City of Columbia

701 East Broadway, Columbia, Missouri 65201

Agenda Item Number: REP 44-15
Department Source: City Manager
To: City Council
From: City Manager & Staff
Council Meeting Date: 4/6/2015
Re: Documentation of Perspectives Related to Single-Use Plastic Bag Ordinance

Documents Included With This Agenda Item

Documents provided to council by Dr. Frederick Vom Saal:

1. Accumulation and fragmentation of plastic debris in global environments;
2. Environmental implications of plastic debris in marine settings--entanglements, ingestion, smothering, hangers-on, hitch-hiking and alien invasions;
3. Preface
4. Plastics recycling: challenges and opportunities;
5. Phthalates and other additives in plastics: human exposure and associated health outcomes;
6. A critical analysis of the biological impacts of plasticizers on wildlife;
7. Monitoring the abundance of plastic debris in the marine environment;
8. Structuring policy problems for plastics, the environment and human health: reflections from the UK;
9. Biodegradable and compostable alternatives to conventional plastics;
10. Components of plastic: experimental studies in animals and relevance for human health;
11. Transport and release of chemicals from plastics to the environment and wildlife;
12. Plastics, the environment and human health: current consensus and future trends;
13. Our plastic age

Environment and Energy Commission report and proposed regulations concerning plastic bags

Links/Documents Related to February 3, 2015 Educational presentation and public comment forum on the proposed single-use plastic bag ordinance:

1. Main City of Columbia Website:
   http://www.gocolumbiamo.com/CMS/announcements/announcement.php?id=744
2. City of Columbia City Channel Website:
   https://www.youtube.com/user/columbiacitychannel
3. YouTube:
   https://www.youtube.com/watch?v=iVTQHDF_bLQ
4. Staff summary of public comments from public forum on proposed single-use bag ordinance.

March 2, 2015 Email from Ms. Amin to City Council: Additional Materials Associated with B51-15

A petition and signatures in support of a single-use plastic bag ban in Missouri, circulated by

the Osage Group of the Sierra Club.

February 16 Council Memo and Resolution/Ordinance Pertaining to the Regulation of Single-Use Plastic Bags: Supporting documentation includes: Missouri Farmer's Association (MFA) Response to Proposed Ordinance, Hy-Vee Response to Proposed Ordinance, American Progressive Bag Alliance (APBA) Response to Proposed Ordinance

Executive Summary

Per the request of Council person Hoppe, the following report provides a documented history of citizen, Environment and Energy Commission, business and staff input on the Single-Use Plastic Bag Ordinance that was introduced at the February 16, 2015 City Council Meeting and discussed at the March 2, 2015 Council Meeting. Although staff and council agreed to withdraw the ordinance, the comprehensive perspectives provided in this report offer insight for potential next steps related to plastic bag use in the City of Columbia.

Discussion

During the scheduled public comment portion of the October 20, 2014 Columbia City Council meeting, Osage Group of the Sierra Club representative, Jan Dye, and University of Missouri Professor, Dr. Frederick Vom Saal, addressed the City Council about the environmental and health impacts associated with plastic bags and offered a single-use plastic bag ordinance to the Council for consideration. Mayor McDavid requested that Dr. Vom Saal send the research he referred to in the discussion to Council. Dr. Vom Saal provided these documents to the City Clerk on November 22, 2014, who then forwarded them to the Columbia City Council and the City of Columbia Environment and Energy Commission (EEC) (See all Vom Saal documents attached and reference titles above).

At the end of the October 20, 2014 council meeting, Ward 6 Council person Barbara Hoppe requested that the Columbia EEC develop a report relating to the potential single-use plastic bag ordinance brought forth by Dye and Vom Saal.

The Columbia EEC sent their final report on the proposed single-use plastic bag ordinance to the Columbia City Council in January of 2015. Consequently, at the January 5, 2015 council meeting, Council person Karl Skala requested that the EEC report be sent to Columbia retailers, so they had an opportunity to provide a formal response to the ordinance, and any input on how they would like to approach the challenge of plastic bag waste (see attached Council Memo and Resolution/Ordinance Pertaining to the Regulation of Single-Use Plastic Bags).

The full EEC report was formally presented to Council at the January 20, 2015 City Council Meeting (see attached EEC Report and Proposed Regulations Concerning Plastic Bags).

During the January 20 council discussion of the EEC report, Ms. Hoppe requested that the Environment and Energy Commission and the Sierra Club hold a public education forum on the issue and for the forum to be videotaped so it could be viewed by people that could not attend. An

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Educational presentation and public comment forum on the proposed single-use plastic bag ordinance was held in Council Chambers on February 3, 2015 and arranged by the Environment and Energy Commission.

The forum can be viewed through the three following links:

Main City of Columbia Website:
http://www.gocolumbiamo.com/CMS/announcements/announcement.php?id=744
City of Columbia City Channel Website:
https://www.youtube.com/user/columbiacitychannel
YouTube:
https://www.youtube.com/watch?v=iVTQHDF_bLQ

In all, 26 residents spoke at the Public Forum on the Proposed Single-Use Plastic Bag Ordinance. One speaker simply had a neutral question. Ten speakers spoke against the potential ordinance. General themes for reasons the ordinance should not be passed centered on freedom of choice, suggestions to focus instead on education and littering, the idea that paper bags are more harmful to the environment than plastic bags, the need for a more precise baseline of current bag use and litter, measurable reduction goals and evidence of results from other communities.

Fifteen speakers spoke in support of the ordinance. However, please note that one of the speakers was Dr. Vom Saal, who was also a presenter. General themes for reasons the ordinance should be passed were the high rates of plastic bag litter in Missouri waterways, high rates of plastic bag litter in the City of Columbia, health impacts related to plastic bag waste in the environment, the thought that many other communities are enacting similar policies, and the idea of good stewardship for the Columbia community. (See Staff summary of public comments from public forum on proposed single-use bag ordinance).

At the January 20, 2015 City Council Meeting Ms. Hoppe also made a motion directing staff to introduce a single-use plastic bag ordinance, as proposed by the Environment and Energy Commission, for the February 16, 2015 Council Meeting. The motion was seconded by Mr. Thomas. The Council Memo and Resolution/Ordinance Pertaining to the Regulation of Single-Use Plastic Bags was included as a first read at the February 16, 2015 Council Meeting, and it was then discussed at the March 2, 2015 Council Meeting.

A number of items were presented with the Council Memo and Resolution/Ordinance Pertaining to the Regulation of Single-Use Plastic Bags at the March 2, 2015 Council Meeting. On February 26, 2015, the Osage Group of the Sierra Club submitted a petition to the city clerk comprising 899 signatures from Columbia residents and 120 signatures from individuals outside of the city limits in support of a single use plastic bag ban in Missouri. Due to time constraints, the first page of the petition describing its contents was included in the staff memo, and a digital version of the full petition with signatures was sent to council the day of the March 2, 2015 Council Meeting (see attached email Additional Materials Associated with B51-15, and the Petition and signatures in support of a single-use plastic bag ban in Missouri).

In addition, the Council Memo included estimates of the cost, staff and volunteer time spent on abating plastic bag waste; an approximate baseline of current city-wide plastic bag usage; reasons plastic bags cannot be recycled at the City's Material Recovery Facility (MRF); and a summary of retailer and plastic manufacturer's feedback to the proposed ordinance (see attached Council Memo and Resolution/Ordinance Pertaining to the Regulation of Single-Use Plastic Bags).

Staff recommended the ordinance be withdrawn to allow staff time to collect more input from the general population and retailers, and to potentially engage in a community education outreach effort, and Council concurred. Ms. Hoppe requested a final report to council, so that all documentation of perspectives on the Single-Use Plastic Bag Ordinance could be found in one place. In addition to this report, an internal webpage has been developed that includes all relevant documentation. Staff has suggested that the EEC add the webpage to their section of the City's website, in order to make this information easily accessible to the public. Staff is attending the next EEC meeting on April 28, 2015 to discuss this possibility further.

Fiscal Impact

Short-Term Impact: N/A
Long-Term Impact: N/A

Vision, Strategic & Comprehensive Plan Impact

Vision Impact: Environment
Strategic Plan Impact: Not Applicable
Comprehensive Plan Impact: Environmental Management

Suggested Council Action

The report is informational only.

Legislative History

Included in the discussion of the report.

Leah Christian
Department Approved

City Manager Approved

City of Columbia
701 East Broadway, Columbia, Missouri 65201

SUPPORTING DOCUMENTS INCLUDED WITH THIS AGENDA ITEM ARE AS FOLLOWS:

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PHILOSOPHICAL TRANSACTIONS OF THE ROYAL SOCIETY B

Phil. Trans. R. Soc. B (2009) **364**, 1985–1998
doi:10.1098/rstb.2008.0205

Accumulation and fragmentation of plastic debris in global environments

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One of the most ubiquitous and long-lasting recent changes to the surface of our planet is the accumulation and fragmentation of plastics. Within just a few decades since mass production of plastic products commenced in the 1950s, plastic debris has accumulated in terrestrial environments, in the open ocean, on shorelines of even the most remote islands and in the deep sea. Annual clean-up operations, costing millions of pounds sterling, are now organized in many countries and on every continent. Here we document global plastics production and the accumulation of plastic waste. While plastics typically constitute approximately 10 per cent of discarded waste, they represent a much greater proportion of the debris accumulating on shorelines.

Mega- and macro-plastics have accumulated in the highest densities in the Northern Hemisphere, adjacent to urban centres, in enclosed seas and at water convergences (fronts). We report lower densities on remote island shores, on the continental shelf seabed and the lowest densities (but still a documented presence) in the deep sea and Southern Ocean. The longevity of plastic is estimated to be hundreds to thousands of years, but is likely to be far longer in deep sea and non-surface polar environments. Plastic debris poses considerable threat by choking and starving wildlife, distributing non-native and potentially harmful organisms, absorbing toxic chemicals and degrading to micro-plastics that may subsequently be ingested. Well-established annual surveys on coasts and at sea have shown that trends in mega- and macro-plastic accumulation rates are no longer uniformly increasing: rather stable, increasing and decreasing trends have all been reported. The average size of plastic particles in the environment seems to be decreasing, and the abundance and global distribution of micro-plastic fragments have increased over the last few decades. However, the environmental consequences of such microscopic debris are still poorly understood.

Keywords: persistent organic pollutants; marine debris; plastic production; landfill; microplastic

1. INTRODUCTION

In the last half-century, there have been many drastic changes on the surface of the planet, but one of the most instantly observable is the ubiquity and abundance of plastic debris. Like many anthropogenic impacts on natural systems, it is one that, despite widespread recognition of the problem, is still growing and even if stopped immediately will persist for centuries. From what started as a perceived aesthetic problem of plastics littering towns, countryside, shores and even far out into the ocean soon emerged as causing the choking and entanglement of wildlife. The number of potentially harmful implications of plastic debris that have been identified has escalated and it is now realized that these items may also transport persistent organic pollutants (POPs; Mato *et al.* 2001), non-indigenous species to new locations (Barnes 2002) and distribute algae associated with red tides (Masó *et al.* 2003). Reports of accumulation of plastics spread rapidly in terms of the taxa influenced, geography and bathymetry of affected sites and countries beginning monitoring and beach clean-up operations. Schools and voluntary organizations have made annual coastal collections of stranded plastics; an important educational issue even on many of the planet's most remote islands. In some areas though, notably on the seabed, assessment of plastic accumulation has been relatively neglected (Goldberg 1994). Since 1990, the dumping of rubbish at sea from ships has been prohibited under the international shipping regulation MARPOL Annex V. A reduction of ship-derived plastic debris should therefore be expected, even if global use of plastics continues to increase. To gain an accurate and meaningful assessment of plastics and their influence, large-scale and long-term monitoring is needed across countries and environments (including the sea floor)

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One contribution of 15 to a Theme Issue 'Plastics, the environment and human health'.

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and across a range of debris sizes. These can broadly be divided into macro-debris (>20 mm diameter), meso-debris (5–20 mm) and micro-debris (<5 mm); here we also use the term mega-debris (>100 mm) (see Ryan *et al.* 2009; Thompson *et al.* 2009).

Natural marine debris of some type (e.g. pumice) has floated on the surface of the global ocean for longer than life itself, but life greatly increased this through floating algae, shells, seeds, fruits and wood. Human activities and travel by water must have further greatly increased flotsam (e.g. by timber), but by far the biggest change in the potential for transport by debris came with the mass production of plastics. The accumulation of both macro- and micro-plastics has consistently increased on shores and in sediments for the last four decades (see Thompson *et al.* 2004; Barnes 2005, respectively). Their inexpensive, lightweight and durable properties have made plastic much more single use and 'throw-away' than previous synthetic artefacts. Such compounds do deteriorate in ultraviolet (UV) light, but haline environments and the cooling effect of the sea mean degradation require very long exposure times (Gregory 1999). Because plastics become fouled by marine organisms relatively quickly, the debris may also become shielded to some extent from UV light, and the persistence of this debris was recently illustrated by accounts that plastic swallowed by an albatross had originated from a plane shot down 60 years previously some 9600 km away (Weiss *et al.* 2006).

Mega-debris at sea was highlighted by tens of thousands of each of basketball shoes, hockey gloves and bath toys released from containers washed off of ships (Weiss *et al.* 2006). There are many sources for plastics accumulating in the environment from direct dropping and dumping of litter on land or at sea to blowing from landfill sites, losses in transport and accidents. Typically, 40–80% of mega- and macro-marine debris items are plastic, much of it packaging, carrier bags, footwear, cigarette lighters and other domestic items (Derraik 2002; Barnes 2005). A recent study by Ivar do Sul & Costa (2007) across Central and South America also found marine debris dominated by land-based plastic (though sometimes fishery gear can be abundant along continental shores as well). At more remote islands, fishing-related sources of debris are often more prevalent. Following establishment of 'long-term' monitoring surveys of stranded debris in the 1990s, there are now sufficient data to explore seasonal, annual and long-term patterns (e.g. Morishige *et al.* 2007).

Most waste plastics, including the large proportion used in single-use applications such as packaging, are disposed of in landfill sites. However, plastic persists in landfill sites and if not properly buried may later surface to become 'debris'. Durability of plastic ensures that wherever it is, it does not 'go-away'; that is, by placing plastics in landfill we may simply be storing a problem for the future. Although accumulation of plastics on land is important, little information is available on the amounts, rates, fate or impacts, whereas there has been a major effort to quantify impacts on shorelines and at sea. In this paper, we examine waste generation and disposal, together with the abundance, composition and fragmentation of plastic. We then consider temporal and spatial trends in accumulation of plastics on strandlines, the sea surface and at depth on the seabed. We assess published data and present new surveys and observations of spatial and temporal patterns to evaluate whether persistent marine debris, such as plastics, are still increasing and whether it varies geographically?

2. ANTHROPOGENIC WASTE AND PLASTIC ACCUMULATION IN LANDFILL

Plastics are present in most waste, and before trends in accumulation of plastic can be explained, it is important to first consider waste generation and disposal. Global production of plastics is estimated at 225 mt yr^{-1} (APME 2006). Waste composition data are useful to identify the relative quantity and types of plastic. As discussed in the contribution by Teuten *et al.* (2009), different plastics and resins have widely varying properties with respect to contaminant sorption and desorption.

(a) *Waste generation*

Waste is typically categorized based on its point of generation. Categories include municipal, commercial, industrial, agricultural and construction and demolition (C&D). However, there is ambiguity within these categories. For example, in the USA, municipal solid waste (MSW) includes that generated in residential, commercial and institutional (e.g. schools, government offices) sectors, while in other countries, MSW may include anything from residential waste only to all waste managed in the municipal system (e.g. C&D, non-hazardous industrial). This complexity is exacerbated by the fact that some municipal systems manage residual materials from the treatment of water and wastewater. This relatively heavy waste will distort the composition of dry wastes such as plastics.

Considering these multiple categories, it is difficult to compare waste composition between countries. Waste is typically classified by the agency in need of the information, and surveys are typically designed with specific goals. For example, a waste sort conducted to support planning of a recycling programme would identify commonly recycled plastics, including pigmented and translucent high-density polyethylene (HDPE) containers and clear and pigmented polyethylene terephthalate (PET), and classify the remaining plastics as 'other'. These categories are useful in this (recycling) context, but are less complete for a study of plastics in the environment. Another confounding issue is that the types of plastics present vary between municipal, agricultural and C&D waste. Municipal waste is dominated by containers (e.g. drink bottles) and films (e.g. carrier bags, packaging sheets), agricultural waste may contain large quantities of a single film and C&D waste may contain polyvinyl chloride (PVC) pipe and large plastic containers. Thus, a municipal stream that contains 10 per cent (by mass) plastics is not equivalent to a C&D stream containing the same percentage.

Waste composition may also be presented on either an 'as-generated' or 'as-discarded' basis. The former includes all the waste generated in a particular sector, prior to separation for recycling, composting or other treatment. In contrast, 'as-discarded'

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indicates the waste remaining for disposal after the aforementioned separation. In areas with significant recycling programmes, the difference between waste generation and waste disposal could be 20–40%, and waste composition will change as recyclables are removed. If properly managed at the end of its useful life, plastic waste may be recycled, burned in combustion facilities to generate energy or buried in landfill. In each of these alternatives, the waste should be destroyed or contained, so that plastic is not released to the environment. The major release of plastics to the environment is the result of inappropriate waste management and improper human behaviour, e.g. littering (abandoning waste away from collection points). For example, plastic films can be released to the environment when not transported properly, and as a result of wind-blown litter at the point of burial in a landfill. Well-operated landfills include a daily cover over the waste consisting of soil or a synthetic material and fences surrounding the landfill to contain wind-blown debris.

(b) *Plastics production and recycling*

Annual global consumption of the major plastic resins is considerable (Andrady & Neal 2009). Films (e.g. carrier bags, plastic sheets) are easiest to escape containment as wind-blown debris and are likely the major component of terrestrial plastic litter but plastic litter also includes discarded fishing equipment, food and beverage packaging and many other items that are present in the marine environment (Koutsodendris *et al.* 2008). Films are dominated by low-density polyethylene (LDPE)/linear LDPE (LLDPE). We present information on plastics in MSW in the USA and their management (table 1). The quantities recovered (i.e. for recycling) as a fraction of total discards shows that recycling rates are relatively low. In the USA, plastic recycling is largely limited to drink containers although local authorities continue to expand the types of plastics collected for recycling. In general, citizen participation rather than industrial capacity limits the quantities of plastics recycled. Efforts to provide incentives for recycling can increase the fraction recycled (Loughlin & Barlaz 2006).

In the USA, durable goods, products that last on average for >3 years and include items such as furniture and appliances, were the most important use for new plastics (figure 1). Non-durable goods, products that are consumed in <3 years such as trash bags and eating utensils, were the next biggest use category. In Europe, data on various packaging applications are typically combined rather than considered separately and hence disposable packaging represents the principal use of plastics (37%, PlasticsEurope 2008).

(c) *The fraction of plastic in household waste*

Plastics in the waste from various countries is estimated at approximately 10 per cent (of mass). Such estimates can only be used as an indication of plastics composition for several reasons. First, the data are not all from the same year. Second, where possible, data are on an 'as-discarded' basis to reflect the composition of waste after diversion for recycling. However, it is not always clear whether the data were reported 'as-generated' or 'as-discarded'. Third, the waste components included in national surveys vary within and between countries. For example, the US data are for wastes defined as MSW. Finally, country-specific data compiled for Europe (Eurostat 2007) are self-reported at the national level and are unlikely to have been generated using a consistent methodology. In the USA, plastics are estimated to comprise 11.8 and 16.3 per cent of MSW as-generated and as-discarded mass, respectively. The composition of discarded plastics is given in table 1 (US EPA 2006). In Europe, plastics are estimated to comprise 7 per cent of waste mass as-generated. Similarly, plastics were estimated to represent 5.8, 7.3, 8–10 and 10 per cent of waste mass in Singapore, Australia, the UK and Finland, respectively (Barlaz 2006; Burnley 2007; Sokka *et al.* 2007). Finally, plastics were estimated to comprise 4 and 13 per cent of waste in regions of China that use coal and natural gas, respectively, and the country-wide average for urban areas is projected to be 14 per cent plastics in 2030 (World Bank 2005). Despite the uncertainty, estimates from around the world are reasonably consistent in estimating plastics to comprise approximately 10 per cent of municipal waste mass. In contrast, plastics comprise 50–80% of the waste stranded on beaches, floating on the ocean surface and on the seabed (Gregory & Ryan 1997; Derraik 2002; Barnes 2005; Morishige *et al.* 2007).

Table 1. Plastics production, recovery and disposal in the USA in 2005 (thousands of metric tonnes). Adopted from US EPA (2006). The data originated in reports of The American Plastics Council and include net imports. Plastic from the construction and agricultural sectors are not included in these quantities.

	generation of plastics in MSW	recovery	discards
PET	2600	491	2109
HDPE	5355	473	4882
PVC	1491	0	1491
LDPE/LLDPE	5864	173	5691
polypropylene	3636	9	3627
polystyrene	2355	0	2355
other	4982	355	4627
total	26 282	1500	24 782

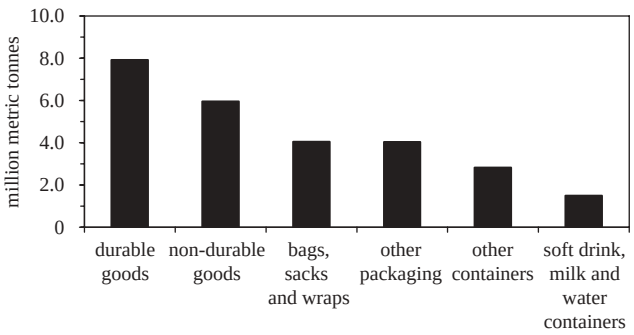

Figure 1. Production of plastic products in the USA in 2005 (reproduced with permission from US EPA, 2006).

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3. TEMPORAL AND SPATIAL TRENDS IN ACCUMULATION

(a) *Ocean surface and beaches*

Many plastics are buoyant (46%; US EPA 2006) and remain so until they become waterlogged or amass too much epibiota to float. Plastic items are commonly found at the sea surface or washed up on the shoreline. Mass production of plastics began in the 1950s, so less than a century ago we estimate that the amount of anthropogenic debris at sea would have been three to four orders of magnitude lower and restricted to much more degradable items. Some of the earliest accounts of plastic debris in the marine environment are of fragments and pellets ingested by seabirds in the 1960s (e.g. Kenyon & Kridler 1969; Harper & Fowler 1987), but now plastic mega- and macro-debris is routinely observed from boats everywhere on the planet. There has been a rapid and substantial increase in anthropogenic debris on the ocean surface and beaches over recent decades (e.g. Dixon & Dixon 1981; Derraik 2002; Barnes 2005), but of more pertinence now are the current spatial trends. Surveys of anthropogenic debris and clean-up operations have generally focused on the larger items along strandlines, and there is a wide geographical variability in the type of data available to examine potential trends. However in the last three of decades, it has become apparent that the raw material for making plastics, tiny pellets and micro-plastics have become more numerous (as marine debris) and, like larger pieces, these can travel considerable distances. Volunteer observations and collections in a growing number of nations are aiding our understanding of the scale and pattern of distribution of larger size fractions of plastics in the marine environment, but specialist examination is generally needed to investigate accumulation of micro-plastic, e.g. in sediments (Thompson *et al.* 2004). Beaches are the most easily accessible areas for studying marine debris (although such studies have some confounding factors), yet despite the establishment of many study sites, irregularity of sampling, differing protocol and observers have led to very few datasets spanning more than a decade (Barnes & Milner 2005).

