although to a much smaller extent than the other components (Waksman *et al*., 1939 a, b). In the studies of Thambirajah and Kuthubutheen (1989) and Thambirajah *et al.* (1995) the number of mesophilic and thermophilic fungi (10^4-10^6 g^{-1}) in mature compost were similar, but in the study of von Klopotek (1962) thermophilic fungi were the dominant type, especially in drier parts of mature compost. *Coprinus* sp. (von Klopotek, 1962; Nussbaumer *et al*., 1996), *Panaeolus* sp., *Corticium coronilla* and possibly *Mycena* sp. (von Klopotek, 1962) are Basidiomycotina that also occur in compost. These organisms were all isolated from compost during the cooling and maturation phase or from mature compost (von Klopotek, 1962; Nussbaumer *et al*., 1996) and are effective lignin degraders.

A wide range of bacteria have been isolated from different compost environments, including species of *Pseudomonas*, *Klebsiella* and *Bacillus* (Nakasaki *et al*., 1985; Strom, 1985 a, b; Falcon *et al*., 1987). Several species of the species of *Bacillus* are present in the thermophilic phase including, e.g. *B. subtilis*, *B. licheniformis* and *B. circulans*. Strom (1985b) reported that as much as 87 % of randomly selected colonies of bacteria isolated during the thermophilic phase of composting belong to the genus *Bacillus*. Many thermophilic species of *Thermus* have been isolated from compost at temperatures as high as 65 °C and even 82 °C (Beffa *et al*., 1996). Actinomycetes are bacteria which form multicellular filaments, and thus resemble fungi. They appear during the thermophilic phase as well as the cooling and maturation phase of composting, and can occasionally become so numerous that they are visible on the surface of the compost. Thermophilic actinomycetes have been isolated from a wide range of organic substrates such as compost (Cross, 1968). The genera of the thermophilic actinomycetes isolated from compost include: *Nocardia*, *Streptomyces*, *Thermoactinomyces* and *Micromonospora* (Waksman *et al*., 1939b; Strom, 1985a). Actinomycetes are able to degrade some cellulose, and solubilise lignin, and tolerate higher temperatures and pH than fungi. Thus, actinomycetes are important agents of lignocellulose degradation during peak heating, although their ability to degrade cellulose and lignin is not as high as that of fungi (Crawford, 1983; Godden *et al*., 1992).

Several published studies of lignin degradation during composting were identified in the literature. Stutzenberger *et al*. (1970) studied the composting of MSW containing paper products for 49 days. The cellulose content of the waste was 46-56 %. Lignin may have inhibited the degradation of cellulose since 40 % of the cellulose remained undegraded. (Stutzenberger *et al*., 1970). Horwath and Elliott (1996) composted ryegrass for 45 days at

temperature conditions of 25 or 50 °C and found the proportion of lignin that was degraded under these conditions was 7 % and 27 %, respectively. At both temperatures the elemental ratio of the lignin content changed, and it was estimated that only 6 % of the residual lignin was unaltered after composting (Horwath and Elliott, 1996). Waksman *et al*. (1939b) conducted an earlier study on the degradation of lignin in composts at different temperatures. The highest rate of lignin degradation occurred at 50 °C, whereas lower rates of decomposition were measured at 28 °C and 65 °C. There was no biodegradation of lignin detected at 75 °C, but 12 % of the lignin was solubilised as a result of the high temperature and alkaline reaction of the compost (Waksman *et al*., 1939b). In experiments where the thermophilic phase lasted for a short time, between 7 and 14 days, lignin was not degraded, even when the total composting time was for an extended period of 295 days (Nusbaumer *et al*., 1996). According to Tomati *et al*. (1995), 70 % of lignin was degraded during 35 days when the temperature of the compost was maintained at 50 °C, whereas only negligible degradation occurred later during the maturation phase. Therefore, on the basis of this evidence, the duration of the thermophilic phase is an important factor in lignin degradation during waste composting. Waksman *et al*. (1939a) examined the degradation capacity of some microorganisms isolated from compost. Two thermophilic actinomycete isolates degraded 0.7-2.5 % of the lignin in 42 days at 50 °C, and 4.2 % was decomposed by the thermophilic fungus *Thermomyces lanuginosus*, but neither of the two bacteria studied were capable of lignin degradation. However, the natural microbial population in manure degraded 11.5 % of the lignin content. The natural population also decomposed 62 % of the total dry material compared to 40% for the fungus alone (Waksman *et al*., 1939a), suggesting consortia of microorganisms working together are more effective at degrading recalcitrant materials like lignin than individual species alone. Finally, Kluczek *et al.* (2003) isolated two strains of the Deuteromycete, *Paecilomyces infatus*, from compost samples consisting of municipal wastes, paper and wood chips, to study the degradation of synthetic $^{14}C_{\beta}$-labelled lignin (side-chain labelled dehydrogenation polymer, DHP). Approximately 6.5 % of the synthetic lignin was mineralised during solid-state cultivation of the fungus in autoclaved compost, and 15.5 % was converted into water-soluble fragments.