The distribution of plastic debris is very patchy at sea for a variety of reasons, including local wind and current conditions, coastline geography and the points of entry into the system such as urban areas and trade routes. For example, stranding of larger size fractions of plastics is between one and two orders of magnitude less per length of coastline on remote shores and at large spatial scales, abundance correlates very strongly (Pearson's correlation $= 0.971$, $p < 0.001$) with human population (per 10° latitude; Barnes 2005). Enclosed seas and semi-enclosed seas such as the Caribbean (Coe *et al.* 1997), typically have high densities of plastic debris but also considerable variability. High densities and variability can also be a feature of open ocean coastlines e.g. Brazil (Santos *et al.* 2005) and Hawaii (Dameron *et al.* 2007). One of the key sources of inter-annual variability seems to be changes in oceanic circulation driven by El Niño events (Matsumura & Nasu 1997; Morishige *et al.* 2007). Typically about 2000 and 500 items of anthropogenic debris strand on north and south Atlantic Ocean shores (respectively) per linear kilometre per year, of which more than half is plastic (scaled up from surveys of items >1 cm in size along 200 m long beach sections; Barnes & Milner 2005). More than six times as much plastic strands in the Mediterranean Sea and less than six times as much strands in the Southern Ocean shores (Barnes & Milner 2005; table 2). Despite considerable variability in observation and accumulation rates of plastic debris, some temporal trends do emerge. Studies initiated in the 1980s and 1990s indicated that the rate of plastic stranding from oceanic sources showed a sustained and considerable increase over time (e.g. Ryan & Moloney 1993; Ribic *et al.* 1997; Torres & Jorquera 1999). Similarly, the occurrence of macro-plastics associated with wildlife (e.g. in bird nests and stomachs, entangling seals, strangling a wide variety of vertebrates or even used by hermit crabs instead of shells; Barnes 2005) also drastically increased. For example, between 1992 and 2005 the frequency of plastic garbage items in kittiwake nests increased from 39.3 to 57.2 per cent in northwest Denmark (Hartwig *et al.* 2007). Monitoring of strandings and effects on mega-fauna (such as birds) has now commenced on at least a few remote island shores in every ocean, and these, with negligible local sources of plastics, have revealed the scale at which anthropogenic debris is accumulating. Barnes (2005) found high levels but no consistent temporal trends in the abundance of anthropogenic debris on northern hemisphere shores compared with much lower levels, but increased densities through the 1980s, 1990s and early 2000s were reported in the southern hemisphere. The highest increases were at high southern latitudes (Barnes 2005). However, new data (reported here) show that patterns of stranding on islands are no longer clearly increasing and may be stabilizing, though often with a 'noisy' signal of annual variability (figure 2, see also Ryan *et al.* 2009). A similar lack of clear temporal trend in stranding densities of plastics is apparent in data collected intermittently at Ascension I., in the tropical Atlantic Ocean, and in the Falkland Is., south Atlantic Ocean (D. K. A. Barnes 2002, 2003 unpublished data). Approximately 27 per cent of macro-debris items stranding at Ascension I. was fishery-related, similar to remote Tern I. in the Hawaiian Is. (Morishige *et al.* 2007). This is much less than on shores adjacent to important fisheries e.g. in Brazil (Oigman-Pszczol & Creed 2007) or even sub-Antarctic Bird I. (Walker *et al.* 1997). Bird I. and Signy I. in the Southern Ocean (figure 2) have stranding densities of plastics an order of magnitude lower than remote localities at low latitudes, which in turn have at least an order of magnitude fewer plastics per kilometre than urban sites. Further south in the Southern Ocean, debris washes ashore much more rarely at Adelaide Island (west Antarctic Peninsula). The relatively consistent level of abundance for macro- and mega-debris at sea at high southern 1latitudes is supported by recent resurveys around the Drake Passage, Scotia Arc and northern Antarctic Peninsula (figure 3). Fifteen years after the first (Barnes & Milner 2005), the most recent survey of this area took place early in 2008 and will involve the first marine debris surveys of the south Bellingshausen and Amundsen seas.

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Table 2. Densities and proportion of plastics among benthic marine litter worldwide (per number of items). M, Mediterranean Sea; B, Baltic Sea; NA, North Atlantic; NP, northern Pacific Ocean; WP, western Pacific Ocean; T, trawling; PT, pole trawling; SA, South Atlantic.

region	sea	method	item Ha^{-1}	plastic (%)	references
NA	Bay of Biscay	T	1.42 ± 0.25	62.2	Galgani *et al.* (1995*a*)
M	NW Mediterranean	T	19.35 ± 6.33	77.1	Galgani *et al.* (1995*b*)
B	Baltic Sea	T	1.26 ± 0.82	35.7	Galgani *et al.* (2000)
NA	North Sea	T	1.56 ± 0.37	48.3	Galgani *et al.* (2000)
NA	Channel East	T	1.176 ± 0.067	84.6	Galgani *et al.* (2000)
NA	Bay of Seine	T	1.72 ± 0.058	89	Galgani *et al.* (2000)
NA	Celtic Sea	T	5.28 ± 2.47	29.5[a]	Galgani *et al.* (2000)
SA	Rio de la Plata	T	0–15.09	74	Acha *et al.* (2003)
M	Greece, 59 sites	T	149	55.5	Katsanevakis & Katsarou (2004)
M	Greece, Patras gulf	T	0.89–2.40	79–83	Stefatos *et al.* (1999)
M	W & S Greece	T	0.72–4.37	55.9	Koutsodendris *et al.* (2008)
M	Gulf of Lion	T	1.43 ± 0.19	70.5	Galgani *et al.* (2000)
M	East Corsica	T	2.29 ± 0.72	45.8	Galgani *et al.* (2000)
M	Adriatic Sea	T	3.78 ± 2.51	69.5	Galgani *et al.* (2000)
M	Sicily/Tunisia channel	T	4.01	75	Cannizarro *et al.* (1995)
M	Oriental basin	P T	5.85–161.98	37	Galil *et al.* (1995)
NP	Kodiak Island, Alaska	T	0.11–1.47	47–59	Hess *et al.* (1999)
NP	Oregon Coast	T	1.49	26[a]	June (1990)
NP	Bering Sea	T	0.075–0.51	27	June (1990)
NP	Norton Sound	T	2.49	49.0	June (1990)
WP	Tokyo Bay	T	2.70–5.50	40.1–41.6	Kanehiro *et al.* (1995)
WP	Tokyo Bay	T	1.85–3.38	48.3–58.9	Kuriyama *et al.* (2003)
WP	Eastern China Sea	T		<5	Lee *et al.* (2006)
WP	South Sea of Korea	T		<10	Lee *et al.* (2006)

[a]fishing area.

Visual surveys such as these are weaker as a source of data than surface-towed trawls but much more common and thus arguably comparable with data collected elsewhere, despite being semi-quantitative. Gregory *et al.* (1984) reported similarly low (on a global scale) levels of floating anthropogenic debris in the Ross Sea (Pacific sector) of the Southern Ocean. Observers from the University of Essex in conjunction with Greenpeace are currently undertaking repeat survey of plastics at sea in this area. As on surrounding strandlines, the north Atlantic Ocean and Pacific Ocean have high densities of floating plastic debris, especially at 20–40° N within a few hundred kilometres of the coast and in the gyre centres, e.g. between the tropical and subarctic waters (Matsumura & Nasu 1997). A recent (2005) survey of the subtropical convergence zone in this area showed plastic debris to be concentrating there remotely using satellite imagery (Pichel *et al.* 2007).

We know much less about the use by and distribution of organisms that hitch hike on plastics and other anthropogenic debris than about the debris itself. Macro- and mega-plastics have the potential to carry a wide range of species and support the growth of many to reproductive viability. The high abundance, lengthy durability and travel of plastics to even the most remote coasts make them a major potential vector for the dispersal of organisms (Gregory 2009). New data from surveys of marine debris stranding in the Seychelles in 2005 and 2006 showed that on some beaches >60 per cent of items carried fouling organisms, the highest reported anywhere (D. K. A. Barnes 2002 onwards, unpublished data). This is of significance because the prevailing currents that travel from north Australia and south Indonesia during summer (South Equatorial) and from Somalia, India and N. Indonesia during winter (Indian Monsoon) could potentially transport a very wide range of species to less biodiverse, mid-ocean islands. Recent surveys of marine debris at Ascension I. (reported here for the first time) found 38, 40 and 41 per cent of debris colonized by fauna in 2002, 2003 and 2005, respectively. Much of this had probably also travelled considerable distances given the prevailing currents come from the cape of South Africa. The likely response of many species to rapid regional warming is to move pole-ward to stay within their normal thermal envelope, but in previous phases of warming (interglacial periods), there were few vectors to travel on. Now plastic debris, ship hulls and other vectors make transport more rapid and frequent, and unprecedented warming at high latitudes also means that establishment success of potential invaders is likely to be higher.

(b) *Seabeds from shallows to abyss*

As at the surface, both in the open ocean and on strandlines, it is clear that the abundance and distribution of anthropogenic debris show considerable spatial variability. The geographical distribution of plastic debris is strongly influenced by hydrodynamics, geomorphology and human factors. Moreover, there is notable temporal, particularly seasonal, variation with a tendency for accumulation and concentration along coastal and particular geographical areas.

Under the weight of fouling by a wide variety of bacteria, algae, animals and accumulated sediment,

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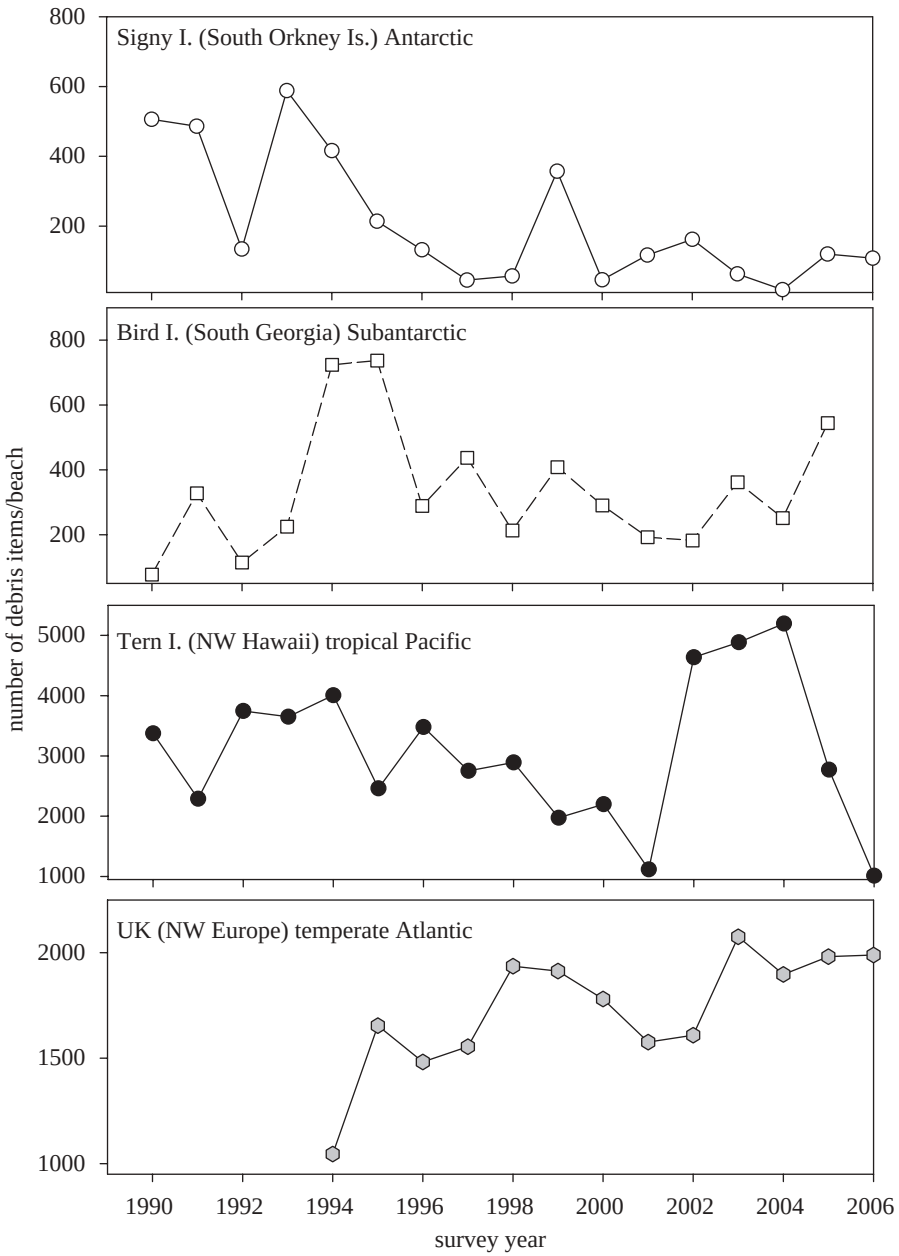

Figure 2. Annual accumulation of all marine debris (predominantly plastic) on shores of selected islands with year. Data for Bird I. and Signy I. are from Walker *et al.* (1997); Convey *et al.* (2002) and CCAMLR. Data for Tern I. are from Morishige *et al.* (2007) and for the UK from Beachwatch 2006 (MCS 2007).

plastics can sink to the seabed (R. C. T. 1997 unpublished data). Change in the nature, presence or abundance of anthropogenic debris on the sea floor is much less widely investigated than surface patterns. Studies that investigate seabed debris typically focus on continental shelves, and research into the deeper seabed, which forms about half the planet's surface, is restricted by sampling difficulties and cost. Patterns in even the shallow subtidal can differ substantially from the adjacent strandlines. Oigman-Pszczol & Creed (2007) found plastic to constitute a much greater proportion of debris on the nearshore Brazilian seabed than on the shore. While sonar does not enable discrimination of different types of debris, trawling (e.g. using Agassiz) is probably the most adequate method to date, particularly when mesh size and opening width can be manipulated (Goldberg 1994, 1995; Galgani & Andral 1998). Such nets are only semi-quantitative and because of their design for collecting epibenthos, probably underestimate the quantities of debris present. Therefore, pole trawling, with a constant mouth width, which works deeper in sediments, is considered the best approach. To date all off-shelf trawl data from submersibles have used this methodology. General strategies to investigate seabed debris are similar to methodology for benthic ecology and place more emphasis on the abundance and nature (e.g. bags, bottles, pieces of plastics) of items rather than their mass. Interpretation of trends is made difficult because the ageing of plastics at depth is not well researched and the fall of plastics to the seabed began long before specific scientific investigations started in the 1990s. Plastics have been found on the seabed of all seas and oceans across the planet, but macro-debris is still very rare in the Southern Ocean, particularly in deep water. For example a

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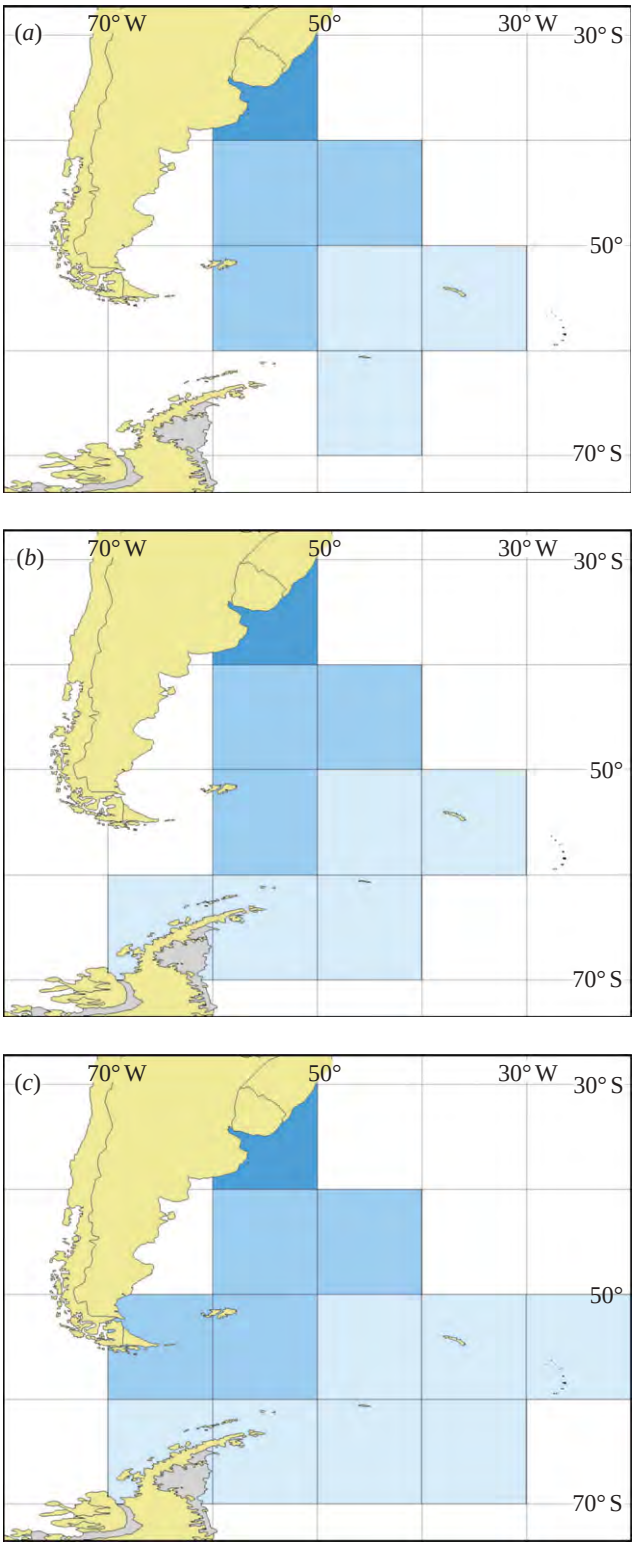

Figure 3. Densities of all marine debris, predominantly plastic, at sea in the southwest Atlantic and Atlantic sector of the Southern Ocean by 10° latitude and longitude areas. Shades of light to dark blue code for [r1]densities 0–1, 2–10, 11–100, 101–1000 and 1001+ items square kilometre, respectively. The survey years are (*a*) April 1993, (*b*) April 2002 and (*c*) April 2006. Data from Barnes & Milner (2005) and reported here for the first time.

recent series of 32 Agassiz trawls and 29 epibenthic sledge tows (at 200–1500 m depth, B. A. S. 2006 unpublished data) around the most (human) visited area, the northern Antarctic Peninsula and Scotia Arc,

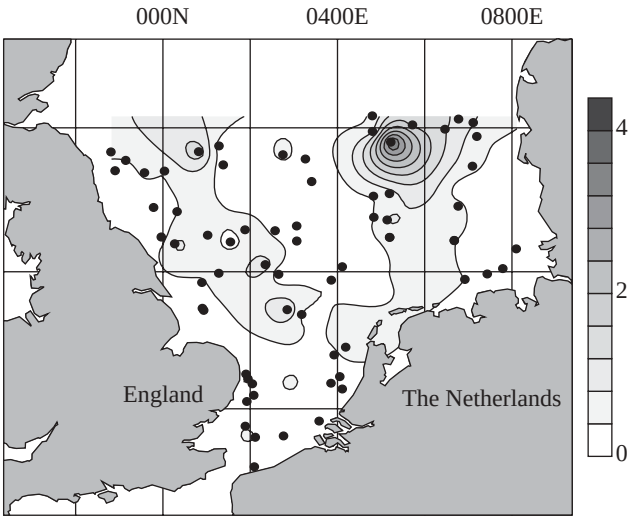

Figure 4. Plastic debris on the seabed from the southern North Sea (North Atlantic) in 1999. Plastics were counted after 30 min trawl time (16 m mouth, 20 mm mesh) at 64 stations (•) on the continental shelf. Results are given as items Ha^{-1} (10 000 m^2).

found just one plastic piece and one metal shot. Large-scale evaluations of seabed debris distribution and densities anywhere are scarce (but see Galgani *et al.* 2000; Lee *et al.* 2006; Koutsodendris *et al.* 2008). However, there are a large number of small-scale studies that have investigated anthropogenic debris in coastal areas such as bays, estuaries and sounds (see table 2 and references therein).

The abundance of plastic debris is very dependent on location, with values ranging from 0 to 7290 items per hectare (Ha) (although an extreme find of 10 110 anthropogenic items Ha^{-1} was found in 1998 at one position, 43°42′84″ N, 7°22′98″ E using a pole trawl). Assessments of abundance clearly demonstrate the domination of this debris by plastics, as at more than half the study sites plastics constituted >50 per cent of debris (table 2). Of the areas investigated to date, Mediterranean sites tend to show the greatest densities owing to the combination of a densely populated coastline and shipping in coastal waters and a lack of dispersion of plastics because of limited tidal flow or water circulation. In general, bottom debris tends to become trapped in areas of low circulation and high sediment accumulation in contrast to floating debris, which accumulates in frontal areas. Debris that reaches the seabed may already have been transported considerable distance, only sinking when weighed down by fouling. The consequence is an accumulation of plastics debris in bays rather than the open sea (Hess *et al.* 1999; Stefatos *et al.* 1999). Some accumulation zones in the Atlantic Sea and the Mediterranean Sea have very high debris densities despite being far from coasts. These densities relate to the consequence of large-scale residual ocean circulation patterns. There are higher densities in particular areas such as around rocks and wrecks or in depressions or channels (Galgani *et al.* 1996). In the North Sea (figure 4), accumulation of plastics 320 km offshore from Denmark (Galgani *et al.* 2000) is a consequence of several factors. These include the eddying circulation

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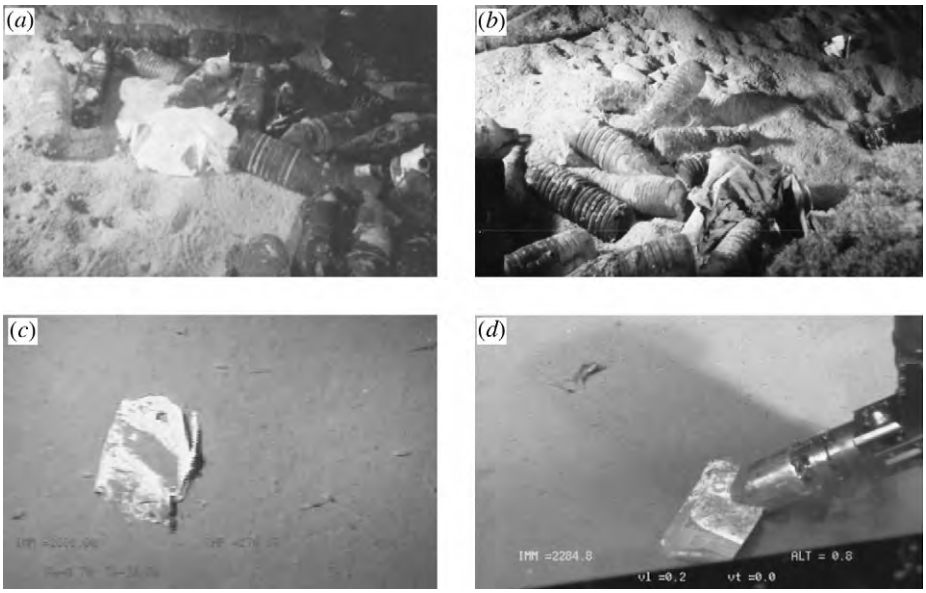

Figure 5. Accumulation of debris in deep sea environments. Submersible observations in Mediterranean canyons (*a* and *b*: plastic bottles at 1000 m depth at two different locations in the Marseille canyon, 43°03′00″ N, 05°00′00″ E) and above the polar circle, under ice floe (*c* and *d*: individual plastic bags, 2200–2600 m depth at Hausgarten, Fram strait, 79°03′80″ N, 04°11′60″ E).

in the central North Sea (Delhez & Martin 1992) and long-term circulation of water from the gulf stream transporting plastics northwards (Breton & Salomon 1995) and to the convergence zone of seabed sediment movements, owing to local decreases in turbidity and turbulence (Tappin *et al.* 1997).