In general, the literature survey showed that very little is known about the degradation of lignin, and therefore, paper packaging products, in composting systems. Although lignin can be degraded or transformed extensively during composting, the extent of total mineralization losses may be relatively limited in practice. The organisms most efficient at mineralising lignin, white-rot fungi, do not survive the thermophilic phase of composting, and thus do not play a significant role in lignin degradation. These and other basidiomycetous fungi, that have been found in composts, appear during the cooling and maturation phase or in mature compost (von Klopotek, 1962; Nusbaumer *et al*., 1996). Thermophilic fungi are probably the most important lignin degraders (Waksman *et al*., 1939 a, b) and synergistic effects with other organisms may enhance the degradation significantly (Waksman *et al*., 1939a). Lignin degradation in composts is regulated by temperature, the original lignin content and the particle size of ligninaceous materials.

6.1.3 Coated paper and cardboard

Most of the paper and cardboard used in packaging manufacture are impregnated with wax compositions, including paraffin, microcrystalline and polyethylene (PE), to impart mechanical strength and relative impermeability to moisture (Asadchii *et al*., 1986). Petroleum wax-based or low-density polyethylene (LDPE) coatings are typically used for moisture resistance, while fluorocarbon-derived treatments are used for grease resistance, either alone or in combination with the other coatings. When petroleum wax-based coatings are used for packaging of hot foods they are typically modified with high-melting temperature synthetic waxes to provide the required high-temperature properties. Wax and polyethylene are also used as laminates in packaging constructions containing more than one paper layer. Biodegradable and compostable plastic films derived from starch are used

for paper lamination and may have potential use in packaging materials (Bastioli, 1997). An example is the Mater-Bi ZF03U/A and ZF02U/A biodegradable and compostable films, which are made of thermoplastic starch and poly-ε-caprolactone. Vikman *et al.* (1995) tested the compostability of these Mater-Bi biodegradable films in two full-scale composting experiments for 49 (experiment 1) and 70 (experiment 2) days. Both experiments were carried out in an insulated commercial compost bin filled with biodegradable waste consisting mainly of vegetable and fruit waste. A mixture of bark and wood chips was also added to maintain aerobic conditions in the waste. Film specimens (2.5 cm x 3.5 cm were attached to a steel frame and buried in the waste. Cellulose-based sausage casing was used as positive control (compostable) and low density polyethylene (LDPE) as a negative control (non-compostable). The cellulose-based material degraded completely in both experiments and, as expected, there was no degradation of the LDPE film. Total weight loss of the Mater-Bi films was in the range 40-45 % and the materials became brittle after 70 days. According to the manufacturer of Mater-Bi products (Novamont, Novara, Italy) the ZF03U/A and ZF02U/A films degrade after 20-45 days in a composting environment (Bastioli, 1997).

Davie *et al.* (1994) examined the composting of paper packaging waste from a fast food restaurant in poultry manure for a period of 56 days. The decomposition of uncoated and heavily waxed cupstock, waxed burger wrap, and double-sided polyethylene coated paper was monitored by isolating 5 cm^2 paper samples in 0.2 mm mesh screens and immersing the screens in the compost. The average dry matter loss due to decomposition was 66 % for all the materials tested. The uncoated cupstock exhibited the largest dry weight loss (81 %), followed by waxed cupstock (79 %), waxed burger wrap (74 %), and double-sided polyethylene paper (31 %). After the removal of paper samples, the remaining compost was mixed with soil in pots at up to 5 % by weight and allowed to incubate for varying periods. The pots were seeded with ryegrass and the yield response was determined. Plant growth was inhibited at the 5 % rate of compost addition but yield increased compared to the control for soil blended with 2.5 % compost. The chemical analysis of the compost showed that samples contained less that 1 % wax residuals and heavy metal concentrations were small.