Large rivers are responsible for substantial inputs of debris to the sea bed (Williams & Simmons 1997). They can transport waste out to sea because of their high flow rate and the strength of bottom currents. In smaller rivers, the displacement is slight, and waste can be found in zones adjacent to or in the estuaries and is often coincident with fronts (Acha *et al.* 2003). Patterns of debris transport should therefore be linked to river flow strength and may follow patterns similar to deposition of sediment load (often depositing only small amounts of material immediately along the coast).

Deep submarine extensions of coastal rivers also influence the distribution of seabed debris. In some areas, local water movements transport plastics away from the coast to accumulate in zones of high sedimentation. Under these conditions, the distal deltas of rivers can fan out in deeper waters, creating areas of high accumulation (Galgani *et al.* 1996). Continental shelves often have lower concentrations of debris since most of the anthropogenic debris in the outer shelf originates from coasts to shelves that are washed offshore by currents associated with river plumes. Data from the shelf areas off the River Rhone (Galgani *et al.* 1995*b*) and California (Moore & Allen 2000) show that circulation can be strongly, locally influenced by storm water events. The accumulation of plastics in coastal canyons may also be related to strong currents occurring in the upper part of canyons, which decrease rapidly in deeper areas resulting from increased confinement. Accordingly, debris distribution seems to be more temporally stable. An inevitable effect of this is the presence of greater amounts of debris in deeper shelf waters than in coastal waters (Galgani *et al.* 1996, 2000).

A wide variety of human activities contribute to these patterns of seabed debris distribution, including proximity to fishing activities, urban development and tourism. Also with plastic as a main component, debris from the fishing industry is prevalent in fishing areas (Kanehiro *et al.* 1995; Galgani *et al.* 2000). This type of material accounts for a high percentage of debris, for example up to 72 per cent in eastern China Sea (Lee *et al.* 2006) and 65 per cent in the Celtic sea (Galgani *et al.* 2000). Finally, fishing gear was also the dominant source of both plastic and overall debris in California (Moore & Allen 2000).

Investigations using submersibles at depths beyond the continental shelf usually consider the number of items per linear kilometre because of variability in transect width. They have revealed substantial quantities of debris (figure 5). Besides the high densities found in coastal canyons (up to 112 items per kilometre and 70% plastics), plastics and other anthropogenic debris were found widely dispersed at slope and abyssal depths (Galgani *et al.* 2000). Deployment of a remotely operated vehicle submarine in the Fram Strait (Arctic) (Galgani & Lecornu 2004) revealed 0.2–0.9 pieces of plastic per linear kilometre at Hausgarten (2500 m). On dives between 5500 and 6770 m, 15 items of debris were observed, of which 13 were plastic, probably carried there by the Norwegian current in the North Atlantic. At such latitude and bathymetry, there is negligible human activity, suggesting long-distance transport of debris. Even more than on the sea surface or strandlines of remote locations, such as in the Southern Ocean, accumulation trends in the deep sea are of special concern. Most polymers are highly persistent in the marine environment and only degrade slowly via

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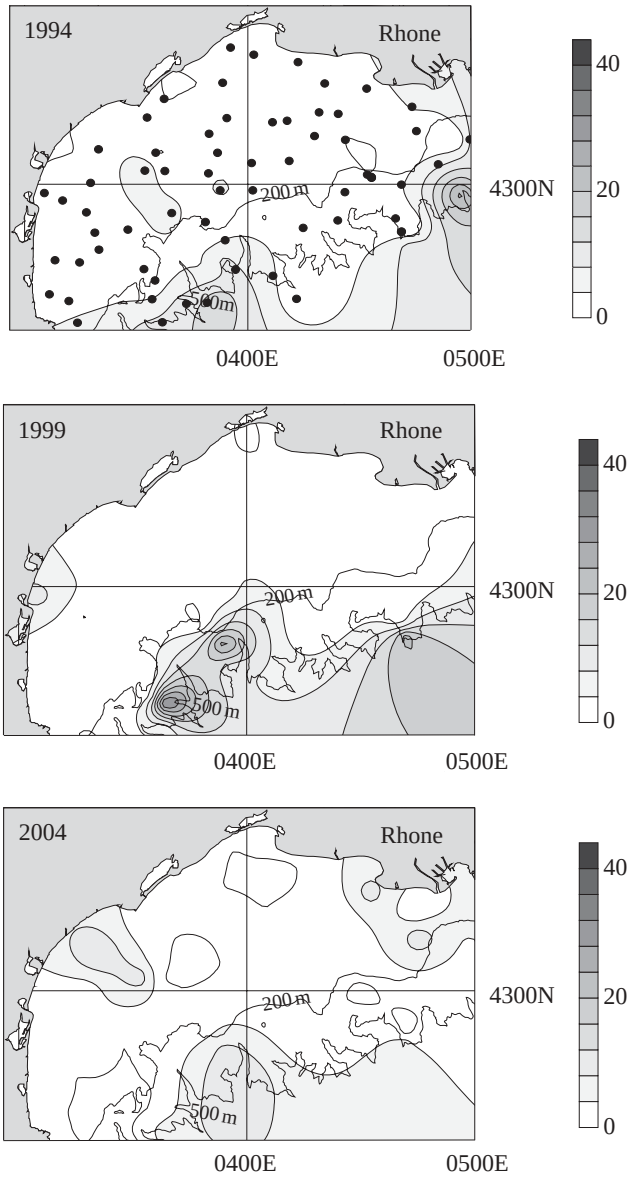

Figure 6. Plastic debris on the sea floor from the Gulf of Lion (Mediterranean Sea, France) between 1994 and 2004. Plastics were counted after 60 min trawl time (net = 16 m mouth, 10 mm mesh) at 65 stations (•) located on the continental shelf and adjacent canyons (down to 800 m) from the gulf. Results are given as items Ha^{-1} (10 000 m^2). Top plot shows the sampling stations from 1994–2004.

photo-catalysis when exposed to UV radiation (Andrady 2003). Estimates for the longevity of plastics are variable but are believed to be in the range of hundreds or even thousands of years depending on the physical and chemical properties of the polymer, but this is likely to be greatly increased at depth where oxygen concentrations are low and light is absent. We know little about trends in accumulation of debris in the deep sea as studies are rare, but the data we have indicate considerable variability. For example, in some areas, such as the Bay of Tokyo, debris densities decreased from 1996 to 2003 (Kanehiro *et al.* 1995; Kuriyama *et al.* 2003). In contrast, abundance remained stable in the Gulf of Lion, France during a similar period (figure 6). Furthermore in some areas around Greece, the abundance of debris at depth has increased over the last 8 years (Stefatos *et al.* 1999; Koutsodendris *et al.* 2008). Interpretation of temporal trends is also complicated by annual variations in debris transport, such as seasonal changes in flow rate of rivers. Other seasonal factors include variation in the position of water fronts, the intensity of currents, swell, winds and upwelling, which influence both the distribution and densities. Nevertheless, if we extrapolate from existing data, it would appear that in the Mediterranean Sea as a whole there are about 3×10^9 debris items (floating or sunk), of which 70–80% are plastic. New initiatives to minimize littering and to reduce, reuse and recycle plastic should ultimately reduce plastic input into the sea, although usage is still very high. However, fragmentation of macro- and mega-plastics to microplastic pieces will also contribute to future trends in the abundance of visible plastics.

4. FRAGMENTATION OF PLASTICS IN THE ENVIRONMENT

The longevity of plastics is a matter for some debate, and estimates range from hundreds to thousands of years. It is considered that (with the exception of materials that have been incinerated) all of the conventional plastic that has ever been introduced into the environment still remains to date unmineralized either as whole items or as fragments (Thompson *et al.* 2005). However, since we have only been mass-producing conventional plastics for around 60 years, it is too early to say exactly how long these materials will persist. Despite the durability of these polymers, plastic items are fragmenting in the environment as a consequence of prolonged exposure to UV light and physical abrasion (Colton *et al.* 1974; Gregory 1978; Andrady 2003; Thompson *et al.* 2004). This is particularly evident on shorelines where photo-degradation and abrasion through wave action make plastic items brittle, increasing their fragmentation.

Some of the first evidence of accumulation of plastic fragments in the environment came indirectly from examination of the gut contents of sea birds in the 1960s (e.g. Kenyon & Kridler 1969). Later, in the early 1970s, small fragments of plastic were observed in seawater collected with plankton samples from the North Sea (Buchanan 1971) and were subsequently reported on much broader scales in the northwestern Atlantic (Colton *et al.* 1974). There have since been numerous reports of fragments in the oceans, on the seabed and on shorelines worldwide (figure 7), and there is clear evidence that the abundance of these fragments is increasing (figure 8). The UK Marine Conservation Society, which organizes annual voluntary beach cleaning on shores all around the UK, reports a 30 per cent increase in the abundance of large fragments (1–50 cm in size) and a 20 per cent increase in the abundance of smaller fragments (<1 cm) between 1998 and 2006 (MCS 2007). On shorelines close to Plymouth, one of us (R. C. T.) recently recorded strandline material with >10 per cent (10.89 ± 0.67, mean ± s.d.) by weight of plastic fragments and pieces (including some pre-production plastic pellets, which are used to manufacture plastic products). In 2004, Thompson *et al.* (2004) reported

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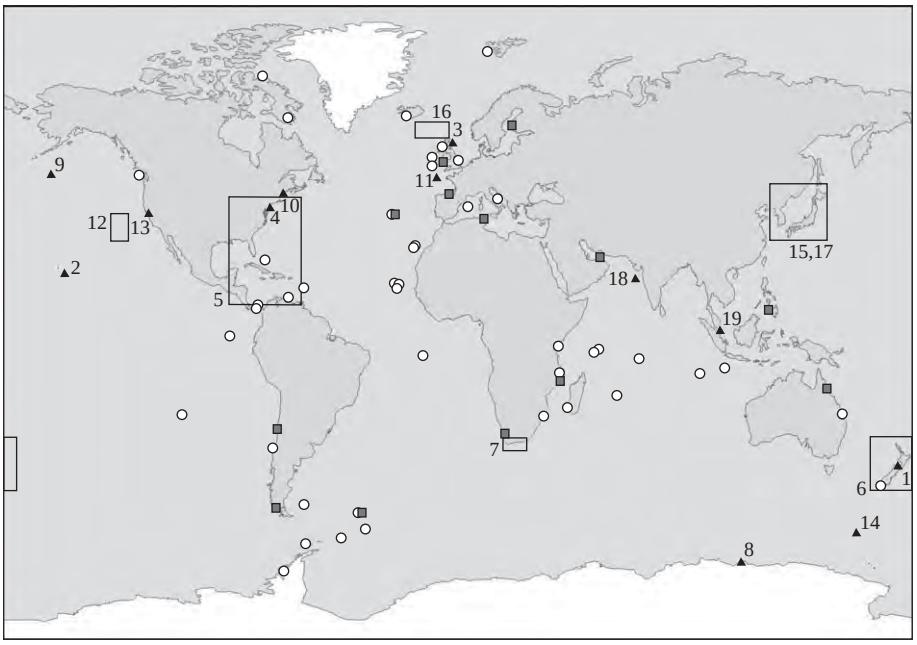

Figure 7. Reports of plastic fragments in the marine environment presented in chronological order: 1, Harper & Fowler (1987) report on plastic (mainly pre-production pellets) ingested by seabirds since 1960; 2, plastic fragments found in body cavity of dead laysan albatrosses during 1966 survey (Kenyon & Kridler 1969); 3, synthetic fibres in medium plankton net hauls (size not specified) (Buchanan 1971); 4, polystyrene spherules (average 500 μm) in coastal waters (Carpenter *et al.* 1972); 5, particles, spheres and discs (1–5 mm) in surface waters (Colton *et al.* 1974); 6, resin pellets (approx. 5 mm) on shoreline (Gregory 1978); 7, temporal trends in abundance and composition of plastic on beaches 1984–1989 (Ryan & Moloney 1990); 8, plastic particles (approx. 3 mm) in gut of storm petrels (van Franeker & Bell 1988); 9, fragments (≥500 μm) at sea surface (Shaw & Day 1994); 10, micro-plastic fibres (≥20 μm) in sewage sludge (Habib *et al.* 1996); 11, fragments in deep sea (size not specified) (Galgani *et al.* 2000); 12, fragments (≥350 μm) at sea surface (Moore *et al.* 2001*a*); 13, fragments and resin pellets on shoreline (size not specified) (Moore *et al.* 2001*b*); 14, fragments (≥1 mm) in scats of fur seals (Eriksson & Burton 2003); 15, fragments (≥1 mm) on beaches (Kusui & Noda 2003); 16, micro-plastics (≥20 μm) in surface waters and on beaches (Thompson *et al.* 2004); 17, resin pellets (approx. 5 mm) on beaches (Endo *et al.* 2005); 18, micro-plastics (≥10 μm) on shorelines near ship breaking yards (Reddy *et al.* 2006); 19, micro-plastics in surface waters and sediments (≥1.6 μm) (Ng & Obbard 2006). Red squares show distribution of micro-plastics (≥ 20 μm) in intertidal sediments (R. C. Thompson *et al.* 2003–2007 unpublished data). White dots show mega- and macro-plastic strandline surveys (Barnes 2002, 2005).

on the abundance of even smaller fragments of plastic, some just 20 μm, in diameter, which had accumulated on shorelines around the UK. Using plankton samples archived by the Sir Alistair Hardy Foundation for Ocean Science, it was evident that the abundance of this microscopic debris had increased significantly in recent years (figure 8). Similar fragments have since been identified from shorelines worldwide (figure 7), and in terms of numerical abundance, micro-plastic can constitute over 80 per cent of intertidal plastic debris at some locations (Browne *et al.* 2007).

Fragments of plastic can be identified using Fourier transform infrared (FT-IR) spectroscopy to match spectra obtained from unknown debris items to those of known polymers. Using this approach, a range of common polymers including polypropylene, polyethylene and polyester have been identified as fragments and microscopic fragments. These materials have a wide range of domestic and industrial uses from rope and packaging to clothing, and it seems likely that the fragments are forming from the breakdown of a wide range of everyday plastic products (Thompson *et al.* 2004). In addition to this 'natural' deterioration, it has been suggested that plastic items are also deliberately being shredded on board some ships in order that plastic waste can be concealed in food waste discharged at sea (van Franeker *et al.* 2004, 2005). The abundance of small items of plastic is further increased by the use of plastic particles as scrubbers and abrasives in commercial cleaning applications (Gregory 1996) and by spillage of pre-production plastic pellets (approx. 5 mm in diameter) and powders such as those used for rotomoulding (approx. 300 μm in diameter) (e.g. Carpenter *et al.* 1972; Colton *et al.* 1974; Gregory 1978). Hence, it is apparent that small items of plastic are entering the environment directly and that larger items of debris are fragmenting.

The accumulation of plastic fragments is of particular concern because they are difficult to remove from the environment and because they have the potential to be ingested by a much wider range of organisms than larger items of debris. Marine mammals, turtles and numerous other organisms are known to ingest large items of plastic including bags and bottles (Laist 1997; Derraik 2002). Smaller fragments can be ingested by birds, fish and invertebrates (Thompson *et al.* 2004; van Franeker *et al.* 2005). Upon ingestion, it is possible that these small fragments may present a physical hazard in a similar way to larger items of debris by clogging feeding appendages or the digestive system (Laist 1997; Derraik 2002). Microscopic fragments are also be taken up from the gut into other body tissues (Browne *et al.* 2008). In addition to concerns about the physical

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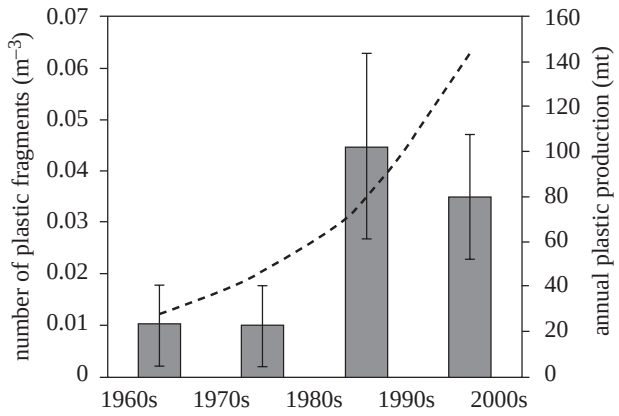

Figure 8. Microscopic plastic in surface waters, collected with continuous plankton recorder, revealed a significant increase in abundance when samples from the 1960s and 1970s were compared with the 1980s and 1990s ($F_{3,3} = 14.42$, $p < 0.05$). Global production of plastic overlain for comparison (APME 2006). Grey boxes, number of plastic fibres (m^{-3}); dashed line, plastic produced per year (million tonnes). (Reproduced with permission from Thompson *et al.* 2004.)

hazards presented by this debris, it has also been suggested that plastics could transfer harmful chemicals to living organisms (e.g. Oehlmann *et al.* 2009; Talsness *et al.* 2009; Koch & Calafat 2009). A range of chemicals are used as additives in the manufacture of plastics. These increase the functionality of the plastics, but some such as phthalate plasticizers and brominated flame retardants are potentially harmful and have been associated with carcinogenic and endocrine disrupting effects (Teuten *et al.* 2009). In seawater, plastics are also known to sorb and concentrate contaminants, which have arisen in the environment from other sources. These contaminants include persistent organic 'pollutants' such as polychlorinated biphenyls (PCBs), dichlorodiphenyldichloroethylene (DDE), nonylphenol and phenanthrene, which can become several orders of magnitude more concentrated on the surface of plastic debris than in the surrounding seawater (Mato *et al.* 2001). It has been widely suggested that these sorbed contaminants and the chemicals additives that are used in manufacture could subsequently be released if the plastics are ingested (Teuten *et al.* 2009). Small and microscopic plastic fragments present a likely route for the transfer of these chemicals because they have a much greater surface area to volume ratio than larger items of debris from which they have originated and because of their size they are available to a wide range of organisms, including deposit feeders such as the lug worm, *Arenicola marina*, that feed by stripping organic matter from particulates (Mayer *et al.* 1997; Voparil *et al.* 2004). Recent *in vitro* modelling studies predict that even very small quantities of micro-plastic have the potential to significantly increase the transport of phenanthrene to *A. marina* (Teuten *et al.* 2007) and work in this volume has examined the uptake of contaminants from plastics by birds (Teuten *et al.* 2009).

Given current levels of production and the quantities of plastic that are already present in the environment, it seems inevitable that the abundance of plastic fragments will continue to increase for the foreseeable future. More work is therefore needed to model the environmental consequences of this debris and to produce environmental risk assessment models to predict the transport of a range of contaminants by fragments of common polymers (Thompson *et al.* 2005; Thompson 2006; Teuten *et al.* 2007).

5. SUMMARY AND CONCLUSIONS

Less than 60 years ago, the mass production of plastics started and now most items that people use, virtually anywhere on the planet are partly or wholly made of this inexpensive, durable material. Plastics have transformed the surface of the planet, far beyond areas of human population density—fragments of all sizes are ubiquitous in soils to lake beds, from remote Antarctic island shores to tropical seabeds. Plastics turn up in bird nests, are worn by hermit crabs instead of shells and are present in turtle stomachs. Humans generate considerable amounts of waste and the quantities are increasing as standards of living and the population increase. Although quantities vary between countries, approximately 10 per cent of solid waste is plastic. Up to 80 per cent or sometimes more of the waste that accumulates on land, shorelines, the ocean surface or seabed is plastic. The most common items are plastic films, such as carrier bags, which are easily wind blown, as well as discarded fishing equipment and food and beverage packaging. Strandline surveys (beach-cleaning operations) are now organized in many countries and provide information about temporal and spatial trends. However, these surveys typically only provide data on coarse trends and larger items. There is considerable variation in methodology between regions and between investigators, and more valuable and comparable data could be obtained by standardizing monitoring approaches (Ryan *et al.* 2009). Accumulation rates vary widely with many factors such as proximity of urban settlements, shore use, prevailing wind and ocean currents and region. There were dramatic increases in quantities of mega- and macro-plastic debris in the northern hemisphere up to the 1990s. Quantities of debris in the oceans appear to have stabilized over the last decade but have increased on shorelines. However, this could indicate quantities of debris entering the sea are declining, but the material already in the sea is progressively being deposited on the shore or sinking to the deep. Accumulation rates are much lower in the Southern Hemisphere but are still increasing significantly, although repeat surveys on remote Antarctic islands and ocean areas suggest stabilization over the last decade. Fouled by organisms and sediment, plastics can sink and form an even higher proportion of human waste reaching the seabed, and quantities in excess of tens of thousands of items square kilometres have been reported. As on beaches and the ocean surface, enclosed seas such as the Mediterranean have the highest densities, but investigations in deeper waters have shown that high accumulation rates can stretch far (hundreds of kilometres) from the coast,

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particularly adjacent to large river mouths or in canyons. As in surface environments, trends of debris accumulation on the seabed increase at some locations, but are stable or decreasing at other sites. Quantities of debris in the oceans appear to have stabilized in the oceans over the last decade but have increased on shorelines. The problem of plastic fragments has taken on increased importance in the last few decades. From the first reports in the 1970s, it was only a few years before the widespread finding of plastic including reports of microscopic fragments (20 μm in diameter). The abundance of microscopic fragments was greater in the 1980s and 1990s than in previous decades. It has also been suggested that plastic waste is deliberately being shredded into fragments to conceal and discarded at sea. Plastics of all sizes are now reaching the most remote and deepest parts of the planet, and although we have much better knowledge of their sources, quantities and distribution, we still understand little about their longevity and affects on organisms. Further, we have made little progress in reducing the release of plastic to the environment (see discussion in Thompson *et al.* 2009). Temporal trends of macro-plastics on remote islands suggest that regulations to reduce dumping at sea have been successful to some extent. However, our sustained demand for plastic means that contamination of the environment by micro-plastic pieces seems set to increase. In addition, future sampling may reveal increasing quantities of debris in the planet's least known habitat, the deep sea.

The authors would like to thank past marine debris observers on beaches and ships who have generously given up their time and effort to recording items. The authors would also like to thank Alison Cook for help in preparation of figure 7; also Mark Brown and Stuart Niven for analysis of micro-plastic data in figure 7.

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Environmental implications of plastic debris in marine settings ––entanglement, ingestion, smothering, hangers-on, hitch-hiking and alien invasions

Murray R. Gregory

Phil. Trans. R. Soc. B 2009 **364**, 2013-2025
doi: 10.1098/rstb.2008.0265

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PHILOSOPHICAL TRANSACTIONS OF THE ROYAL SOCIETY B

Phil. Trans. R. Soc. B (2009) **364**, 2013–2025
doi:10.1098/rstb.2008.0265

Review

Environmental implications of plastic debris in marine settings—entanglement, ingestion, smothering, hangers-on, hitch-hiking and alien invasions

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Over the past five or six decades, contamination and pollution of the world's enclosed seas, coastal waters and the wider open oceans by plastics and other synthetic, non-biodegradable materials (generally known as 'marine debris') has been an ever-increasing phenomenon. The sources of these polluting materials are both land- and marine-based, their origins may be local or distant, and the environmental consequences are many and varied. The more widely recognized problems are typically associated with entanglement, ingestion, suffocation and general debilitation, and are often related to stranding events and public perception. Among the less frequently recognized and recorded problems are global hazards to shipping, fisheries and other maritime activities. Today, there are rapidly developing research interests in the biota attracted to freely floating (i.e. pelagic) marine debris, commonly known as 'hangers-on and hitch-hikers' as well as material sinking to the sea floor despite being buoyant. Dispersal of aggressive alien and invasive species by these mechanisms leads one to reflect on the possibilities that ensuing invasions could endanger sensitive, or at-risk coastal environments (both marine and terrestrial) far from their native habitats.

Keywords: pelagic plastics; marine debris; entanglement and ingestion; hitch-hiking; alien invasions

1. INTRODUCTION

The environmental and other problems arising from indiscriminate disposal of plastics and other persistent synthetic materials (marine debris) into the global oceans and seas are chronic in nature rather than acute, and are long-recognized international problems (e.g. Mattlin & Cawthorn 1986; Thompson *et al.* 2009*b*). The endangering impacts of these materials on marine environments are many and are succinctly reviewed by Derraik (2002). These undesirable contaminants may have either land- or marine-based sources, although the latter is generally considered to be the more significant. Management, and preferably prevention, or at least reducing the problems created by marine debris are difficult to address. Available evidence suggests that the quantities involved are ever increasing and hence so is the magnitude of the resulting problems (see Barnes *et al.* 2009; Ryan *et al.* 2009). It has recently been estimated that the 1982 report of 8 million marine debris items entering the world's oceans and seas each day now needs to be updated by being multiplied several fold (Barnes 2005). Even the most remote of localities of both Northern and Southern hemispheres are no longer immune from littering by marine debris: e.g. Antarctica and sub-Antarctic Islands of the Southern Ocean (Gregory *et al.* 1984; Eriksson & Burton 2003; Barnes & Milner 2005); North Pacific gyre (Moore 2003; Ebbesmeyer *et al.* 2007) and South Pacific Islands (Gregory 1999*a*). Nevertheless, and in contrast to the above comments, censuses at crudely 10 year intervals (mid-1970s, 1980s, 1990s, and mid-2000s in progress) of virgin plastic granules (pellets or nibs) suggest the quantities are slowly and steadily, but somewhat irregularly, declining on the shores of New Zealand, eastern Canada and possibly Bermuda (M. R. Gregory, unpublished). This may be a reflection of changes in handling and transport procedures rather than conscientious or focused efforts addressing the problem. Similar decreases in the composition of plastic litter in surface waters of the Atlantic and southwestern Indian oceans, and reductions in amounts ingested by several seabirds, have been reported recently (Ryan 2008; Ryan *et al.* 2009).

Many of the problems associated with marine debris attract considerable media and public attention. Foremost of these are the visual affront of unsightly discarded plastic and aesthetic values in general (figure 1). There are also tourist perceptions and emotive issues arising from widely published images of seabirds, marine mammals and fish entangled in abandoned or lost netting; furthermore, entanglement

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Figure 1. Debris (mainly plastic) collected during an annual beach clean at Mason Bay, South Island, New Zealand.

(figure 2) and ingestion (figure 3) may lead to death from starvation and debilitation, with a reduced quality of life and lowered reproductive performance (Laist 1987). Other impacts to receive limited attention are of no less importance, e.g. damage to subsistence fisheries (Nash 1992); hazards to recreational boating and larger commercial vessels; impact of plastic sheeting that blankets the biota of soft sediment, reef and rocky substrata (Uneputty & Evans 1997) as well as anoxia and hypoxia induced by inhibition of gas exchange between pore waters and overlying sea water (Goldberg 1997; Gregory & Andrady 2003).

2. AESTHETIC VALUES, ENTANGLEMENT, INGESTION, SMOTHERING, GHOST FISHING, THE WRACK AND BEACH CLEANING

(a) *Aesthetic values*

Many of the litter problems associated with marine debris attract considerable media and public attention. Foremost of these is the visual affront of unsightly, discarded and/or accidentally lost plastic and other manufactured materials that tend to strand and concentrate along shorelines and sandy beaches (figure 1)—ones often of considerable recreational importance. There are also strongly emotive issues associated with both local beach users and tourist perceptions. Financial concerns over visitor numbers may also be a significant factor. Harshly critical public responses are common and may reflect personal observations or widely published and often harrowing images of seabirds, marine mammals and fish entangled in abandoned and beach-cast or lost netting (figure 2). Terrestrial vertebrates may also be snared or trapped in wrack debris. Where tidal range is moderate, and particularly during periods of consecutive spring high tides, unsightly littering material may be buried and hidden from view. Exhumation of litter may occur during later periods of higher wave activity (Williams & Tudor 2001*a*) and may also be cyclic in character. In addition to those factors identified previously (above) and later (below), concerns are commonly expressed about economic losses, health issues and harm to local biota, and otherwise general impressions of longer term deterioration in beach aesthetic values (e.g. Gabrielides 1995).

(b) *Entanglement and ghost fishing*

Laist (1997) has compiled a comprehensive list of marine species known to be impacted by entanglement (E) and ingestion (I). He identifies over 250 species, how impacted (E or I), the material involved, as well as location and source. The identified taxa include: turtles; penguins; albatrosses, petrels and shearwaters; shorebirds, skuas, gulls and auks; coastal birds other than seabirds; baleen whales, toothed whales and dolphins; earless or true seals, sea lions and fur seals; manatees and dugong; sea otters; fish and crustaceans.

Prior to the 1950s, rope and cordage used in all marine activities, including fisheries, was made of natural fibres—typically Indian or Manila hemp and cotton, and it was often strengthened with a coating of tar or strips of worn canvas. These materials lose their resilience in usage and if lost or discarded at sea tend to disintegrate quickly. For reasons summarized and simplified in table 1, over the past 50+ years these natural fibres have been replaced by nylon and other synthetic materials that are generally buoyant and far more endurable. The very properties that humankind find so desirable in plastic materials are also those responsible for the plethora of problems they are creating (globally) in marine environments.

Many marine animals (sea turtles, mammals, seabirds, fish and crustaceans) are either drawn to or accidentally entangled in netting, rope and monofilament lines that have their sources in discards and losses from commercial fishing activities. Set and drift nets are particularly hazardous. Many animals, if not most so caught, find it difficult to escape entanglement and are doomed to drown or die from injury, starvation and general debilitation. There are numerous reports of packing loops (cut and uncut) attracting the interest of seals and sea lions (e.g. Hanni & Pyle 2000, Southeast Farallon Island, California; Henderson 2001, Monk seals, Northwestern Hawaiian Islands; Page *et al.* 2004, Australian sea lions and New Zealand fur seals, Kangaroo Island, Australia; Boren *et al.* 2006, New Zealand fur seals, Kaikoura, NZ; Hofmeyr *et al.* 2006, Antarctic fur seals, Bouvetøya Island). Sharks also are often caught by 'debris collars' (e.g. Sazima *et al.* 2002). Recorded changing rates of entanglement in these studies are difficult to decipher, but it is evident that with intervention, individuals with severe wounds have a good chance of survival (see Page *et al.* 2004; Boren *et al.* 2006). Plastic packing loops may tighten and cut into flesh as animals grow, creating 'lethal necklaces' (figure 2*b*) ultimately leading to strangulation. Carelessly discarded plastic six-pack carriers may similarly capture fish and other wildlife; paperboard is not so endangering (Thompson & Côté 1997). Other biologically harmful factors can include suppurating skin lesions, ulcerating body wounds, interruption of feeding activity and failed predator avoidance (Gregory 1991).

In recent years, sightings have regularly been made of pods of the endangered humpback whales travelling northwards along the east coast of the South Island of New Zealand and on their annual passage between Antarctic waters and tropical waters to the north. Over the last 7 or 8 years at least seven whales have had in tow a mass of tangled rope and other debris.

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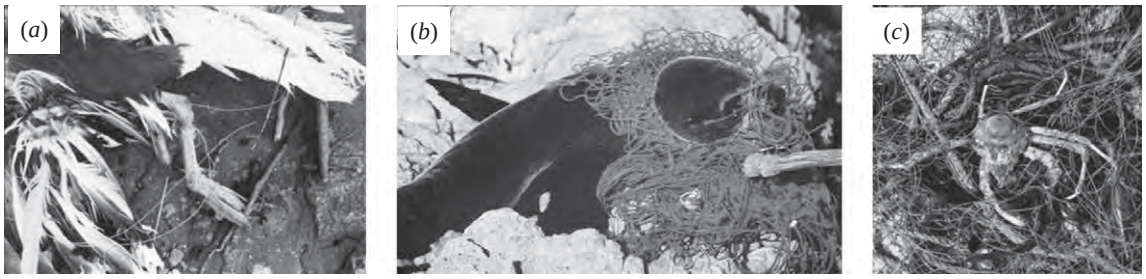

Figure 2. Examples of entanglement from New Zealand that draw immediate public sympathy and anger: (*a*) Karoro (southern black-backed gull, *Larus dominicanus*) caught and hooked in nylon filament fishing line; (*b*) a New Zealand fur seal trapped in discarded netting and (*c*) Ghost fishing—derelict fishing gear dredged from >100 m on the Otago shelf.

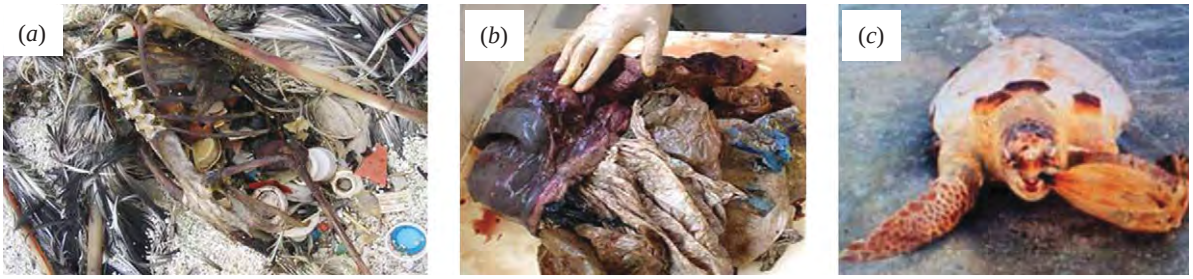

Figure 3. Examples of ingestion: (*a*) Laysan Albatross (*Phoebastria immutabilis*, at Kure Atoll, Courtesy of AMRF); (*b*) plastic from the stomach of a young Minke whale (*Balaenoptera acutorostrata)* that had been washed ashore dead in France (Courtesy of G. Mauger & F. Kerleau, Groupe d'Études de Cétacés du Cotentin GECC) and (*c*) stranded sea turtle disgorging an inflated plastic bag. One infers that it has been mistaken for an edible jellyfish (medusoid).

Table 1. Summary of factors complicating and compromising analyses of marine entanglement (taken and adapted from Laist, D.W. 1996, p. 106). (Entanglement records are biased towards shoreline surveys. They may remain unpublished and/or be anecdotal in character: local and regional, as well as geographic and temporal comparisons will be difficult to evaluate.)

detection and discovery	sampling and reporting biases
entanglements are isolated events scattered over wide areas	limited at-sea sampling and few long-term surveys
entangling debris often difficult to identify on active animals at sea; readily recognized when stranded	inconsistent sampling methodologies; strandings are an unknown portion of local and regional entanglements
dead animals difficult to see if floating just below sea surface and if concealed within matted debris mass	shore counts of live entangled animals are biased towards survivors sporting minor amounts of debris
entangled dead animals may disappear from view quickly through sinking and/or predation	some entanglements may involve interactions with active rather than derelict fishing gear

In at least two instances the mass has included a crayfish (i.e. lobster) pot and a buoy with marker pole and flag. Attempts to free entanglements were successful in at least one instance and failed in others (D. Hayes, personal communication).

Tangled masses of relatively intact, but lost and abandoned or derelict, trawl net, gillnet, webbing and monofilament line can retain the ability to continue to capture target fish and other species for lengthy periods of time (Laist 1996; Carr & Harris 1997). This may lead to ghost fishing, with conspicuous mortality and catch losses. Comparable waste problems were associated with 'drift-net' fisheries in the South Pacific in the 1980s (Wright & Doulman 1991; Richards 1994). Ghost-net fishing is not restricted to surface or shallow waters. Over the past 20+ years, important seamount fisheries have developed around New Zealand and Tasmania and are known from elsewhere around the region (e.g. Koslow 1997; de Forges *et al.* 2000; O'Driscoll & Clark 2005). It is widely accepted that seamounts are fragile habitats. Trawl gear is today being deployed across steeply irregular, and often boulder-strewn, sea floor surfaces at depths typically lying between 500 and 1000/2000 m. Netting caught during passage across the seabed can cause considerable damage to seabed environments (e.g. deep water coral reefs), and if not recovered may remain there, out of sight, and continue ghost fishing almost indefinitely. The potential magnitude of disturbance to seabed environments can be likened to 'forest clear cutting' (Watling & Norse 1998).

(c) *Ingestion*

The literature on ingestion (and entanglement) of plastic items in marine debris is voluminous and often repetitive, and the widely reported environmental problems identified are global in character. These include: wounds (internal and external), suppurating skin lesions and ulcerating sores; blockage

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Table 2. Occurrence of plastic pellets in five prion species collected from New Zealand beaches, between 1958 and 1977 (from Harper & Fowler 1987; p. 66).

species	age class		gizzards examined	number with plastic pellets		% total pellets		species status
	Imm	adult		Imm	adult	Imm	adult	
broad-billed prion *Pachyptila vittata*	170	140	310	18	33	10.6	23.6	NZ resident
Salvin's prion *Pachyptila salvani*	651	12	663	133	0	20.4	0	Indian Ocean migrant
Antarctic prion *Pachyptila desolata*	29	6	35	4	1	13.8	16.7	NZ sub-Antarctic migrant
thin-billed prion *Pachyptila belcheri*	147	5	152	10	0	6.8	0	Indian Ocean migrant
fairy prion *Pachyptila turtur*	714	105	819	88	13	12.4	2.4	NZ resident
totals	1711	266	1979	253	47			

Imm, immature, birds of the year; adults, all others. Age determined by bone ossification, gonad condition, bill and feet shrinkage, measurements, plumage.

of digestive tract followed by satiation, starvation and general debilitation often leading to death; reduction in quality of life and reproductive capacity; drowning and limited predator avoidance; impairment of feeding capacity; and the possibility that plastic resin pellets may adsorb and concentrate potentially damaging toxic compounds from sea water (e.g. Gregory 1978, 1991; Laist 1997; Mato *et al.* 2001; see also the discussions in Oehlmann *et al.* 2009; Teuten *et al.* 2009).

Over 100 species of seabirds are known to ingest plastic artefacts and/or become entangled with them (Laist 1997). First local New Zealand recognition of high virgin plastic pellet concentrations on Auckland City beaches was made in the astral summer of 1971–1972 (Gregory 1977). Subsequent observations on remote beaches north of Auckland over the astral summers of 1972/1973/1974 and 1975 and examination of wrack along strandlines revealed a surprising abundance of plastic pellets and other marine debris. Observers quickly became aware of a developing environmental problem. Recording pellet quantities was a diversion during coastal studies along the extensive sandy beaches and dune fields of northern New Zealand. In subsequent years, 'plastic pellet' expeditions were made to beaches around the rest of the country (Gregory 1978). Occasionally, one came across beach-cast birds and attention was drawn to plastic pellets associated with disintegrating carcasses and also entangling monofilament line, often with attached fishhook. Over a 21-year period (1958–1977), observations were made of five prion species (*Pachyptila*) cast ashore on exposed beaches near Wellington (southern North Island). Gizzards and proventriculi were removed and examined. Harper & Fowler (1987) noted that the lightest birds carried the most pellets and concluded that the proportion of starved beach-cast prions suggested these birds would eat anything resembling food before they died. They also suggested that prions began ingesting plastic pellets by the early 1960s, and an accompanying graphical presentation shows irregular but rapid increases in the percentages of plastic carried in three prion species which grew significantly (from <5 to 25%) between 1960 and 1977 (Harper & Fowler 1987; table 2).

Plastic materials of varying kinds had spread to all oceans and adjacent seas by the late 1970s or early 1980s and wide concern was being expressed over the amounts of cylindrical, virgin plastic pellets that are industrial feedstock, together with fragmented plastic particles of varying size and shape that were being ingested by pelagic seabirds (e.g. Shomura & Yoshida 1985). Over the past four or five decades, there have been numerous accounts of marine debris ingestion by a great variety of seabirds (see Appendix 2 in Laist 1997). Some representative examples typifying the global spread of plastic ingestion behaviour include red phalaropes (Connors & Smith 1982); 15 species of sea birds, Gough Island, South Atlantic Ocean (Furness 1985); Wilsons storm-petrels, Antarctica (Van Franeker & Bell 1988); storm-petrels, etc. (Blight & Burger 1997); short-tailed shearwaters, Bering Sea (Vlietstra & Parga 2002); southern giant petrels, Southern Atlantic Ocean (Copello & Quintana 2003); northern fulmars, Nunavut, Davis Strait (Mallory

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et al. 2006). Cadee (2002) has drawn attention to conspicuous bird pecking marks (possibly made by Northern Fulmars) in cuttlebones cast ashore on the Dutch coast near Texel. It was also noted that similar peck marks were common locally on beach-cast styrofoam and spongy plastic and it was suggested that fulmars were mistaking plastic artefacts for cuttlebone.

As well as being entangled in discarded fishing gear, many marine vertebrate species have a record of regularly ingesting discarded plastic materials (see Laist 1996; Appendices 1 and 3). Several, if not most, sea turtle species are seriously threatened by 'feeding on' plastic and other marine debris (e.g. Hawaiian Islands, Balazs 1985; coastal Florida, Bjorndal *et al.* 1994; Western Mediterranean, Tomás *et al.* 2002; Paraíba, Brazil, Mascarenhas *et al.* 2004). Particular hazards are discarded and semi-inflated, floating plastic bags that are often mistaken for jelly fish (medusoids), which block the oesophagus (figure 3*c*). Manatee also have felt the undesirable impact of marine debris (e.g. Florida, Beck & Barros 1991). An unusual accumulation of small plastic particles recovered in the scats of fur seals from Macquarie Island has been recorded by Eriksson & Burton (2003). These were small, often angular in shape and buoyant, with surface striations, and could not be related to plastic pellet feedstock. It is suggested that the breakdown of larger, user plastic fragments was a response to being washed ashore and ground down by abrasion on high energy cobble beaches. Eriksson & Burton (2003, p. 380) furthermore hypothesized that the plastic particles were initially washed out to sea, before being size-selected and consumed by pelagic fish, and that the latter were the prey of fur seals.

(d) *Smothering*

Most plastic materials entering the marine environment are buoyant and float on the sea surface. It is therefore perhaps surprising to find that there are numerous reports of sunken marine debris of all kinds settling to the sea floor at all depths—from inter-tidal to abyssal environments; e.g. the Skagerrak (Hollström 1975); Tokyo Bay, Japan (Kanehiro *et al.* 1995); tidal flats, Ambon Bay, Indonesia (Uneputty & Evans 1997); Bristol Channel (Williams & Simmons 1997); European and Mediterranean waters (Galil *et al.* 1995; Stefatos *et al.* 1999; Galgani *et al.* 2000); Kodiak Island, Alaska (Hess *et al.* 1999); southern California Bight (Moore & Allen 2000); Hauraki Gulf, New Zealand (Backhurst & Cole 2000); Saronikos Gulf, Greece (Katsanevakis *et al.* 2007). Once these items reach the sea floor, particularly in deeper and still waters, they are doomed to a slow and yet permanent entombment.

Several authorities now consider that the sea floor is the ultimate sink for much marine debris (e.g. Williams *et al.* 1993; Goldberg 1997). The mechanisms by which these materials may reach the deep sea floor are poorly understood. Land-sourced materials are common on canyon floors of the western Mediterranean Sea. These can be tracked from the coast in their progressive passage to abyssal depths and at considerable distance offshore (Galgani *et al.* 2000). The pattern is strongly suggestive of rapid transport through near-shore zones and entrainment in bottom hugging currents (Williams *et al.* 2005). There is also evidence from the Rio de la Plata that bottom salinity fronts in estuarine environments may act as debris-accumulating barriers (Acha *et al.* 2003), similar to those associated with surface waters along convergence zones, oceanic fronts and eddies (e.g. Gregory 1999*b*). Furthermore, rapid and heavy fouling of floating plastic (and other objects) may so increase density that they sink to the sea floor. However, grazing organisms may episodically clean fouled surfaces leading to yo-yoing periods of submergence and resurfacing until permanent settlement to the sea floor occurs (Ye & Andrady 1991).

Sediment settling on pelagic plastic materials may also take them to the sea floor. Observations made in shallow, near-shore waters, by Backhurst & Cole (2000) and Katsanevakis *et al.* (2007), have confirmed that once there, gradual changes may occur in community structure and that the environment can no longer be considered pristine. Goldberg (1997) has suggested that the blanketing effects of plastic sheeting on the sea floor could lead to anoxia and hypoxia induced by inhibition of gas exchange between pore water and sea water. Furthermore, sediment settling on pelagic plastic materials and taking them to the sea floor can lead to the creation of artificial hardgrounds (e.g. Harms 1990). Following in a somewhat similar vein, Williams *et al.* (2005, p. 627), perhaps with a degree of irony, have claimed that benthic marine debris once settled on the sea floor could '… enhance or enrich local biodiversity in the short term, for in the long term it is doomed to permanent interment in a slowly accumulating sediment cover'. An interesting and disturbing aside that relates to settling rates of plastic items is Oshima's (2000, p. 73) report of numerous white plastic shopping bags suspended upside down and freely drifting in the ocean at water depths of 2000 m—and looking like an assembly of ghosts.

(e) *The wrack and beach cleaning*

Natural flotsam, of both marine and terrestrial origin (seaweeds and plants) together with jetsam of indeterminate sources, tends to accumulate along high-tide strandlines, where it is commonly known as 'the wrack'. These areas are often ephemeral, dynamic and seasonal environments and also tend to accumulate significant quantities of manufactured materials, in particular those made of plastic and other non-destructibles. As a consequence, wrack environments are commonly unsightly and the demands of local authorities to 'clean up the mess' are frequent and can be expensive (e.g. Ryan & Swanepoel 1996; Ballance *et al.* 2000). Often, and increasingly, the demands are for mechanical and complete removal of the strandline and any debris that is concentrated there. Llewellyn & Shackley (1996) demonstrated that a consequence of this may be the destruction of ecologically significant habitats. These habitats support rich and diverse marginal marine-to-terrestrial invertebrate biota and may also be visited by vertebrates, mostly birds—in New Zealand, and in many oceanic islands, it may typically be birds (and rats) but elsewhere it may include a number of scavenging small mammals (e.g. Llewellyn & Shackley 1996).