6.1.4 Biodegradable plastic packaging

Biodegradable plastic packaging materials are broadly classified into biodegradable polymers and biopolymers. Biodegradable polymers are synthetic oil-based polymers that have certain degrees of inherent biodegradability such as polycaprolactone, polyhydroxybutyrate and poly (vinyl alcohol) (Brody and Marsh, 1997) or are chemically modified to assist biodegradation (Bastioli *et al*., 1994). Biopolymers are naturally occurring polymers such as cellulose, polysaccharides and proteins, but the definition has been extended to describe materials made or derived from these natural polymers. Unlike synthetic polymers, most biopolymers are biodegradable, and hence, they can be decomposed by fungal or bacterial activity into natural metabolic products. Most commercially available biodegradable plastic packaging materials are based on natural materials, e.g. polysaccharides (starch). This is because starch is a renewable, abundant and inexpensive material (Lockes, 1998; Petersen *et al*., 1999; Davies, 2006). When used alone in packaging applications, starch exhibits a poor performance because of its brittleness and hydrophilic nature. To overcome these problems, starch is often modified mechanically, physically or chemically, and/or combined with a plasticizer or polymeric additives. Where starch is blended with biodegradable polymers or copolymers, the concentration of starch in the mixture is used to classify the material. Thus, a ‘starch containing biodegradable polymer’ contains <50 % of starch by weight and a ‘starch-based biopolymer’ contains >50 % of starch by weight. The concentrations of starch in degradable polymer mixtures may vary from 5 to 90 % by weight (Davis, 2006).

The design of biodegradable polymers must ensure its functionality during use, but also its destruction in response to an environmental trigger (such as temperature, light, hydration or microbial) after use. Alternatively, degradation may be triggered by additives that catalyse the breakdown of the polymer chains under specific environmental conditions (Narayan 2001). Polymers must remain stable during manufacture and use but breakdown rapidly when discarded into landfills (Scott 2000). Degradation can be monitored using physical changes or chemical changes, for example, by the observation of new functional groups in Fourier transform infrared spectroscopy (FTIR) spectra. The routine approach to detect degradation is to measure weight loss. More sophisticated methods include measuring the reduction in molar mass using gel permeation chromatography (GPC) or to determine the loss of tensile properties using an Instron tensile test machine designed to detect changes in mechanical properties (Karlsson and Albertsson 1995). Other methods used to measure degradation rates include differential scanning calorimetry (DSC), scanning electron microscopy (SEM), chemiluminsence (CL), gas chromatography (GC) and liquid chromatography (LC) together with mass spectroscopy (MS).

The degradation mechanism of biodegradable polymers in an aerobic composting environment is similar to that for organic matter. Biodegradable polymers are attacked and disintegrated by enzymes from naturally occurring microorganisms, such as bacteria and fungi, encountered under specific conditions in composts. Biodegradation occurs when microorganisms colonise the surface of the polymer and secrete enzymes that break down the macromolecules (Nayak, 1999). The biodegradation process depends on several factors such as microbial activity, the surface area of the polymer, temperature, pH, molecular weight and polymer crystallinity (Davis, 2006). The rate of biodegradation is ultimately affected by the environment where the polymer is incorporated, the microorganisms utilized, and the nature of the polymeric substrate (Moore and Saunders, 1997). In an aerobic composting environment, biodegradable polymers are expected to go through complete mineralisation to CO_2 and H_2O:

$$\text{Polymer} + O_2 \rightarrow CO_2 + H_2O$$

The degradation products must not be toxic to the environment or persistent within the environment.

Composting of biodegradable polymers is considered as a permitted recovery option under the Producer Responsibility Regulations for Packaging Waste as amended in 1997. Davis (2006) stated that home composting of biodegradable packaging materials has the potential to divert waste from the municipal waste stream and compliment municipal composting. However, it was noted that polymer residues may be more persistent in compost bins than plant and food waste materials. Uncertainty about the suitability of different packaging materials for home composting could also discourage homeowners from attempting to compost packaging materials.