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Many local and managing authorities appear to accept blindly that damage from mechanical beach cleansing is cosmetic in character and that the strandline readily returns to its natural state. However, a recent and limited, small-scale cleaning experiment has concluded that while the near-surface meiofauna can quickly recover, repeated cleanings or deeper excavations '... may certainly result in much slower recolonization rates' (Gheskiere *et al.* 2006). Commonly held opinions suggest beach clean-ups are short sighted, and a temporary cure at best, although with some educational values (Williams & Tudor 2001*b*). In part, the problems are being addressed through local activities of the Marine Conservation Society (a United Kingdom charity) and the European Blue Flag Scheme of beach evaluations and awards (e.g. Williams & Morgan 1995; Tudor & Williams 2006). A New Zealand example of problems with marine debris is informative (Gregory 1999*b*). At almost 47°S, Stewart Island's Mason Bay is a spectacular, remote and isolated, *c.*10 km sandy beach that is open to the Southern Ocean and also faces into the Roaring Forties. The immediately close and offshore waters are intensely fished. The beach has been heavily fouled with marine debris dominated by fisheries-related items, most of which were from New Zealand sources. A minor, but significant, component came from Korea and Japan; rarer sources included Argentina, Australia, Belgium, Chile, France, Norway, Poland, Russia, Spain, South Africa and United Kingdom. Annual clean-ups have been organized since 1989 and it has been estimated that some 2–3 tonnes of debris was cleared each year. Disposal of the vast quantities collected is difficult (figure 1). After the 1989 exercise, a pyre was built on the beach and set alight with the aid of diesel and driftwood—this reduced the bulk to a quarter. Clinker and burnt remains were removed and placed in a pit set in dunes behind the beach. Since that time the collected marine debris has been placed at designated sites behind the fore-dunes. Local scarfing of fore-dunes has exposed once-buried plastic and other marine debris at several places. Strong on-shore winds blow shredded plastic bags and sheeting far inland to unsightly adorn and blanket the sparse coastal vegetation and may also be a contributor to environmental degradation of dune fields behind the beach. While burial may remove the unsightly debris from view in coastal settings of this kind, it cannot be considered a cure—in many instances it is at best a palliative.

3. HANGERS-ON AND HITCH-HIKING ALIENS—INVASIVE SPECIES

For untold millennia, floating, terrestrial plant matter, whether large and solitary tree trunks or smaller shrubs and stems with soil still attached, as well as matted masses of these materials, have freely voyaged, traversed and dispersed across the open oceans just as 'sea beans' (see Gunn & Dennis 1999, p. 3); logs, pumice and other natural flotsam continue so to do to this day. 'Floating Islands' with cargoes that include exotic plants and vertebrate animals have been recognized since medieval times (see Van Duzer 2004). Through the distant past to modern times, these materials have also attracted a diverse biota of sessile and motile marine organisms—freedom travellers (hitch-hikers and hangers-on if one likes). This process has been a mechanism in the slow trans-oceanic dispersal of marine and some terrestrial organisms; e.g. Wheeler's (1916) report of ants carried in a floating log from the mainland of Brazil to offshore San Sebastian Island. Similarly, Ingólfsson (2000) has demonstrated that rafting on floating clumps of seaweed around Iceland may see inter-tidal species dispersed for considerable distances offshore. The hard surfaces of pelagic plastics provide an attractive and alternative substrate for a number of opportunistic colonizers. With the quantities of these synthetic and non-biodegradable materials in marine debris increasing manifold over the last five decades, dispersal will be accelerated and prospects for invasions by alien and possibly aggressive invasive species could be enhanced (e.g. Gregory 1978, 2004; Winston *et al.* 1997; Barnes 2002*a*,*b*) (examples illustrating some of the possibilities are provided in appendix A).

Pelagic plastic items are commonly colonized by a diversity of encrusting and fouling epibionts (e.g. figure 4). Most of these are sessile, hard-shelled or crustose organisms and dominated by bryozoans. Also included are barnacles, tube worms, foraminifera, coralline algae, hydroids and bivalve molluscs. Furthermore, they are also attractive substrates for a varied motile biota. The pseudo-planktic community that develops is comparable to that associated with *Sargassum* and other drifting seaweed, although with reduced species richness and diversity (Stevens *et al.* 1996; Winston *et al.* 1997). Flexible rope may also attract hangers-on (figure 4; see appendix A(ii)). Aggregations of marine debris can provide habitats suiting the larval and juvenile stages of numerous marine organisms. They may also attract free-living, ocean-roaming predators that often gather under fish aggregating devices, and where others simply sought a protective haven (see Winston *et al.* 1997, fig. 7.10). Aspects of floating substrata and colonizing biota are comprehensively reviewed in Theil & Gutow (2004).

4. NEW ZEALAND AND THE PACIFIC SECTOR OF THE SOUTHERN OCEAN

In the Southern Hemisphere, a latitudinal gradient has been recognized in the extent to which drifting plastics are colonized by epibionts. Surface cover, particularly by bryozoans, as well as species richness and diversity is greatest at low latitudes (tropical and subtropical), decreasing through temperate mid latitudes and least in high (polar) latitudes (Barnes & Sanderson 2000; Barnes *et al.* 2006). Across the South Pacific Ocean and the Antarctic sector of the Southern Ocean, there are several important oceanic fronts along which marine debris tends to concentrate, e.g. Humbolt Front off Valparaiso, Chile (Bourne & Clark 1984). The southwards flowing East Australian Current rises in the Coral Sea. Eddies from it swing eastwards across Tasman Sea periodically bringing exotic tropical 'sea beans' to the shores of Aupouri Peninsula (northern New Zealand). These eddies also carry significant quantities of marine debris that

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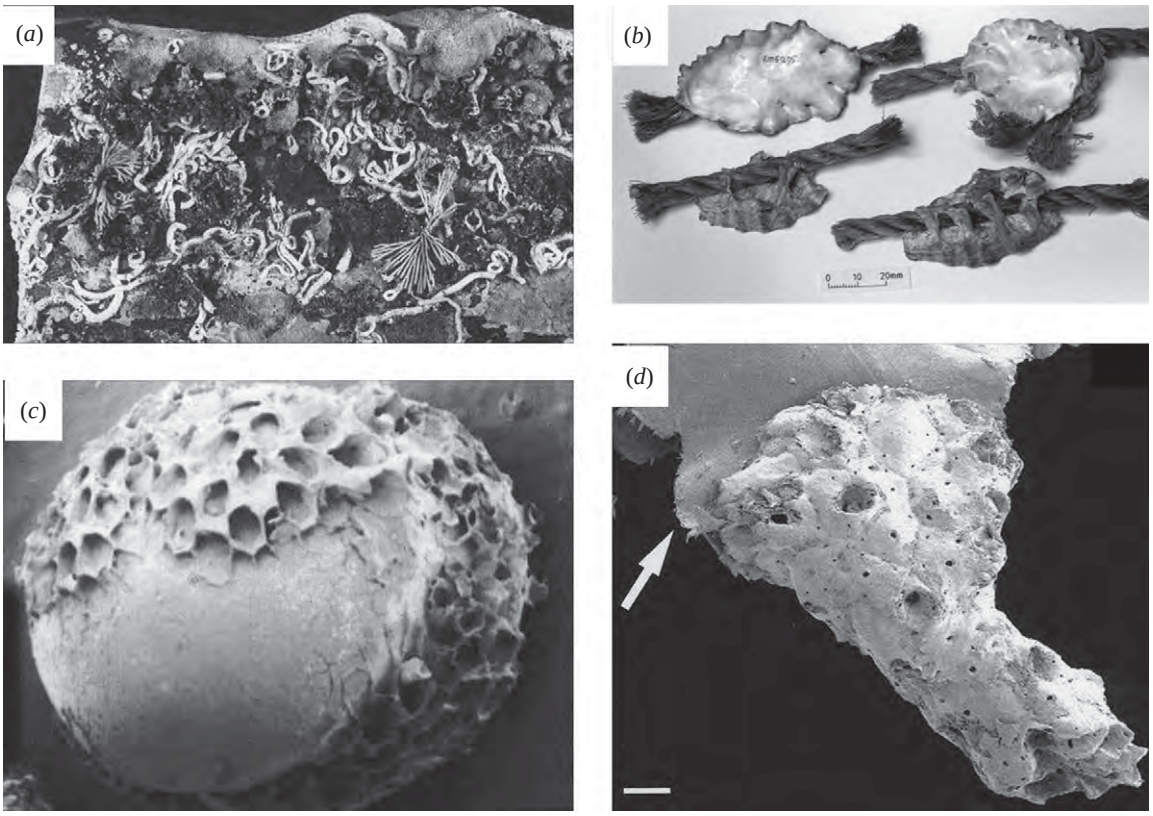

Figure 4. Example of colonization and encrustation on plastic debris from the New Zealand coastline: (*a*) heavy and varied colonization of a plastic slab recovered (note the hard bodied encrustations and soft fleshy epibionts; (*b*) cuttings from a tangled mass of synthetic rope, carrying a cargo of the warm-water Indo-Pacific oyster, *Lopha cristagalli*, a species that is alien to New Zealand waters (appendix A(ii)); (*c*) plastic pellet (raw material for manufacture of plastic products) encrusted by the bryozoan *Membranipora taberculata*, see appendix A(i); (*d*) small bryozoan colony (*Galeopsis mimicus*) attached to a frayed plastic flake (arrowed) recovered from a depth of 393 m off the east coast shelf off the South Island (appendix A(xii)); scale bar 200 μm. Recently a tropical hermatypic coral has also been reported on a remote South Island shoreline (J. Lindqvist, personal communication).

originates from eastern Australia. The sources may be land-based or fisheries and other maritime activities and the cargo carried may include taxa alien to New Zealand.

Over 150 marine species are known from plastic debris stranding on the shores of northern New Zealand or as colonizers in experiments with moored plastic bottles suspended at from the surface at varying depths to 10 m (L. M. Stevens 1992, unpublished data). Most of the identified biota are hard-shelled or crustose organisms and are dominated by bryozoans (Stevens *et al.* in preparation). Around northernmost New Zealand, at least 60 bryozoan species have been identified. Of these, 28 had not been recorded previously—at first glance this suggests recent introductions. In truth it reflects lack of local research, as most of these taxa are known from eastern Australia and the Kermadec Ridge to the northeast of New Zealand (Gregory 1998). The cosmopolitan, warm-water, low-latitude bryozoan, *Membranipora tuberculata* (figure 4*c*) that now dominates beach-cast plastic items around northern New Zealand is a relatively recent arrival—perhaps from eastern Australia (Gregory 1978, 1998). The biota recorded from beached marine debris are strongly biased towards those taxa with hard and resistant parts. Recovery of freely drifting items in open waters as well as study over a nine-month period of moored panels has revealed the importance of colonization by a soft fleshy biota. This included a representative suite of well-known, northern New Zealand marine biota (e.g. brown and filamentous algae, hydroids, ascidians, sea anemones and sponges as well as motile organisms including crabs, amphipods, isopods, errant polychaetes, gastropods, limpets, chitons, echinoderms, sea slugs and sea cucumbers) (Winston *et al.* 1997; Stevens *et al.* in preparation). Soft and fleshy organisms disintegrate rapidly once out of water and left stranded and exposed to the harshness of beach environments. They are seldom recorded in beach surveys. Ye & Andrady (1991) have also recognized the importance of an adhering soft and fleshy biota.

Weakening eddies from the East Australian Current pass down the east coast of northern New Zealand, and off East Cape merge into the Subtropical Convergence zone. Remote Chatham Island, lying 850 km to the east of mainland New Zealand, sits virtually astride this zone. Marine debris is abundant on the island's north- and west-facing shores (Gregory 1999*b*). Much of this comes from the local fishery and is generally clean of any attached biota. Nevertheless, some debris items support a varied suite of hitch-hikers and hangers-on. The degradation and weathering state of these materials as well as labelling suggests that these items have been afloat for some time and that they may have come from afar. Virgin plastic nibs, and for

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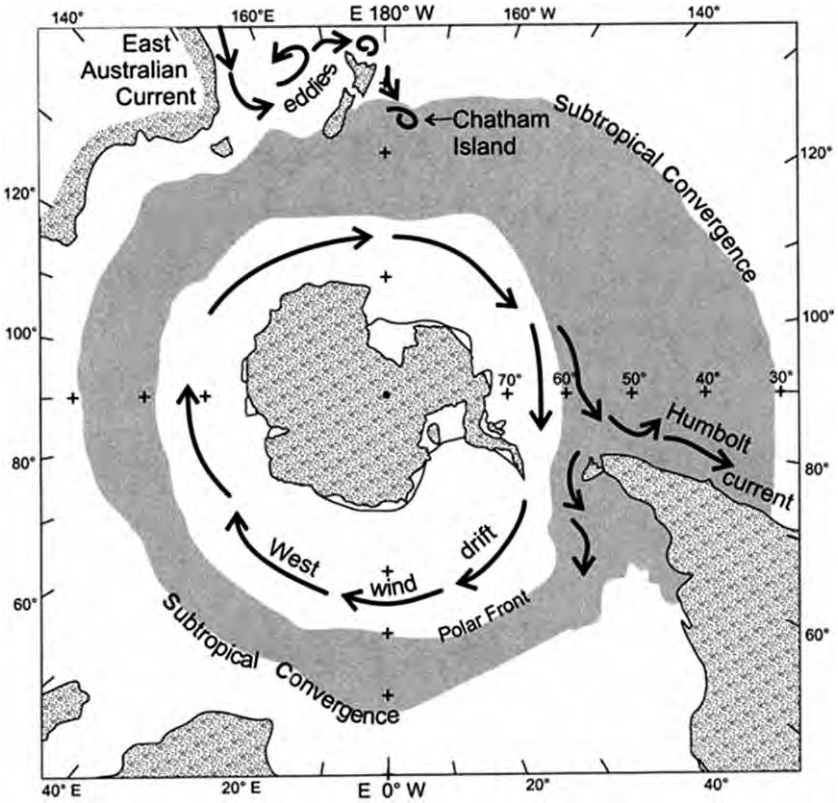

Figure 5. The Subtropical Convergence and strong easterly flowing Antarctic Circumpolar Polar Current are frontal zones and are 'leaky barriers' which some organisms are now traversing.

which there is no local (i.e. Chatham Island) source, are also common on these shores. The evidence for long-distance transport is irrefutable and the possibility of alien introductions must be acknowledged.

Following the separation of Antarctica from Gondwanaland, initiation in the Southern Ocean of the strong easterly-flowing Antarctic Circumpolar Current and development of the Polar Front some 30 million years ago, the Antarctic Continent has been effectively isolated (figure 5). As a consequence, the biota of shallow marine environments around the continent are highly endemic (Knox 1994). The Polar Frontal Zone (PFZ) (Antarctic Convergence) encircles the Antarctic Continent. Although noting records of pumice, dunnage and tree trunks escaping its clasp, Gregory *et al.* (1984), Gregory (1990) and Gregory & Ryan (1997) suggested that oceanic fronts, such as the Subtropical Convergence and Polar Front, were obstructions along which marine debris tended to collect and concentrate, and which would be difficult for it to cross. It is now appreciated that these obstructions are in reality somewhat 'leaky barriers' (see Barnes & Fraser 2003; Barnes *et al.* 2006, 2009; appendix A(vii)). However, the quantities of plastic trash and other debris here are many magnitudes less than the concentrations recorded by Moore (2003) in the North Pacific subtropical gyre.

For marine debris with hitch-hiking aliens aboard, the possibilities of north–south (incursions) and south–north (excursions) transfers are probably greatest through disturbances and eddying as waters of the PFZ are forced through Drake Passage, as well as gyral circulation patterns that develop off the Weddell and Ross Seas. The recent report of 10 invertebrate species attached to plastic strapping that stranded on the ice-strewn shore of Adelaide Island (68°S) west of the Antarctic Peninsula exemplifies a developing problem (appendix A(viii)) (Barnes & Fraser 2003). Predicted climate changes and surface water warming of the Southern Ocean will only enhance the possibilities of this two-way latitudinal traffic. This is unlikely to be of benefit for endemic species that have been long isolated and are adapted and restricted to local cold-water environments.

5. DISCUSSION

The environmental, cultural, aesthetic, commercial and other problems arising from pelagic plastics in particular and varied marine debris items in general are manifold, widely acknowledged and often difficult to address (see discussion in Thompson *et al.* 2009*a*,*b*). For instance, in today's world, 'beach clean-ups', whether by mechanical means and managed by local authorities, or following responses organized through public interest groups, have become phenomena of global proportions (see Ryan *et al.* 2009). The latter often involve tedious 'hand picking' and in some situations may endanger the health of participants. While recovery and/or collection of marine debris through 'herding' and use of barriers in harbour, port, estuary and near-shore settings is not uncommon, it is a difficult, if not nigh-on impossible, task for dispersed material afloat on the high seas. These approaches are not a panacea, for to date they do not seem to have led to any great reduction of marine debris materials afloat in the global oceans and enclosed seas or being cast ashore. I am of the opinion that attacking the

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source(s) at their varied places of origin may be the only viable approach in the longer term (see discussion in Thompson *et al.* 2009*b*). The possibility of long-distance slow dispersal of common 'fouling organisms' (marine and non-marine) through hitch-hiking, hanging-on and/or rafting has been recognized for some time (e.g. Wheeler 1916; WHOI 1952; Gregory 1978; Jokiel 1990) and the environmental importance of this process is now widely acknowledged (e.g. Barnes 2002*a*,*b*; Barnes *et al.* 2006). The possibility that pelagic plastics may be potential vectors in the dispersal of aggressive and invasive marine (and terrestrial) organisms that could endanger endemic biota now warrants serious consideration. The dangers are probably greatest where endemism is significant, such as in the remote tropical and mid-latitude islands of Oceania, and isolated sub-Antarctic islands. In a forthcoming era of global warming, shallow marine waters around Antarctica could be similarly threatened. Mechanisms for the evaluation of biosecurity and management of aggressive alien marine bioinvasions in the Southern Ocean are important recent developments (e.g. Lewis *et al.* 2004; Hewitt & Campbell 2007).

Despite numerous informal gatherings and beach clean-up exercises, local authority concerns, regional and international meetings, together with more formal conference settings, an ever-expanding volume of research literature (often repetitive), as well as attracting the interest of UNEP's Regional Seas Programme since 1974, the environmental and multiple other problems associated with plastic-dominated marine debris appear to be ever expanding! For the present there seem to be no satisfactory and/or practical answers to the varied problems plastic debris creates in marine settings. Longer term successes of beach clean-ups and prescribed action plans are questionable. There are clearly needs for new approaches—foremost among these will probably be further development of biodegradable plastics with significantly reduced and tightly managed disintegration times (see Song *et al.* 2009; Thompson *et al.* 2009*b*).

My interest since the early 1970s in the varied problems associated with 'marine debris' was funded initially through University of Auckland's research grants and later by the New Zealand Ministry of Environment research agenda, as well as the Marine Mammal Commission, Washington. I also acknowledge technical and other support over many years from R. Harris, K. Johnston, B. Curram and L. Cotterall. I have valued Allan Williams' commentaries on the topic of marine debris for many years and also appreciate the careful attention given to the manuscript by two anonymous referees.

APPENDIX A: A CATALOGUE WITH EXAMPLES OF SOME INVADING SPECIES, NOT ALL ARE NECESSARILY AGGRESSIVE ALIENS (EXPANDED FROM GREGORY 2004)

(i) *Membranipora tuberculata*: This common bryozoan was not identified in New Zealand waters until the early 1970s when identifications around northern New Zealand were made. Specimens were found on plastic substrates including virgin plastic pellets (or nibs) and rarely on some larger artefacts. It was inferred that there had been eastwards dispersal from Australia across the northern Tasman sea by way of eddies in the East Australian Current (Gregory 1978). Later, L. M. Stevens (1992, unpublished data) was to report that it was abundant on both eastern and western shores around northernmost New Zealand. Several specimens were later noted (M. R. Gregory, unpublished) on occasional nibs found in concentrations on northern shores of Chatham Island. This island lies virtually astride the Subtropical Convergence where marine debris tends to gather and it is probably driven ashore by northerly winds (Gregory 1999*a*).

(ii) *Lopha cristagalli*: Numerous specimens of this common tropical water, Indo-Pacific oyster have been found attached to a tangled mass of synthetic rope stranded on a remote and isolated beach of Fiordland, southwestern New Zealand (Winston *et al.* 1997). The only previous local record of this taxon was in 1971 when a length of synthetic rope, hauled up from shallow water off Parengarenga Harbour in the far north, carried several recently dead specimens. It was suggested that their presence was due to an overseas fishing vessel (Gardner 1971). Recently, a similarly entangled mass of rope encrusted with a hermatypic coral has stranded at the same Fiordland locality (J. Lindqvist, personal communication).

(iii) Plastic toy boats: West's (1981) report of a child's small (<30 cm) plastic toy boat stranded on a small island (Tiritiri matangi) lying *c.* 4 km offshore in the Hauraki gulf near Auckland, New Zealand, is most informative. It was carrying a cargo with soil and the seeds of eight plant species. Of these, five were native and three exotic; one was of a species not known from the island and at least three were viable.

(iv) *Thalamoporella evelinae*: Bryozoans resembling this taxon, which is known from Brazil, arrived in significant quantities on Florida shores through attachment to pelagic plastic artefacts and later stranding on beaches where it had not been previously identified (Winston *et al.* 1997).

(v) *Pinctata spp.* A large blue fish crate with prominent Venezuelan markings stranding on Bermuda beaches with several single attached valves of this taxon suggests long-distance transport by way of the Gulf Stream (personal observation).

(vi) *Diadumene lineata*: In November 2000, numerous individuals of this widely recognized, aggressive and invasive inter-tidal sea anenome were discovered on derelict trawl netting in the lagoon of Pearl and Hermes Reef, Northwestern Hawaiian Islands. This is a cosmopolitan taxon that is native to Japan and had not been previously identified in Hawaiian waters. It was suggested that the net with its hitch-hiking cargo of *D. lineata* could have drifted from afar—possibly Japan (Zabin *et al.* 2004).

(vii) *Mytilis galloprovincialis*: This exotic smooth-shelled blue mussel arrived in Pearl Harbor, Hawaii (June 1998) as a component of the fouling community carried by *USS Missouri*. Apte *et al.* (2000) reported that spawning took place shortly thereafter and were later recruited to another 'shipping vector'. They infer that a 'stepping stone' model between temperate latitudes could lead to dispersal and/or range expansion.

(viii) Adelaide Island, Antarctic Peninsula: Barnes & Fraser (2003) reported a plastic strapping

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band washing ashore, and on which were attached 10 species belonging to five phylla, including bryozoa, porifera, annelida, cnidaria and mollusca. It was suggested that this plastic artefact could have been afloat for at least 1 year.

(ix) Harmful microalgae: Masó *et al.*'s (2003) observations along the Catalan coast (northwestern Mediterranean) and suggestions that pelagic plastic debris could be a vector in the dispersal of harmful microalgae.

(x) *Elminius modestus*: This barnacle is endemic to Australasian waters. It arrived in southern England during the Second World War—perhaps through attachment to convoyed vessels. Subsequently, this aggressive and alien invasive taxon advanced northwards, colonizing rocky inter-tidal shores around the British Isles and also adjacent coasts of Europe. By 1978 it had reached the Shetland Islands. There are suggestions that in later years plastic substrates could have been implicated in this dispersal (Barnes & Milner 2005).

(xi) Macrobenthos, Ligurian Sea: In these waters, Aliani & Molcard (2003) have documented macrobenthic species colonizing plastic artefacts and occasional pieces of wood. Of the 14 stations sampled, the barnacle *Lepas pectinata* was present at 12 and the isopod *Idotea metallica* at 9. Hydroids and bryozoa were also common. They also noted that no alien species had been identified.

(xii) *Galeopsis mimicus*: This bryozoan was previously known from two sampling stations off the west coast of the South Island of New Zealand at water depths of 297 and 520 m. It is also known from >2000 km to the north and in water depths of 470–825 m. It has recently been identified on a small piece of frayed plastic substrate recovered from the top of a core taken *c.* 60 km off the Canterbury east coast at a depth of 393 m (figure 4*d*) (Carter & Gregory 2005).

(xiii) *Giamardia trapesina*: Long-distance dispersal in the sub-Antarctic and Southern Ocean waters through kelp rafting of brooding bivalves (Helmuth *et al.* 1994).

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PHILOSOPHICAL TRANSACTIONS OF THE ROYAL SOCIETY B BIOLOGICAL SCIENCES

Preface

Brian Heap

Phil. Trans. R. Soc. B 2009 **364**, 1971
doi: 10.1098/rstb.2009.0030

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PHILOSOPHICAL TRANSACTIONS OF THE ROYAL SOCIETY B

Phil. Trans. R. Soc. B (2009) 364, 1971
doi:10.1098/rstb.2009.0030

Preface

Plastics, the environment and human health featured prominently in an international seminar on Planetary Emergencies held at Erice, Sicily in August 2006. Enormous advantages have accrued from the industrial development and application of plastics, but whereas plastics production has increased from 0.5 to 260 million tonnes p.a. since 1950 the amount regenerated is relatively small, so that the gap between production and regeneration continues to expand. Plastics litter has reached a figure of approximately 10 per cent of municipal waste mass worldwide, material that has a longevity measured in hundreds of years.

Among the steps taken so far, biodegradable plastics have proved an important advance yet not all are destroyed completely in natural environments, raising the question of the definition of biodegradable. Close attention has been given to risks associated with additives that improve the properties of plastics yet leak into the environment, food and beverages. Plastic piping used to deliver potable water is now widespread, but phthalate additives in a wide range of flexible plastics, industrial solvents, personal care products and medical devices can show anti-androgenic properties at certain concentrations. Bisphenol A (BPA), a major additive in polycarbonate plastic for lining metal cans and many other products, can reveal oestrogen-mimicking properties.

I, along with several in the Erice audience, had accepted that leach was below the levels of toxicity of concern for humans, other vertebrates or invertebrates. However, independent laboratory work in animals treated with BPA suggested effects on body weight and hormonal parameters resembling those found in diabetes and obesity. Phthalates and BPA have been found in pregnant women and foetuses, and many in the USA have phthalates in their urine. So the big question is—does it matter? Has the tolerable daily intake of additives been exceeded, affecting human health, wildlife and the marine environment?

Some of the Erice papers caused us to ask whether sufficient attention was being given to the discovery of alternative additives that would, for example, avoid any association with endocrine disruption. Was there a scientific consensus about risks associated with the rising accumulation, deposition and interaction of multiple forms of plastics and leached additives in the environment? Could a product-recycling loop be developed that effectively reduced the plastics burden on humans and the environment? Could an international partnership be forged between scientists, policy-makers and industry to arrive at new solutions (as in the displacement of CFCs)?

Such questions were believed to be of sufficient scientific and public merit that a themed issue of peer-reviewed papers has been assembled by guest editors closely involved in the subject. The papers, some of which were aired at Erice, address the implications of our dependency on plastics in today's consumer society. They aim to inform and update the debate about plastics and promote further studies relevant to the assessment and management of risk, if it exists.

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February 2009

One contribution of 15 to a Theme Issue 'Plastics, the environment and human health'.

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PHILOSOPHICAL TRANSACTIONS OF THE ROYAL SOCIETY B BIOLOGICAL SCIENCES

Plastics recycling: challenges and opportunities

Jefferson Hopewell, Robert Dvorak and Edward Kosior

Phil. Trans. R. Soc. B 2009 **364**, 2115-2126
doi: 10.1098/rstb.2008.0311

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PHILOSOPHICAL TRANSACTIONS OF THE ROYAL SOCIETY B

Phil. Trans. R. Soc. B (2009) **364**, 2115–2126
doi:10.1098/rstb.2008.0311

Review

Plastics recycling: challenges and opportunities

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Plastics are inexpensive, lightweight and durable materials, which can readily be moulded into a variety of products that find use in a wide range of applications. As a consequence, the production of plastics has increased markedly over the last 60 years. However, current levels of their usage and disposal generate several environmental problems. Around 4 per cent of world oil and gas production, a non-renewable resource, is used as feedstock for plastics and a further 3–4% is expended to provide energy for their manufacture. A major portion of plastic produced each year is used to make disposable items of packaging or other short-lived products that are discarded within a year of manufacture. These two observations alone indicate that our current use of plastics is not sustainable. In addition, because of the durability of the polymers involved, substantial quantities of discarded end-of-life plastics are accumulating as debris in landfills and in natural habitats worldwide.

Recycling is one of the most important actions currently available to reduce these impacts and represents one of the most dynamic areas in the plastics industry today. Recycling provides opportunities to reduce oil usage, carbon dioxide emissions and the quantities of waste requiring disposal. Here, we briefly set recycling into context against other waste-reduction strategies, namely reduction in material use through downgauging or product reuse, the use of alternative biodegradable materials and energy recovery as fuel.

While plastics have been recycled since the 1970s, the quantities that are recycled vary geographically, according to plastic type and application. Recycling of packaging materials has seen rapid expansion over the last decades in a number of countries. Advances in technologies and systems for the collection, sorting and reprocessing of recyclable plastics are creating new opportunities for recycling, and with the combined actions of the public, industry and governments it may be possible to divert the majority of plastic waste from landfills to recycling over the next decades.

Keywords: plastics recycling; plastic packaging; environmental impacts; waste management; chemical recycling; energy recovery

1. INTRODUCTION

The plastics industry has developed considerably since the invention of various routes for the production of polymers from petrochemical sources. Plastics have substantial benefits in terms of their low weight, durability and lower cost relative to many other material types (Andrady & Neal 2009; Thompson *et al.* 2009*a*). Worldwide polymer production was estimated to be 260 million metric tonnes per annum in the year 2007 for all polymers including thermoplastics, thermoset plastics, adhesives and coatings, but not synthetic fibres (PlasticsEurope 2008*b*). This indicates a historical growth rate of about 9 per cent p.a. Thermoplastic resins constitute around two-thirds of this production and their usage is growing at about 5 per cent p.a. globally (Andrady 2003).

Today, plastics are almost completely derived from petrochemicals produced from fossil oil and gas. Around 4 per cent of annual petroleum production is converted directly into plastics from petrochemical feedstock (British Plastics Federation 2008). As the manufacture of plastics also requires energy, its production is responsible for the consumption of a similar additional quantity of fossil fuels. However, it can also be argued that use of lightweight plastics can reduce usage of fossil fuels, for example in transport applications when plastics replace heavier conventional materials such as steel (Andrady & Neal 2009; Thompson *et al.* 2009*b*).

Approximately 50 per cent of plastics are used for single-use disposable applications, such as packaging, agricultural films and disposable consumer items, between 20 and 25% for long-term infrastructure such as pipes, cable coatings and structural materials, and the remainder for durable consumer applications with intermediate lifespan, such as in electronic goods, furniture, vehicles, etc. Post-consumer plastic waste generation across the European Union (EU) was 24.6 million tonnes in 2007 (PlasticsEurope 2008*b*). Table 1 presents a breakdown of plastics consumption in the UK during the year 2000, and contributions to waste generation (Waste Watch 2003).

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One contribution of 15 to a Theme Issue 'Plastics, the environment and human health'.

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Table 1. Consumption of plastics and waste generation by sector in the UK in 2000 (Waste Watch 2003).

	usage		waste arising	
	ktonne	(%)	ktonne	(%)
packaging	1640	37	1640	58
commercial and industrial	*490*			
household	*1150*			
building and construction	1050	24	284	10
structural	*800*		*49*	
non-structural	*250*		*235*	
electrical and electronics	355	8	200	7
furniture and housewares	335	8	200[a]	7
automotive and transport	335	8	150	5
agriculture and horticulture	310	7	93	3
other	425	10	255[a]	9
total	4450		2820	

[a]estimate

This confirms that packaging is the main source of waste plastics, but it is clear that other sources such as waste electronic and electrical equipment (WEEE) and end-of-life vehicles (ELV) are becoming significant sources of waste plastics.

Because plastics have only been mass-produced for around 60 years, their longevity in the environment is not known with certainty. Most types of plastics are not biodegradable (Andrady 1994), and are in fact extremely durable, and therefore the majority of polymers manufactured today will persist for at least decades, and probably for centuries if not millennia. Even degradable plastics may persist for a considerable time depending on local environmental factors, as rates of degradation depend on physical factors, such as levels of ultraviolet light exposure, oxygen and temperature (Swift & Wiles 2004), while biodegradable plastics require the presence of suitable micro-organisms. Therefore, degradation rates vary considerably between landfills, terrestrial and marine environments (Kyrikou & Briassoulis 2007). Even when a plastic item degrades under the influence of weathering, it first breaks down into smaller pieces of plastic debris, but the polymer itself may not necessarily fully degrade in a meaningful timeframe. As a consequence, substantial quantities of end-of-life plastics are accumulating in landfills and as debris in the natural environment, resulting in both waste-management issues and environmental damage (see Barnes *et al.* 2009; Gregory 2009; Oehlmann *et al.* 2009; Ryan *et al.* 2009; Teuten *et al.* 2009; Thompson *et al.* 2009*b*).

Recycling is clearly a waste-management strategy, but it can also be seen as one current example of implementing the concept of industrial ecology, whereas in a natural ecosystem there are no wastes but only products (Frosch & Gallopoulos 1989; McDonough & Braungart 2002). Recycling of plastics is one method for reducing environmental impact and resource depletion. Fundamentally, high levels of recycling, as with reduction in use, reuse and repair or re-manufacturing can allow for a given level of product service with lower material inputs than would otherwise be required. Recycling can therefore decrease energy and material usage per unit of output and so yield improved eco-efficiency (WBCSD 2000). Although, it should be noted that the ability to maintain whatever residual level of material input, plus the energy inputs and the effects of external impacts on ecosystems will decide the ultimate sustainability of the overall system.

In this paper, we will review the current systems and technology for plastics recycling, life-cycle evidence for the eco-efficiency of plastics recycling, and briefly consider related economic and public interest issues. We will focus on production and disposal of packaging as this is the largest single source of waste plastics in Europe and represents an area of considerable recent expansion in recycling initiatives.

2. WASTE MANAGEMENT: OVERVIEW

Even within the EU there are a wide range of waste-management prioritizations for the total municipal solid waste stream (MSW), from those heavily weighted towards landfill, to those weighted towards incineration (figure 1)—recycling performance also varies considerably. The average amount of MSW generated in the EU is 520 kg per person per year and projected to increase to 680 kg per person per year by 2020 (EEA 2008). In the UK, total use of plastics in both domestic and commercial packaging is about 40 kg per person per year, hence it forms approximately 7–8% by weight, but a larger proportion by volume of the MSW stream (Waste Watch 2003).

Broadly speaking, waste plastics are recovered when they are diverted from landfills or littering. Plastic packaging is particularly noticeable as litter because of the lightweight nature of both flexible and rigid plastics. The amount of material going into the waste-management system can, in the first case, be reduced by actions that decrease the use of materials in products (e.g. substitution of heavy packaging formats with lighter ones, or downgauging of packaging). Designing products to enable reusing, repairing or re-manufacturing will result in fewer products entering the waste stream.

Once material enters the waste stream, recycling is the process of using recovered material to manufacture a new product. For organic materials like plastics, the concept of recovery can also be expanded to include energy recovery, where the calorific value of the material is utilized by controlled combustion as a fuel, although this results in a lesser overall environmental performance than material recovery as it does not reduce the demand for new (virgin) material. This thinking is the basis of the 4Rs strategy in waste management parlance—in the order of decreasing environmental desirability—reduce, reuse, recycle (materials) and recover (energy), with landfill as the least desirable management strategy.

It is also quite possible for the same polymer to cascade through multiple stages—e.g. manufacture into a re-usable container, which once entering the waste stream is collected and recycled into a durable application that when becoming waste in its turn, is recovered for energy.

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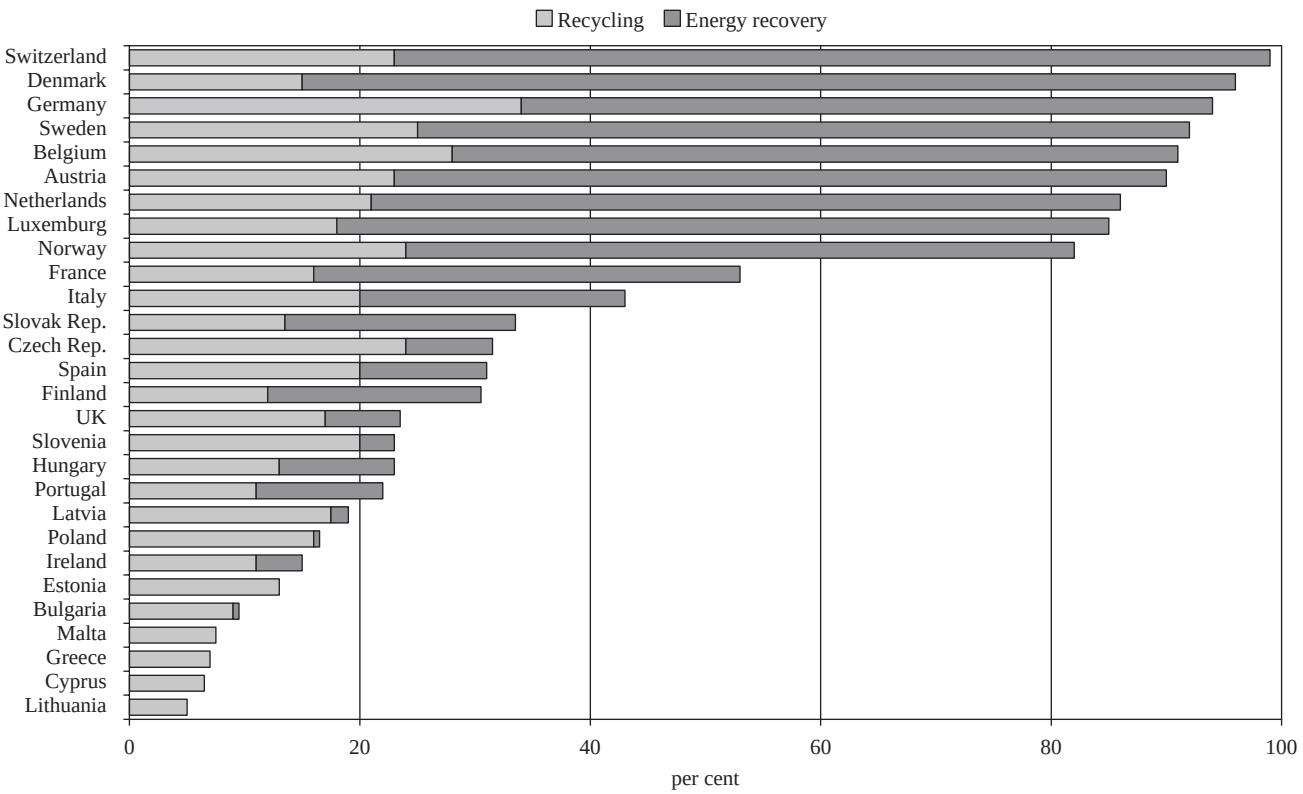

Figure 1. Rates of mechanical recycling and energy recovery as waste-management strategies for plastics waste in European nations (PlasticsEurope 2008*b*).

(a) *Landfill*

Landfill is the conventional approach to waste management, but space for landfills is becoming scarce in some countries. A well-managed landfill site results in limited immediate environmental harm beyond the impacts of collection and transport, although there are long-term risks of contamination of soils and groundwater by some additives and breakdown by-products in plastics, which can become persistent organic pollutants (Oehlmann *et al.* 2009; Teuten *et al.* 2009). A major drawback to landfills from a sustainability aspect is that none of the material resources used to produce the plastic is recovered—the material flow is linear rather than cyclic. In the UK, a landfill tax has been applied, which is currently set to escalate each year until 2010 in order to increase the incentive to divert wastes from landfill to recovery actions such as recycling (DEFRA 2007).

(b) *Incineration and energy recovery*

Incineration reduces the need for landfill of plastics waste, however, there are concerns that hazardous substances may be released into the atmosphere in the process. For example, PVC and halogenated additives are typically present in mixed plastic waste leading to the risk of dioxins, other polychlorinated biphenyls and furans being released into the environment (Gilpin *et al.* 2003). As a consequence primarily of this perceived pollution risk, incineration of plastic is less prevalent than landfill and mechanical recycling as a waste-management strategy. Japan and some European countries such as Denmark and Sweden are notable exceptions, with extensive incinerator infrastructure in place for dealing with MSW, including plastics.

Incineration can be used with recovery of some of the energy content in the plastic. The useful energy recovered can vary considerably depending on whether it is used for electricity generation, combined heat and power, or as solid refuse fuel for co-fuelling of blast furnaces or cement kilns. Liquefaction to diesel fuel or gasification through pyrolysis is also possible (Arvanitoyannis & Bosnea 2001) and interest in this approach to produce diesel fuel is increasing, presumably owing to rising oil prices. Energy-recovery processes may be the most suitable way for dealing with highly mixed plastic such as some electronic and electrical wastes and automotive shredder residue.

(c) *Downgauging*

Reducing the amount of packaging used per item will reduce waste volumes. Economics dictate that most manufacturers will already use close to the minimum required material necessary for a given application (but see Thompson *et al.* 2009*b*, Fig 1). This principle is, however, offset against aesthetics, convenience and marketing benefits that can lead to over-use of packaging, as well as the effect of existing investment in tooling and production process, which can also result in excessive packaging of some products.

(d) *Re-use of plastic packaging*

Forty years ago, re-use of post-consumer packaging in the form of glass bottles and jars was common. Limitations to the broader application of rigid container re-use are at least partially logistical, where distribution

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and collection points are distant from centralized product-filling factories and would result in considerable back-haul distances. In addition, the wide range of containers and packs for branding and marketing purposes makes direct take-back and refilling less feasible. Take-back and refilling schemes do exist in several European countries (Institute for Local Self-Reliance 2002), including PET bottles as well as glass, but they are elsewhere generally considered a niche activity for local businesses rather than a realistic large-scale strategy to reduce packaging waste.

There is considerable scope for re-use of plastics used for the transport of goods, and for potential re-use or re-manufacture from some plastic components in high-value consumer goods such as vehicles and electronic equipment. This is evident in an industrial scale with re-use of containers and pallets in haulage (see Thompson *et al.* 2009*b*). Some shift away from single-use plastic carrier bags to reusable bags has also been observed, both because of voluntary behaviour change programmes, as in Australia (Department of Environment and Heritage (Australia) 2008) and as a consequence of legislation, such as the plastic bag levy in Ireland (Department of Environment Heritage and Local Government (Ireland) 2007), or the recent banning of lightweight carrier bags, for example in Bangladesh and China.

(e) *Plastics recycling*

Terminology for plastics recycling is complex and sometimes confusing because of the wide range of recycling and recovery activities (table 2). These include four categories: primary (mechanical reprocessing into a product with equivalent properties), secondary (mechanical reprocessing into products requiring lower properties), tertiary (recovery of chemical constituents) and quaternary (recovery of energy). Primary recycling is often referred to as closed-loop recycling, and secondary recycling as downgrading. Tertiary recycling is either described as chemical or feedstock recycling and applies when the polymer is de-polymerized to its chemical constituents (Fisher 2003). Quaternary recycling is energy recovery, energy from waste or valorization. Biodegradable plastics can also be composted, and this is a further example of tertiary recycling, and is also described as organic or biological recycling (see Song *et al.* 2009).

It is possible in theory to closed-loop recycle most thermoplastics, however, plastic packaging frequently uses a wide variety of different polymers and other materials such as metals, paper, pigments, inks and adhesives that increases the difficulty. Closed-loop recycling is most practical when the polymer constituent can be (i) effectively separated from sources of contamination and (ii) stabilized against degradation during reprocessing and subsequent use. Ideally, the plastic waste stream for reprocessing would also consist of a narrow range of polymer grades to reduce the difficulty of replacing virgin resin directly. For example, all PET bottles are made from similar grades of PET suitable for both the bottle manufacturing process and reprocessing to polyester fibre, while HDPE used for blow moulding bottles is less-suited to injection moulding applications. As a result, the only parts of the post-consumer plastic waste stream that have routinely been recycled in a strictly closed-loop fashion are clear PET bottles and recently in the UK, HDPE milk bottles. Pre-consumer plastic waste such as industrial packaging is currently recycled to a greater extent than post-consumer packaging, as it is relatively pure and available from a smaller number of sources of relatively higher volume. The volumes of post-consumer waste are, however, up to five times larger than those generated in commerce and industry (Patel *et al.* 2000) and so in order to achieve high overall recycling rates, post-consumer as well as post-industrial waste need to be collected and recycled.

Table 2. Terminology used in different types of plastics recycling and recovery.

ASTM D5033 definitions	equivalent ISO 15270 (draft) definitions	other equivalent terms
primary recycling	mechanical recycling	closed-loop recycling
secondary recycling	mechanical recycling	downgrading
tertiary recycling	chemical recycling	feedstock recycling
quaternary recycling	energy recovery	valorization

In some instances recovered plastic that is not suitable for recycling into the prior application is used to make a new plastic product displacing all, or a proportion of virgin polymer resin—this can also be considered as primary recycling. Examples are plastic crates and bins manufactured from HDPE recovered from milk bottles, and PET fibre from recovered PET packaging. Downgrading is a term sometimes used for recycling when recovered plastic is put into an application that would not typically use virgin polymer—e.g. 'plastic lumber' as an alternative to higher cost/shorter lifetime timber, this is secondary recycling (ASTM Standard D5033).

Chemical or feedstock recycling has the advantage of recovering the petrochemical constituents of the polymer, which can then be used to re-manufacture plastic or to make other synthetic chemicals. However, while technically feasible it has generally been found to be uneconomic without significant subsidies because of the low price of petrochemical feedstock compared with the plant and process costs incurred to produce monomers from waste plastic (Patel *et al.* 2000). This is not surprising as it is effectively reversing the energy-intensive polymerization previously carried out during plastic manufacture.

Feedstock recycling of polyolefins through thermal-cracking has been performed in the UK through a facility initially built by BP and in Germany by BASF. However, the latter plant was closed in 1999 (Aguado *et al.* 2007). Chemical recycling of PET has been more successful, as de-polymerization under milder conditions is possible. PET resin can be broken down by glycolysis, methanolysis or hydrolysis, for example to make unsaturated polyester resins (Sinha *et al.* 2008). It can also be converted back

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into PET, either after de-polymerization, or by simply re-feeding the PET flake into the polymerization reactor, this can also remove volatile contaminants as the reaction occurs under high temperature and vacuum (Uhde Inventa-Fischer 2007).

(f) *Alternative materials*

Biodegradable plastics have the potential to solve a number of waste-management issues, especially for disposable packaging that cannot be easily separated from organic waste in catering or from agricultural applications. It is possible to include biodegradable plastics in aerobic composting, or by anaerobic digestion with methane capture for energy use. However, biodegradable plastics also have the potential to complicate waste management when introduced without appropriate technical attributes, handling systems and consumer education. In addition, it is clear that there could be significant issues in sourcing sufficient biomass to replace a large proportion of the current consumption of polymers, as only 5 per cent of current European chemical production uses biomass as feedstock (Soetaert & Vandamme 2006). This is a large topic that cannot be covered in this paper, except to note that it is desirable that compostable and degradable plastics are appropriately labelled and used in ways that complement, rather than compromise waste-management schemes (see Song *et al.* 2009).