6.1.5 Compostability of starch-based biodegradable polymers

The compostability of two starch-based biodegradable plastics used in food packaging was tested in the research reported here (Section 6.2). These included a potato-starch tray and a polylactic acid (PLA) container. Vikman *et al*. (1995) evaluated the biodegradability of several starch-based materials, including native potato starch, by enzymatic hydrolysis at 37 and 80 ^{o}C. A total of 100 mg of the sample was incubated in 10 ml 0.1 M acetate buffer, pH 5, containing 250 µg per sample of glucoamylase and 0.02 % sodium azide to prevent microbial growth. Native potato starch with amylomaize starch were found to be more resistant to enzymic hydrolysis than native barley starch. Native potato starch was hyrdolysed completely at 80 ^{o}C in 1 h. Kale *et al*. (2006) studied the degradation of three commercially available biodegradable packaging materials made of poly (LD-lactic lactide) (PLA) under experimental composting conditions. The materials (500 ml bottles containing

96 % of L-lactide and 4 % D-lactide (NatureWorks™ PLA - Blair, NE), PLA tray and PLA deli container, both made of 94 % L-lactide and 6% D-lactide (Wilkinson Manufacturing Company – Fort Calhoun, NE)) were subjected to composting for 30 days with temperature (T) > 55 ^{o}C, relative humidity (RH) > 65 % and pH ≈ 7.5. The compost feedstock comprised cow manure and wood shavings (11.6 m^3 cow manure and 7.8 m^3 wood shavings) mixed with cows' feed in a proportion of 2:1. The polymer materials were buried into the waste pile after been placed on mesh. The PLA tray and deli container with 96 % L-lactide degraded more rapidly than the bottles with 94 % L-lactide. The PLA tray and PLA deli container degraded within 30 days, whereas the PLA bottles degraded in 45 days under the experimental composting conditions. The rate of degradation was mainly affected by the L-lactide content, the crystallinity of PLA, and the composting parameters (T, RH, pH). Kale *et al.* (2006) demonstrated that packages made of PLA could be composted in municipal/industrial facilities, but they could be difficult to destroy by home composting since PLA degradation is driven by hydrolysis, which needs high temperature conditions. Ho *et al.* (1999) showed that PLA films (NatureWorks™) physically disintegrated in a leaf composting environment within 2 weeks when the temperature and relative humidity ranged from 55 – 60 ^{o}C and 50 – 70 %, respectively.

Klauss and Bidlingmair (2004) examined the quality of compost derived from feedstocks containing biodegradable polymers. Packaging materials collected from households were mixed with organic waste and composted at a large-scale operational facility. The packaging materials included bags, trays, racks for fruit and vegetables, diaper packaging, dairy products, bakery and meat packaging, bin liners, and compostable catering products such as plates, cups and cutlery. Most of the packaging materials were made of starch or starch blends. The compost feedstock was monitored to ensure a ratio of 99:1 organic waste to packaging materials on a weight basis. The mature compost was identical to compost produced without inputs of packaging materials for a range of compost quality parameters (dry matter content, pH, organic matter content, rotting degree, mass of impurities, the degree of optical pollution, total zinc content, as an indicator for heavy metal contamination, and plant tolerance). Plant growing tests using the finished compost indicated that the compost containing the biodegradable polymers had the same positive effect on soil and plant characteristics as the control compost.

6.2 Experimental biodegradation results

Full details of the experimental treatments and procedures are described in Section 2.3

The degradation rates of the tested materials in home compost bins after 126 days varied depending on the composition of the packaging products. The results are shown in Figures 6.1 - 6.3 and a summary of the mean degradation profiles of the different packaging materials tested is shown in Figure 6.4.

Potato starch packaging was the most rapidly degraded material by HC tested in this investigation and 91 % of the input mass was destroyed after 67 days and complete degradation occurred after 126 days. The decomposition of waxed cardboard materials also occurred in home composters and varied from approximately 60 % degradation to almost complete destruction. Thus, solid bleached cardboard (doughnut box) achieved the highest degradation value (99 %) followed by waxed corrugated cardboard (Pizza box), which was degraded by 77 %. Lower rates of degradation (58-59 %) were measured for heavily waxed corrugated cardboard (laundry tablets box) and solid unbleached cardboard. Bleached white line chipboard (disposable plate) was degraded by 41 % whereas white line chipboard (breakfast cereal boxes) had lower rates of decomposition of 28 - 36 %. The degradation of folding boxboard (cheese box), which is heavily waxed for moisture resistance, and non-packaging waste cardboard (typical backing board to a notepad) was also relatively limited and equivalent to 37 and 38 % of the input mass. No degradation of PLA took place in home compost bins. This may be explained because the temperature conditions in home

compost (Table 5.1) are too low to support the decomposition of this material (Section 6.1.5).