3. SYSTEMS FOR PLASTIC RECYCLING

Plastic materials can be recycled in a variety of ways and the ease of recycling varies among polymer type, package design and product type. For example, rigid containers consisting of a single polymer are simpler and more economic to recycle than multi-layer and multi-component packages.

Thermoplastics, including PET, PE and PP all have high potential to be mechanically recycled. Thermosetting polymers such as unsaturated polyester or epoxy resin cannot be mechanically recycled, except to be potentially re-used as filler materials once they have been size-reduced or pulverized to fine particles or powders (Rebeiz & Craft 1995). This is because thermoset plastics are permanently cross-linked in manufacture, and therefore cannot be re-melted and re-formed. Recycling of cross-linked rubber from car tyres back to rubber crumb for re-manufacture into other products does occur and this is expected to grow owing to the EU Directive on Landfill of Waste (1999/31/EC), which bans the landfill of tyres and tyre waste.

A major challenge for producing recycled resins from plastic wastes is that most different plastic types are not compatible with each other because of inherent immiscibility at the molecular level, and differences in processing requirements at a macro-scale. For example, a small amount of PVC contaminant present in a PET recycle stream will degrade the recycled PET resin owing to evolution of hydrochloric acid gas from the PVC at a higher temperature required to melt and reprocess PET. Conversely, PET in a PVC recycle stream will form solid lumps of undispersed crystalline PET, which significantly reduces the value of the recycled material.

Hence, it is often not technically feasible to add recovered plastic to virgin polymer without decreasing at least some quality attributes of the virgin plastic such as colour, clarity or mechanical properties such as impact strength. Most uses of recycled resin either blend the recycled resin with virgin resin—often done with polyolefin films for non-critical applications such as refuse bags, and non-pressure-rated irrigation or drainage pipes, or for use in multi-layer applications, where the recycled resin is sandwiched between surface layers of virgin resin.

The ability to substitute recycled plastic for virgin polymer generally depends on the purity of the recovered plastic feed and the property requirements of the plastic product to be made. This has led to current recycling schemes for post-consumer waste that concentrate on the most easily separated packages, such as PET soft-drink and water bottles and HDPE milk bottles, which can be positively identified and sorted out of a co-mingled waste stream. Conversely, there is limited recycling of multi-layer/multi-component articles because these result in contamination between polymer types. Post-consumer recycling therefore comprises of several key steps: collection, sorting, cleaning, size reduction and separation, and/or compatibilization to reduce contamination by incompatible polymers.

(a) *Collection*

Collection of plastic wastes can be done by 'bring-schemes' or through kerbside collection. Bring-schemes tend to result in low collection rates in the absence of either highly committed public behaviour or deposit-refund schemes that impose a direct economic incentive to participate. Hence, the general trend is for collection of recyclable materials through kerbside collection alongside MSW. To maximize the cost efficiency of these programmes, most kerbside collections are of co-mingled recyclables (paper/board, glass, aluminium, steel and plastic containers). While kerbside collection schemes have been very successful at recovering plastic bottle packaging from homes, in terms of the overall consumption typically only 30–40% of post-consumer plastic bottles are recovered, as a lot of this sort of packaging comes from food and beverage consumed away from home. For this reason, it is important to develop effective 'on-the-go' and 'office recycling' collection schemes if overall collection rates for plastic packaging are to increase.

(b) *Sorting*

Sorting of co-mingled rigid recyclables occurs by both automatic and manual methods. Automated pre-sorting is usually sufficient to result in a plastics stream separate from glass, metals and paper (other than when attached, e.g. as labels and closures). Generally, clear PET and unpigmented HDPE milk bottles are positively identified and separated out of the stream. Automatic sorting of containers is now widely used by material recovery facility operators and also by many plastic recycling facilities. These

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systems generally use Fourier-transform near-infrared (FT-NIR) spectroscopy for polymer type analysis and also use optical colour recognition camera systems to sort the streams into clear and coloured fractions. Optical sorters can be used to differentiate between clear, light blue, dark blue, green and other coloured PET containers. Sorting performance can be maximized using multiple detectors, and sorting in series. Other sorting technologies include X-ray detection, which is used for separation of PVC containers, which are 59 per cent chlorine by weight and so can be easily distinguished (Arvanitoyannis & Bosnea 2001; Fisher 2003).

Most local authorities or material recovery facilities do not actively collect post-consumer flexible packaging as there are current deficiencies in the equipment that can easily separate flexibles. Many plastic recycling facilities use trommels and density-based air-classification systems to remove small amounts of flexibles such as some films and labels. There are, however, developments in this area and new technologies such as ballistic separators, sophisticated hydrocyclones and air-classifiers that will increase the ability to recover post-consumer flexible packaging (Fisher 2003).

(c) *Size reduction and cleaning*

Rigid plastics are typically ground into flakes and cleaned to remove food residues, pulp fibres and adhesives. The latest generation of wash plants use only 2–3 m^3 of water per tonne of material, about one-half of that of previous equipment. Innovative technologies for the removal of organics and surface contaminants from flakes include 'dry-cleaning', which cleans surfaces through friction without using water.

(d) *Further separation*

After size reduction, a range of separation techniques can be applied. Sink/float separation in water can effectively separate polyolefins (PP, HDPE, L/LLDPE) from PVC, PET and PS. Use of different media can allow separation of PS from PET, but PVC cannot be removed from PET in this manner as their density ranges overlap. Other separation techniques such as air elutriation can also be used for removing low-density films from denser ground plastics (Chandra & Roy 2007), e.g. in removing labels from PET flakes.

Technologies for reducing PVC contaminants in PET flake include froth flotation (Drelich *et al.* 1998; Marques & Tenorio 2000)[JH1], FT-NIR or Raman emission spectroscopic detectors to enable flake ejection and using differing electrostatic properties (Park *et al.* 2007). For PET flake, thermal kilns can be used to selectively degrade minor amounts of PVC impurities, as PVC turns black on heating, enabling colour-sorting.

Various methods exist for flake-sorting, but traditional PET-sorting systems are predominantly restricted to separating; (i) coloured flakes from clear PET flakes and (ii) materials with different physical properties such as density from PET. New approaches such as laser-sorting systems can be used to remove other impurities such as silicones and nylon.

'Laser-sorting' uses emission spectroscopy to differentiate polymer types. These systems are likely to significantly improve the ability to separate complex mixtures as they can perform up to 860 000 spectra s^{-1} and can scan each individual flake. They have the advantage that they can be used to sort different plastics that are black—a problem with traditional automatic systems. The application of laser-sorting systems is likely to increase separation of WEEE and automotive plastics. These systems also have the capability to separate polymer by type or grade and can also separate polyolefinic materials such as PP from HDPE. However, this is still a very novel approach and currently is only used in a small number of European recycling facilities.

(e) *Current advances in plastic recycling*

Innovations in recycling technologies over the last decade include increasingly reliable detectors and sophisticated decision and recognition software that collectively increase the accuracy and productivity of automatic sorting—for example current FT-NIR detectors can operate for up to 8000 h between faults in the detectors.

Another area of innovation has been in finding higher value applications for recycled polymers in closed-loop processes, which can directly replace virgin polymer (see table 3). As an example, in the UK, since 2005 most PET sheet for thermoforming contains 50–70% recycled PET (rPET) through use of A/B/A layer sheet where the outer layers (A) are food-contact-approved virgin resin, and the inner layer (B) is rPET. Food-grade rPET is also now widely available in the market for direct food contact because of the development of 'super-clean' grades. These only have slight deterioration in clarity from virgin PET, and are being used at 30–50% replacement of virgin PET in many applications and at 100 per cent of the material in some bottles.

A number of European countries including Germany, Austria, Norway, Italy and Spain are already collecting, in addition to their bottle streams, rigid packaging such as trays, tubs and pots as well as limited amounts of post-consumer flexible packaging such as films and wrappers. Recycling of this non-bottle packaging has become possible because of improvements in sorting and washing technologies and emerging markets for the recyclates. In the UK, the Waste Resource Action Programme (WRAP) has run an initial study of mixed plastics recycling and is now taking this to full-scale validation (WRAP 2008*b*). The potential benefits of mixed plastics recycling in terms of resource efficiency, diversion from landfill and emission savings, are very high when one considers the fact that in the UK it is estimated that there is over one million tonne per annum of non-bottle plastic packaging (WRAP 2008*a*) in comparison with 525 000 tonnes of plastic bottle waste (WRAP 2007).

4. ECOLOGICAL CASE FOR RECYCLING

Life-cycle analysis can be a useful tool for assessing the potential benefits of recycling programmes. If recycled plastics are used to produce goods that would

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Table 3. Comparing some environmental impacts of commodity polymer production and current ability for recycling from post-consumer sources.

polymer	LCI data cradle-to-gate (EU data)				closed-loop recycling	effectiveness in current recycling processes
	energy (GJ tonne^{-1})	water (kL tonne^{-1})	CO_2-e[a] (t tonne^{-1})	Usage[b] (ktonne)		
PET	82.7	66	3.4	2160	yes	high with clear PET from bottles coloured PET is mostly used for fibre additional issues with CPET trays, PET-G
HDPE	76.7	32	1.9	5468	some	high with natural HDPE bottles, but more complex for opaque bottles and trays because of wide variety of grades and colour and mixtures with LDPE and PP
PVC	56.7	46	1.9	6509	some	poor recovery because of cross-contamination with PET PVC packages and labels present a major issue with PET bottle and mixed plastics recycling
LDPE	78.1	47	2.1	7899	some	poor recovery rates, mostly as mixed polyolefins that can have sufficient properties for some applications. Most post-consumer flexible packaging not recovered
PP	73.4	43	2.0	7779	in theory	not widely recycled yet from post-consumer, but has potential. Needs action on sorting and separation, plus development of further outlets for recycled PP
PS	87.4	140	3.4	2600	in theory	poor, extremely difficult to cost-effectively separate from co-mingled collection, separate collection of industrial packaging and EPS foam can be effective
recycled plastics	8–55	typical 3.5[c]	typical 1.4	3130	some	considerable variability in energy, water and emissions from recycling processes as it is a developing industry and affected by efficiency of collection, process type and product mix, etc.

[a] CO_2-e is GWP calculated as 100-yr equivalent to CO_2 emissions. All LCI data are specific to European industry and covers the production process of the raw materials, intermediates and final polymer, but not further processing and logistics (PlasticsEurope 2008*a*).
[b] Usage was for the aggregate EU-15 countries across all market sectors in 2002.
[c] Typical values for water and greenhouse gas emissions from recycling activities to produce 1 kg PET from waste plastic (Perugini *et al.* 2005).

otherwise have been made from new (virgin) polymer, this will directly reduce oil usage and emissions of greenhouse gases associated with the production of the virgin polymer (less the emissions owing to the recycling activities themselves). However, if plastics are recycled into products that were previously made from other materials such as wood or concrete, then savings in requirements for polymer production will not be realized (Fletcher & Mackay 1996). There may be other environmental costs or benefits of any such alternative material usage, but these are a distraction to our discussion of the benefits of recycling and would need to be considered on a case-by-case basis. Here, we will primarily consider recycling of plastics into products that would otherwise have been produced from virgin polymer.

Feedstock (chemical) recycling technologies satisfy the general principle of material recovery, but are more costly than mechanical recycling, and less energetically favourable as the polymer has to be depolymerized and then re-polymerized. Historically, this has required very significant subsidies because of the low price of petrochemicals in contrast to the high process and plant costs to chemically recycle polymers.

Energy recovery from waste plastics (by transformation to fuel or by direct combustion for electricity generation, use in cement kilns and blast furnaces, etc.) can be used to reduce landfill volumes, but does not reduce the demand for fossil fuels (as the waste plastic was made from petrochemicals; Garforth *et al.* 2004). There are also environmental and health concerns associated with their emissions.

One of the key benefits of recycling plastics is to reduce the requirement for plastics production. Table 3 provides data on some environmental impacts from production of virgin commodity plastics (up to

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the 'factory gate'), and summarizes the ability of these resins to be recycled from post-consumer waste. In terms of energy use, recycling has been shown to save more energy than that produced by energy recovery even when including the energy used to collect, transport and re-process the plastic (Morris 1996). Life-cycle analyses has also been used for plastic-recycling systems to evaluate the net environmental impacts (Arena *et al.* 2003; Perugini *et al.* 2005) and these find greater positive environmental benefits for mechanical recycling over landfill and incineration with energy recovery.

It has been estimated that PET bottle recycling gives a net benefit in greenhouse gas emissions of 1.5 tonnes of CO_2-e per tonne of recycled PET (Department of Environment and Conservation (NSW) 2005) as well as reduction in landfill and net energy consumption. An average net reduction of 1.45 tonnes of CO_2-e per tonne of recycled plastic has been estimated as a useful guideline to policy (ACRR 2004), one basis for this value appears to have been a German life-cycle analysis (LCA) study (Patel *et al.* 2000), which also found that most of the net energy and emission benefits arise from the substitution of virgin polymer production. A recent LCA specifically for PET bottle manufacture calculated that use of 100 per cent recycled PET instead of 100 per cent virgin PET would reduce the full life-cycle emissions from 446 to 327 g CO_2 per bottle, resulting in a 27 per cent relative reduction in emissions (WRAP 2008*e*).

Mixed plastics, the least favourable source of recycled polymer could still provide a net benefit of the vicinity of 0.5 tonnes of CO_2-e per tonne of recycled product (WRAP 2008*c*). The higher eco-efficiency for bottle recycling is because of both the more efficient process for recycling bottles as opposed to mixed plastics and the particularly high emissions profile of virgin PET production. However, the mixed plastics recycling scenario still has a positive net benefit, which was considered superior to the other options studied, of both landfills and energy recovery as solid refuse fuel, so long as there is substitution of virgin polymer.

5. PUBLIC SUPPORT FOR RECYCLING

There is increasing public awareness on the need for sustainable production and consumption. This has encouraged local authorities to organize collection of recyclables, encouraged some manufacturers to develop products with recycled content, and other businesses to supply this public demand. Marketing studies of consumer preferences indicate that there is a significant, but not overwhelming proportion of people who value environmental values in their purchasing patterns. For such customers, confirmation of recycled content and suitability for recycling of the packaging can be a positive attribute, while exaggerated claims for recyclability (where the recyclability is potential, rather than actual) can reduce consumer confidence. It has been noted that participating in recycling schemes is an environmental behaviour that has wide participation among the general population and was 57 per cent in the UK in a 2006 survey (WRAP 2008*d*), and 80 per cent in an Australian survey where kerbside collection had been in place for longer (NEPC 2001).

Some governments use policy to encourage post-consumer recycling, such as the EU Directive on packaging and packaging waste (94/62/EC). This subsequently led Germany to set-up legislation for extended producer responsibility that resulted in the *die Grüne Punkt* (Green Dot) scheme to implement recovery and recycling of packaging. In the UK, producer responsibility was enacted through a scheme for generating and trading packaging recovery notes, plus more recently a landfill levy to fund a range of waste reduction activities. As a consequence of all the above trends, the market value of recycled polymer and hence the viability of recycling have increased markedly over the last few years.

Extended producer responsibility can also be enacted through deposit-refund schemes, covering for example, beverage containers, batteries and vehicle tyres. These schemes can be effective in boosting collection rates, for example one state of Australia has a container deposit scheme (that includes PET soft-drink bottles), as well as kerbside collection schemes. Here the collection rate of PET bottles was 74 per cent of sales, compared with 36 per cent of sales in other states with kerbside collection only. The proportion of bottles in litter was reduced as well compared to other states (West 2007).

6. ECONOMIC ISSUES RELATING TO RECYCLING

Two key economic drivers influence the viability of thermoplastics recycling. These are the price of the recycled polymer compared with virgin polymer and the cost of recycling compared with alternative forms of acceptable disposal. There are additional issues associated with variations in the quantity and quality of supply compared with virgin plastics. Lack of information about the availability of recycled plastics, its quality and suitability for specific applications, can also act as a disincentive to use recycled material.

Historically, the primary methods of waste disposal have been by landfill or incineration. Costs of landfill vary considerably among regions according to the underlying geology and land-use patterns and can influence the viability of recycling as an alternative disposal route. In Japan, for example, the excavation that is necessary for landfill is expensive because of the hard nature of the underlying volcanic bedrock; while in the Netherlands it is costly because of permeability from the sea. High disposal costs are an economic incentive towards either recycling or energy recovery.

Collection of used plastics from households is more economical in suburbs where the population density is sufficiently high to achieve economies of scale. The most efficient collection scheme can vary with locality, type of dwellings (houses or large multi-apartment buildings) and the type of sorting facilities available. In rural areas 'bring schemes' where the public deliver their own waste for recycling, for example when they visit a nearby town, are considered more cost-effective than kerbside collection. Many local authorities and some supermarkets in the UK operate 'bring banks',

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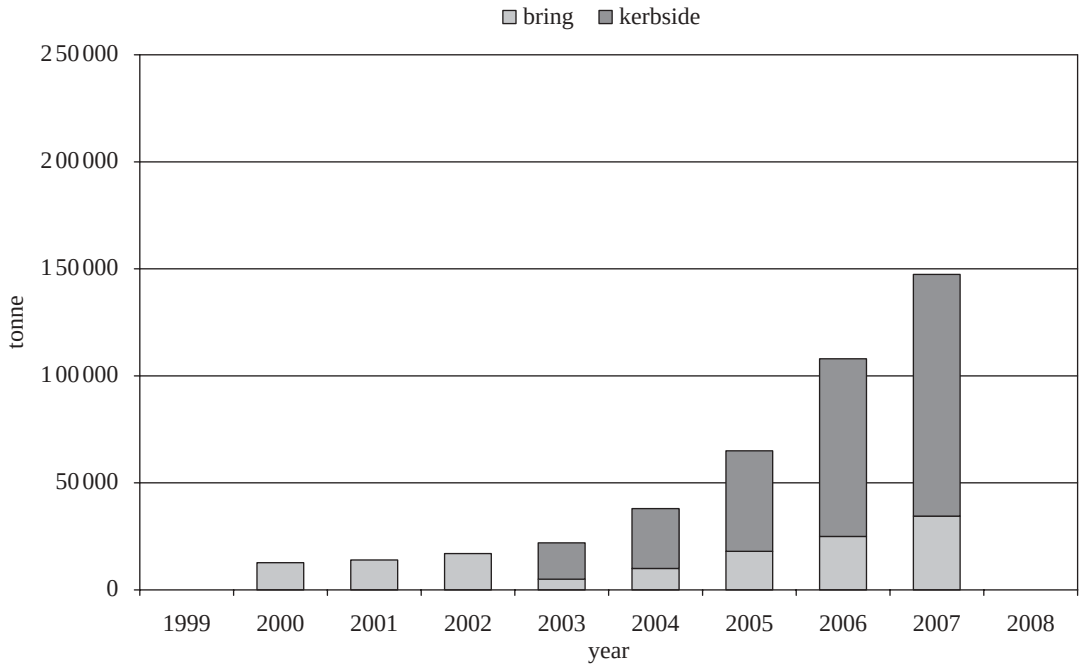

Figure 2. Growth in collection of plastic bottles, by bring and kerbside schemes in the UK (WRAP 2008*d*).

or even reverse-vending machines. These latter methods can be a good source of relatively pure recyclables, but are ineffective in providing high collection rates of post-consumer waste. In the UK, dramatic increases in collection of the plastic bottle waste stream was only apparent after the relatively recent implementation of kerbside recycling (figure 2).

The price of virgin plastic is influenced by the price of oil, which is the principle feedstock for plastic production. As the quality of recovered plastic is typically lower than that of virgin plastics, the price of virgin plastic sets the ceiling for prices of recovered plastic. The price of oil has increased significantly in the last few years, from a range of around USD 25 per barrel to a price band between USD 50–150 since 2005. Hence, although higher oil prices also increase the cost of collection and reprocessing to some extent, recycling has become relatively more financially attractive.

Technological advances in recycling can improve the economics in two main ways—by decreasing the cost of recycling (productivity/efficiency improvements) and by closing the gap between the value of recycled resin and virgin resin. The latter point is particularly enhanced by technologies for turning recovered plastic into food grade polymer by removing contamination—supporting closed-loop recycling. This technology has been proven for rPET from clear bottles (WRAP 2008*b*), and more recently rHDPE from milk bottles (WRAP 2006).

So, while over a decade ago recycling of plastics without subsidies was mostly only viable from post-industrial waste, or in locations where the cost of alternative forms of disposal were high, it is increasingly now viable on a much broader geographic scale, and for post-consumer waste.

7. CURRENT TRENDS IN PLASTIC RECYCLING

In western Europe, plastic waste generation is growing at approximately 3 per cent per annum, roughly in line with long-term economic growth, whereas the amount of mechanical recycling increased strongly at a rate of approximately 7 per cent per annum. In 2003, however, this still amounted to only 14.8 per cent of the waste plastic generated (from all sources). Together with feedstock recycling (1.7 per cent) and energy recovery (22.5 per cent), this amounted to a total recovery rate of approximately 39 per cent from the 21.1 million tonnes of plastic waste generated in 2003 (figure 3). This trend for both rates of mechanical recycling and energy recovery to increase is continuing, although so is the trend for increasing waste generation.

8. CHALLENGES AND OPPORTUNITIES FOR IMPROVING PLASTIC RECYCLING

Effective recycling of mixed plastics waste is the next major challenge for the plastics recycling sector. The advantage is the ability to recycle a larger proportion of the plastic waste stream by expanding post-consumer collection of plastic packaging to cover a wider variety of materials and pack types. Product design for recycling has strong potential to assist in such recycling efforts. A study carried out in the UK found that the amount of packaging in a regular shopping basket that, even if collected, cannot be effectively recycled, ranged from 21 to 40% (Local Government Association (UK) 2007). Hence, wider implementation of policies to promote the use of environmental design principles by industry could have a large impact on recycling performance, increasing the proportion of packaging that can economically be collected and diverted from landfill (see Shaxson *et al.* 2009). The same logic applies to durable consumer goods designing for disassembly, recycling and specifications for use of recycled resins are key actions to increase recycling.