The results presented here demonstrate the wide potential variation in degradabilities of common packaging materials used for household products. Waxing and coatings tend to render packaging materials less susceptible to decomposition by HC and increase degradation times. The extent of the degradation achieved for other types of bleached and unbleached cardboard and chipboards is also highly variable and this may depend on the properties and lignin content of the primary packaging components. This research emphasizes the need for improved guidance and advice regarding the suitability of different packaging materials for HC.

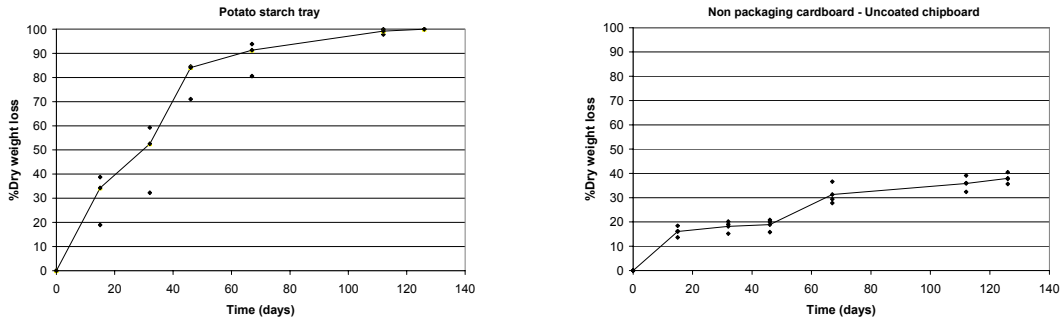

Figure 6.1 Degradation of potato starch-based polymer packaging and non-packaging cardboard (note-pad backing board) in home compost bins

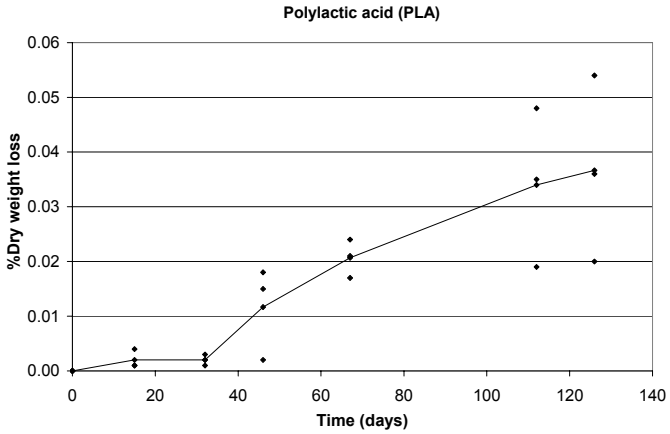

Figure 6.2 Degradation of polylactic acid-based polymer packaging in home compost bins

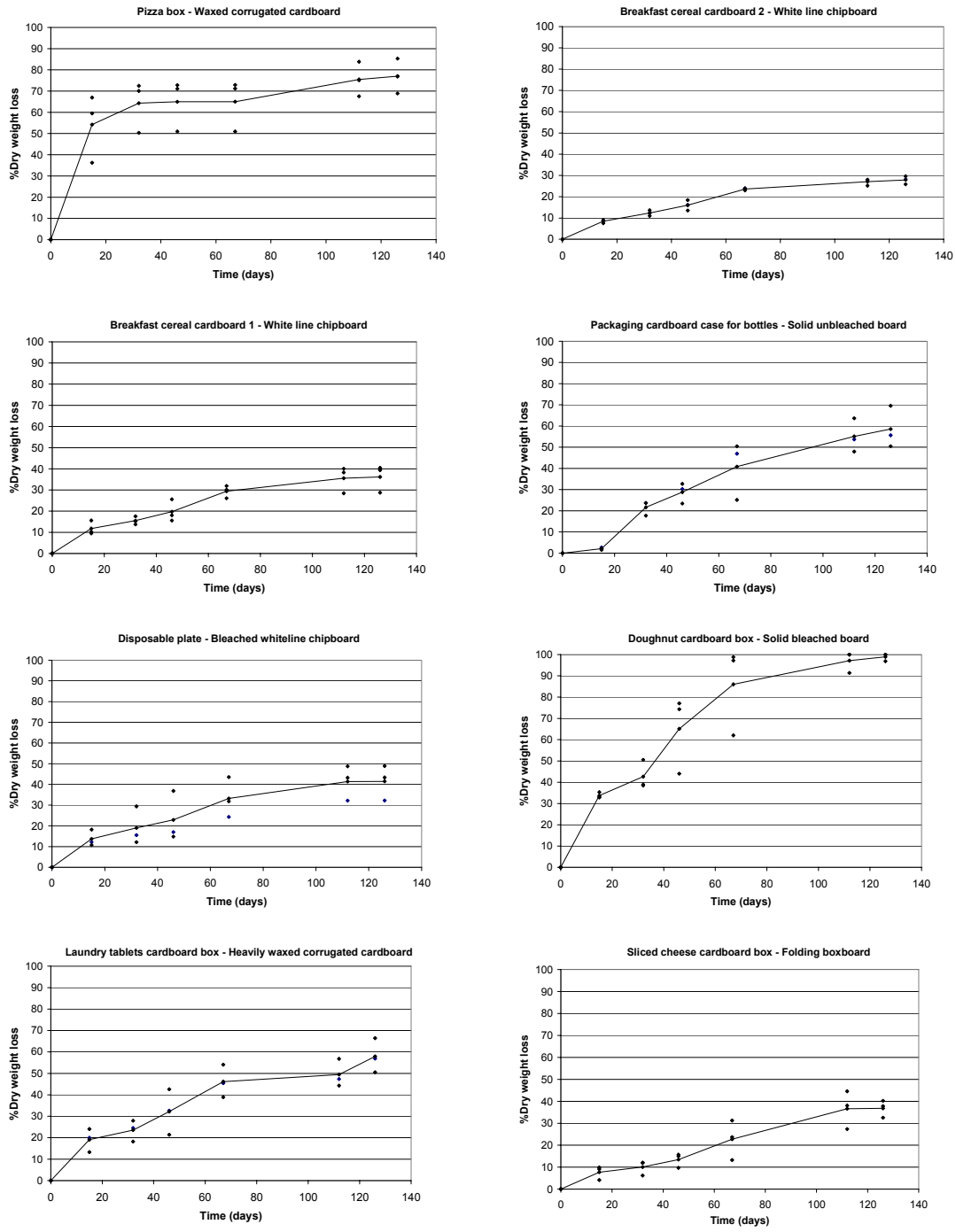

Figure 6.3 Biodegradation of cardboard packaging materials in home compost bins

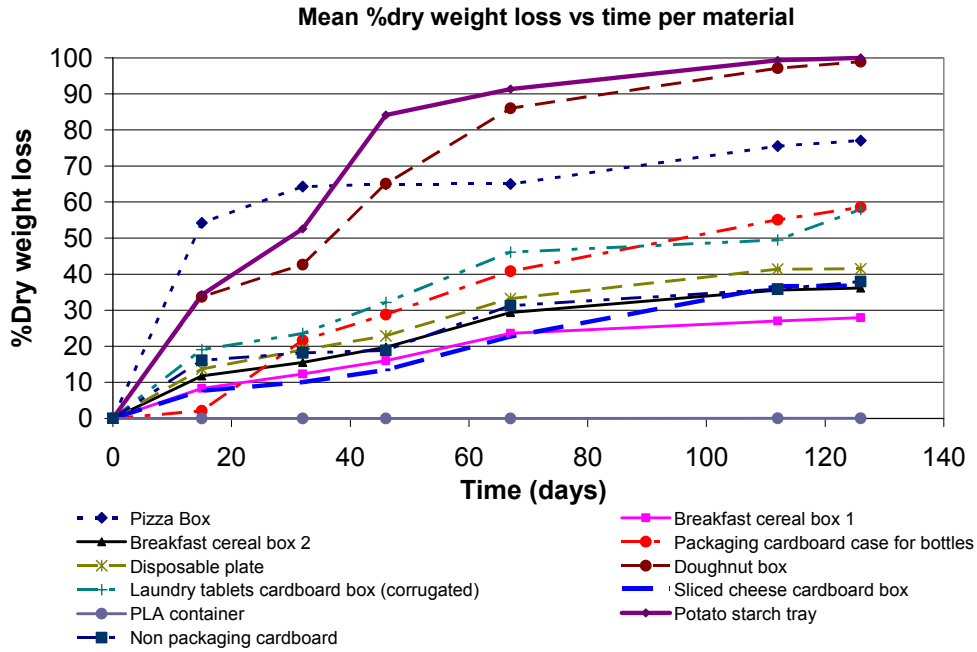

Figure 6.4 Summary of degradation profiles of different packaging materials in home compost bins