Most post-consumer collection schemes are for rigid packaging as flexible packaging tends to be problematic during the collection and sorting stages. Most current

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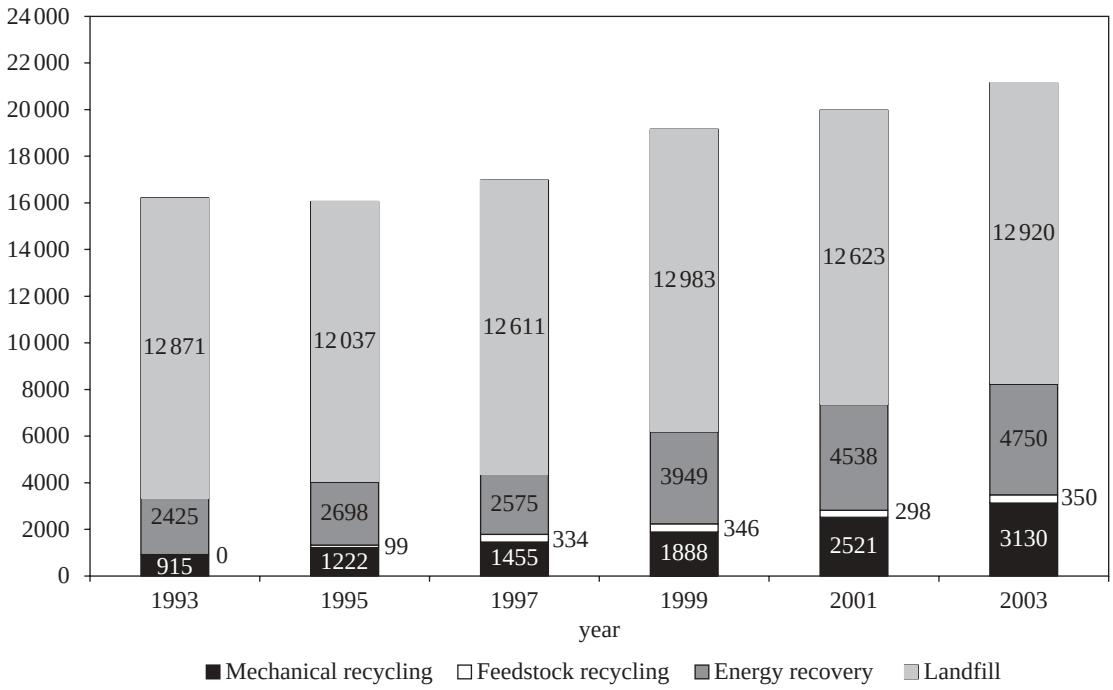

Figure 3. Volumes of plastic waste disposed to landfill, and recovered by various methods in Western Europe, 1993–2003 (APME 2004).

material recovery facilities have difficulty handling flexible plastic packaging because of the different handling characteristics of rigid packaging. The low weight-to-volume ratio of films and plastic bags also makes it less economically viable to invest in the necessary collection and sorting facilities. However, plastic films are currently recycled from sources including secondary packaging such as shrink-wrap of pallets and boxes and some agricultural films, so this is feasible under the right conditions. Approaches to increasing the recycling of films and flexible packaging could include separate collection, or investment in extra sorting and processing facilities at recovery facilities for handling mixed plastic wastes. In order to have successful recycling of mixed plastics, high-performance sorting of the input materials needs to be performed to ensure that plastic types are separated to high levels of purity; there is, however, a need for the further development of endmarkets for each polymer recyclate stream.

The effectiveness of post-consumer packaging recycling could be dramatically increased if the diversity of materials were to be rationalized to a subset of current usage. For example, if rigid plastic containers ranging from bottles, jars to trays were all PET, HDPE and PP, without clear PVC or PS, which are problematic to sort from co-mingled recyclables, then all rigid plastic packaging could be collected and sorted to make recycled resins with minimal cross-contamination. The losses of rejected material and the value of the recycled resins would be enhanced. In addition, labels and adhesive materials should be selected to maximize recycling performance. Improvements in sorting/separation within recycling plants give further potential for both higher recycling volumes, and better eco-efficiency by decreasing waste fractions, energy and water use (see §3). The goals should be to maximize both the volume and quality of recycled resins.

9. CONCLUSIONS

In summary, recycling is one strategy for end-of-life waste management of plastic products. It makes increasing sense economically as well as environmentally and recent trends demonstrate a substantial increase in the rate of recovery and recycling of plastic wastes. These trends are likely to continue, but some significant challenges still exist from both technological factors and from economic or social behaviour issues relating to the collection of recyclable wastes, and substitution for virgin material.

Recycling of a wider range of post-consumer plastic packaging, together with waste plastics from consumer goods and ELVs will further enable improvement in recovery rates of plastic waste and diversion from landfills. Coupled with efforts to increase the use and specification of recycled grades as replacement of virgin plastic, recycling of waste plastics is an effective way to improve the environmental performance of the polymer industry.

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Phthalates and other additives in plastics: human exposure and associated health outcomes

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Phil. Trans. R. Soc. B 2009 **364**, 2097-2113
doi: 10.1098/rstb.2008.0268

Supplementary data "Data Supplement"
http://rstb.royalsocietypublishing.org/content/suppl/2009/06/12/364.1526.2097.DC1.html

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PHILOSOPHICAL TRANSACTIONS OF THE ROYAL SOCIETY B

Phil. Trans. R. Soc. B (2009) **364**, 2097–2113
doi:10.1098/rstb.2008.0268

Phthalates and other additives in plastics: human exposure and associated health outcomes

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Concern exists over whether additives in plastics to which most people are exposed, such as phthalates, bisphenol A or polybrominated diphenyl ethers, may cause harm to human health by altering endocrine function or through other biological mechanisms. Human data are limited compared with the large body of experimental evidence documenting reproductive or developmental toxicity in relation to these compounds. Here, we discuss the current state of human evidence, as well as future research trends and needs.

Because exposure assessment is often a major weakness in epidemiological studies, and *in utero* exposures to reproductive or developmental toxicants are important, we also provide original data on maternal exposure to phthalates during and after pregnancy ($n = 242$). Phthalate metabolite concentrations in urine showed weak correlations between pre- and post-natal samples, though the strength of the relationship increased when duration between the two samples decreased. Phthalate metabolite levels also tended to be higher in post-natal samples.

In conclusion, there is a great need for more human studies of adverse health effects associated with plastic additives. Recent advances in the measurement of exposure biomarkers hold much promise in improving the epidemiological data, but their utility must be understood to facilitate appropriate study design.

Keywords: bisphenol A; endocrine disruption; epidemiology; phthalate; polybrominated diphenyl ether; reproductive health

1. INTRODUCTION

Advances in materials science and engineering in recent decades have led to the widespread and diverse use of plastics to provide cheaper, lighter, stronger, safer, more durable and versatile products and consumer goods that serve to improve our quality of life. Plastics can be designed to keep our foods fresher for longer periods of time, can provide therapeutic benefits through timed-release pharmaceuticals and other medical applications, and can prevent electronics and other household items from starting or spreading fires (see Andrady & Neal 2009; Thompson *et al.* 2009*a*,*b*). However, scientific, governmental and public concern exists over the potential adverse human health risks related to ubiquitous exposures to plastic additives among the general population. The leading hypothesis for these growing concerns is that certain chemicals, used in plastics to provide beneficial physical qualities, may also act as endocrine-disrupting compounds (EDCs) that could lead to adverse reproductive and developmental effects (NRC 1999). In men, environmental or occupational exposures to EDCs may be associated with or lead to declined reproductive capacity or possibly increased risk of testicular or prostate cancer (Fleming *et al.* 1999; Pflieger-Bruss *et al.* 2004; Toft *et al.* 2004). In fact, a number of studies have suggested the use of circulating reproductive hormone levels (follicle-stimulating hormone (FSH) and/or inhibin B) as a surrogate measure for semen quality or fecundity in epidemiologic studies (Jensen *et al.* 1997; Uhler *et al.* 2003; Mabeck *et al.* 2005), although other recent studies suggest hormone levels may lack sufficient ability to predict poor semen quality (Dhooge *et al.* 2007; Meeker *et al.* 2007*a*). Endocrine alterations in women resulting from environmental or occupational exposure may represent increased risk for endometriosis, reproductive and other endocrine-related cancers, or impaired oocyte competence, ovarian function or menstrual cycling (Nicolopoulou-Stamati & Pitsos 2001; Pocar *et al.* 2003; Windham *et al.* 2005). Effects of early life exposures to EDCs remain unclear, though it has been suggested that foetal or childhood exposure may lead to altered sex differentiation (Toppari & Skakkebaek 1998), effects on neurological and reproductive development (Tilson 1998; Teilmann *et al.* 2002; Colborn 2004, 2006; Swan *et al.* 2005) and

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Electronic supplementary material is available at http://dx.doi.org/rstb20080268 or via http://rstb.royalsocietypublishing.org.

One contribution of 15 to a Theme Issue 'Plastics, the environment and human health'.

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increased risk of reproductive problems or cancer later in life (Damgaard *et al.* 2002; Aksglaede *et al.* 2006; Main *et al.* 2006*a*). Programming in early life can determine an individual's future health; therefore, early chemical exposures may have long-term impacts later on in life (Gluckman *et al.* 2008). A leading hypothesis for a collection of linked conditions in human males exposed to EDCs *in utero* is termed 'testicular dysgenesis syndrome (TDS)'. TDS represents a number of reproductive disorders of varying severity that are associated with disturbed gonadal development, including cryptorchidism, hypospadias and smaller reproductive organs (Olesen *et al.* 2007). Later in life, the effects of TDS are hypothesized to manifest as a reduction in semen quality and infertility as well as an increased risk for testicular cancer.

Exposure to plastic additives and other EDCs may cause altered endocrine activity and reproductive development through a number of biological mechanisms, which can target different levels of the hypothalamic–pituitary–gonad/thyroid axis, ranging from effects on hormone receptors to effects on hormone synthesis, secretion or metabolism (Boas *et al.* 2006; Bretveld *et al.* 2006). The purpose of this manuscript is not to discuss the various biological pathways or the hundreds of animal and *in vitro* studies that have been conducted on plastic additives as reproductive and developmental toxicants, but rather to review the existing epidemiologic literature on human exposure to these compounds and the relationship with adverse reproductive or developmental endpoints. Because exposure assessment is a fundamental and frequently weak component in large epidemiological studies due to technical, logistic and financial constraints, and *in utero* exposures are among the exposure periods of greatest concern with regard to EDCs, we also provide original data on human exposure to a class of potential endocrine-disrupting plastic additives during and after pregnancy.

Despite the increasing concern for human health impacts associated with plastic additives, there remains a paucity of human studies that have investigated these relationships. While the clinical significance of some markers of endocrine disruption, reproductive health or altered development that commonly appear in the human research literature remains unclear, such as declines in semen quality or subclinical alterations in circulating hormone levels, there is a limited but growing body of evidence for such changes to be associated with environmental and occupational exposure to plastic additives and other potential EDCs. In addition, these markers may serve as intermediate indicators that altered endocrine function is the pathway linking environmental exposures to clinical reproductive and developmental effects. Also, because such a large number of people are exposed to background levels of a number of proven or suspected EDCs, even seemingly subtle epidemiologic associations may result in large increases in reproductive and other endocrine-related disease among populations and thus should be of great public health concern. The background material presented in this manuscript is meant as an introductory review of human studies conducted in this area to date. The reader is directed to the individual references for additional study detail. We focus here on three types of plastic additives—phthalates, bisphenol A (BPA) and polybrominated diphenyl ethers (PBDEs)—because there is laboratory evidence for reproductive or developmental effects in relation to exposure to these compounds. These chemicals were also chosen to be discussed here based on strong evidence for widespread human exposure (CDC 2005; Calafat *et al.* 2008; Sjodin *et al.* 2008).

2. PHTHALATES

(a) *Exposure*

The diesters of 1,2-benzenedicarboxylic acid (phthalic acid), commonly known as phthalates, are a group of man-made chemicals widely used in industrial applications. High-molecular weight phthalates (e.g. di(2-ethylhexyl) phthalate (DEHP)) are primarily used as plasticizers in the manufacture of flexible vinyl plastic which, in turn, is used in consumer products, flooring and wall coverings, food contact applications and medical devices (David *et al.* 2001; ATSDR 2002; Hauser & Calafat 2005). Manufacturers use low-molecular weight phthalates (e.g. diethyl phthalate (DEP) and dibutyl phthalate (DBP)) as solvents in personal-care products (e.g. perfumes, lotions, cosmetics), and in lacquers, varnishes and coatings, including those used to provide timed releases in some pharmaceuticals (ATSDR 1995, 2001; David *et al.* 2001).

As a result of the ubiquitous use of phthalates in personal-care and consumer products, human exposure is widespread. Exposure through ingestion, inhalation and dermal contact is considered important routes of exposure for the general population (Adibi *et al.* 2003; Rudel *et al.* 2003). For infants and children, added skin contact with surfaces and frequent mouthing of fingers and other objects (e.g. plastic toys) may lead to higher phthalate exposures, as might ingestion of phthalates present in breast milk, infant formula, cow's milk or food packaging (Sathyanarayana 2008). Frequent use of personal-care products may lead to higher exposures to the lower molecular weight phthalates, as increased exposures have been found among men reporting recent use of cologne and aftershave (Duty *et al.* 2005*a*) and among infants whose mothers reported recent use of certain infant-care products (lotions, powders and shampoos) (Sathyanarayana *et al.* 2008). Parenteral exposure from medical devices and products containing phthalates is also an important source of high exposure to phthalates, primarily DEHP (ATSDR 2002; Green *et al.* 2005; Weuve *et al.* 2006), for hospitalized populations.

Upon exposure, phthalates are rapidly metabolized and excreted in urine and faeces (ATSDR 1995, 2001, 2002). Owing to the ubiquitous presence of phthalates in indoor environments and concern for sample contamination when measuring the parent diesters in biological samples, the most common approach for investigating human exposure to phthalates is the measurement of urinary concentrations (biomarkers) of phthalate metabolites. The Centers for Disease Control and Prevention's (CDC) Third National Report on Human Exposure to Environmental Chemicals showed

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that the majority of people in the USA have detectable concentrations of several phthalate monoesters in urine (mono-ethyl phthalate (MEP), mono-(2-ethylhexyl) phthalate (MEHP), mono-butyl phthalate (MBP) and mono-benzyl phthalate (MBzP)), reflecting widespread exposure to the parent diester compounds among the general population (CDC 2005). Two oxidative metabolites of DEHP, mono-(2-ethyl-5-hydroxylhexyl) phthalate (MEHHP) and mono-(2-ethyl-5-oxohexyl) phthalate (MEOHP) were present in most subjects at urinary concentrations higher than those of MEHP, the hydrolytic metabolite of DEHP (CDC 2005).

(b) *Exposure assessment for epidemiologic studies*

Exposure assessment is a vital component in environmental epidemiologic studies, yet the details of measuring exposure and the appropriateness of the exposure measures being used are often overlooked. Inadequate attention to exposure estimation can lead to substantial exposure measurement error or misclassification, which in most cases will bias study results towards the null hypothesis and hinder the investigator's ability to see associations between exposure and outcome (Armstrong 2003). Phthalates have short biologic half-lives, from hours to days, and are quickly excreted from the body. Most studies involving human health effects from phthalates are small and include only a single urine sample that may or may not reflect an individual's long-term exposure level. However, because health endpoints of interest are likely associated with windows of exposure over time intervals longer than a few days, information on the temporal variability of urinary levels of phthalates is needed to optimize the design of an exposure assessment in human studies. Temporal variability in exposure can result from changes in exposure sources, such as diet and product use, as well as from changes in xenobiotic metabolism. Therefore, an individual's exposure level may depend on several factors and it is probable that levels would vary considerably over short time periods, such as days. Alternatively, consistent individual time-activity patterns from day-to-day and month-to-month coupled with stable micro-environmental phthalate concentrations (or stable concentrations in food) may lead to 'pseudo-steady state' metabolite concentrations over long periods of time (NRC 2006).

Although urinary phthalate metabolite levels accurately measure a person's exposure at a single point in time, determining exposure over time intervals of weeks or months may require multiple measurements if substantial within-individual variability exists over time. Several recent studies have explored temporal variability of urinary phthalate metabolites, and high within-individual variability has been reported over the course of several days (Fromme *et al.* 2007). In adult men, Hauser *et al.* (2004) reported high day-to-day and month-to-month variability in a person's urinary monoester concentrations, but demonstrated that a single sample may adequately predict average monoester concentrations over a three-month period. However, the predictive values and temporal reliability measures differed between the various monoesters, with a single urinary measure being most predictive for MEP and least predictive for MEHP, and the authors concluded that a second urine sample collected at least 30 days after the first urine sample would serve to assist researchers in more adequately predicting a subject's relative monoester concentration rank within a population (Hauser *et al.* 2004). Results from a recent study among New York City children with repeat urine samples collected over the course of six months supported the findings and recommendations reported by Hauser *et al.* (2004), and suggest that they may extend to other populations and exposure periods of interest (Teitelbaum *et al.* 2008).

Recent studies indicate that certain phthalate diesters and their metabolites are measurable in human breast milk, cord blood and other pregnancy-related specimens (Adibi *et al.* 2003; Latini *et al.* 2003; Main *et al.* 2006*b*). However, despite the ability to quantify phthalate or phthalate metabolite concentrations in a range of biological sample types, the measurement of phthalate monoesters in urine is probably the best approach for estimating phthalate exposures in perinatal epidemiologic studies (Hogberg *et al.* 2008). Owing to the concern for reproductive and developmental effects from *in utero* exposures to phthalates, and the lack of data on the temporal stability of phthalate exposure biomarkers in pregnant women, more information is needed on how phthalate exposure and urinary metabolite concentrations vary during these time periods. We investigated the relationship between maternal pre- and post-natal urine phthalate metabolite concentrations in an effort to better understand temporal variability in concentration that may result from altered exposures or possible changes in phthalate metabolism during and after pregnancy.

Women in our study were originally recruited in the first phase of the Study for Future Families (SFFI), a multicentre pregnancy cohort study. SFFI methods are described in detail elsewhere (Swan *et al.* 2003, 2005). Participants in this portion of the study ($n = 242$) completed a questionnaire and gave a single pre-natal (at median 28.6 weeks gestation) and single post-natal (median time post-delivery 12.6 months) spot urine sample. Urinary phthalate metabolite measurements were carried out by the Division of Laboratory Sciences, National Center for Environmental Health, CDC using high-performance liquid chromatography with isotope-dilution tandem mass spectrometry (Silva *et al.* 2004; Kato *et al.* 2005). This approach allows for the simultaneous quantification in human urine of the following phthalate metabolites: MEP, MBP, mono-methyl phthalate (MMP), mono-3-carboxypropyl phthalate (MCPP), a metabolite of di-*n*-octyl phthalate and a minor metabolite of DBP, mono-isobutyl phthalate (MIBP), MBzP, MEHP and two oxidative metabolites of DEHP, MEHHP and MEOHP.

Most metabolite concentrations were above the limit of detection (LOD), which was between 0.95 and 1.07 $\mu g\, l^{-1}$ for all phthalates. Concentrations below the LOD were assigned the specific metabolite value LOD divided by the square root of 2 for statistical analyses, as has been recommended (Hornung & Reed 1990).

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Table 1. Distribution of pre- and post-natal phthalate metabolite concentrations in mothers ($\mu g\, l^{-1}$), $N = 242$. Abbreviations: MMP, mono-methyl phthalate; MCPP, mono-(3-carboxypropyl) phthalate; MEHP, mono-(2-ethylhexyl) phthalate; MEOHP, mono-(2-ethyl-5-oxohexyl) phthalate; MEHHP, mono-(2-ethyl-5-hydroxylhexyl) phthalate; MEP, mono-ethyl phthalate; MBP, mono-butyl phthalate; MBzP, mono-benzyl phthalate; MIBP, mono-isobutyl phthalate.

	pre-/post-natal visit	% above limit of detection	mean ± s.d.	median	90%	max
MMP	pre	44	2.54 ± 4.73	0.71	5.30	42.3
	post	75	3.09 ± 4.33	2.10	5.90	40.7
MCPP	pre	72	2.89 ± 4.05	1.90	5.50	33.2
	post	73	2.95 ± 2.94	1.90	6.80	21.5
MEHP	pre	78	15.4 ± 104	4.00	21.0	1560
	post	92	12.7 ± 19.5	7.15	23.6	173
MEOHP	pre	96	39.6 ± 214	10.2	42.6	2833
	post	99	33.0 ± 73.9	16.0	61.7	728
MEHHP	pre	98	50.8 ± 318	11.3	44.9	4440
	post	98	45.4 ± 106	20.4	83.1	867
MEP	pre	99	816 ± 2839	131	1340	30 528
	post	99	416 ± 1080	133	873	9520
MBP	pre	97	24.7 ± 29.6	17.2	51.8	337
	post	98	30.2 ± 43.3	19.4	68.7	499
MBzP	pre	95	20.9 ± 38.3	9.95	45.8	436
	post	99	26.6 ± 33.8	14.8	64.1	227
MIBP	pre	78	4.16 ± 4.92	2.65	9.00	39.8
	post	85	7.95 ± 17.2	3.60	14.0	179

All phthalate metabolite concentrations were logarithmically transformed to normalize distributions for statistical analysis. Phthalate concentrations were also categorized into quartiles. Using the Pearson correlation coefficient, we examined correlations between maternal pre- and post-natal concentrations of single metabolites and compared quartiles of pre-natal versus post-natal phthalate metabolites. We then compared the differences between pre- and post-natal concentrations within women using a paired Student's *t*-test. Linear regression analyses were then conducted to determine what factors predicted the difference: post-natal phthalate metabolite concentration − pre-natal phthalate metabolite concentration. We considered a number of covariates including difference in (square root) urinary creatinine between pre- and post-natal samples, difference in time between pre- and post-natal study samples, study location, maternal age and race. Results did not differ significantly between creatinine-unadjusted and creatinine-adjusted presentations of the data distribution (data not shown); therefore, we present only creatinine-unadjusted data for all analyses except the regression analyses where creatinine is included as a covariate (Barr *et al.* 2005).

Mothers in the study were primarily Caucasian (77 per cent) with 13 per cent Hispanic/Latina, and 89 per cent reported having health insurance (see supplementary electronic archive table 1). Pre- and post-natal phthalate metabolite concentrations were widely variable with large differences between mean and median concentrations (table 1). Median post-natal concentrations were consistently higher than pre-natal phthalate concentrations for each metabolite except MCPP, for which the median pre- and post-natal concentrations were equal (see supplementary electronic archive figure 1). Pearson correlation coefficients for quartiles of pre- and post-natal phthalate metabolites show that the coefficients increased as the number of days between samples decreased (table 2). Specifically, correlation coefficients for MMP, MEP, MBzP and MIBP were all above 0.40 (coefficient *p*-values <0.05) if samples were collected within 180 days of another. Results of the Student *t*-test comparisons of log pre- and post-natal phthalate metabolites within women showed significant differences ($p < 0.05$; results not shown) between means for each log metabolite except MEP (log pre-natal mean = 5.03 $\mu g\, l^{-1}$ and log post-natal mean = 4.83 $\mu g\, l^{-1}$) and MBP (log pre-natal mean = 2.71 $\mu g\, l^{-1}$ and log post-natal mean = 2.85 $\mu g\, l^{-1}$). Factors predicting the log difference between maternal pre- and post-natal phthalate concentrations were assessed using multivariate linear regression models (see supplementary electronic archive table 2). The creatinine adjusted values did not differ from the unadjusted values in correlation coefficient and *t*-test analyses, and only creatinine unadjusted values are presented. In linear regression analyses, the difference between pre- and post-natal square root creatinine was a significant predictor for the difference in pre- and post-natal phthalate concentrations for each phthalate metabolite measured. The differences in date of sample and race were not significant predictors of the difference between pre- and post-natal phthalate metabolite concentrations, and maternal age was a significant predictor for pre- and post-natal differences in MEP, MBP and MIBP. There were 19 women with extreme values of metabolite concentration (exceeding six standard deviations of the mean), but results were largely unchanged in analyses excluding these women (results not shown).

To summarize, we found that concentrations of maternal pre- and post-natal phthalate metabolites were widely variable and generally were not significantly related to one another. Pearson correlation coefficients for MEP, MMP, MBzP and MIBP increased as the number of days between sampling decreased, suggesting that pre- and post-natal samples

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Table 2. Correlation coefficients between quartile of pre- and post-natal phthalate metabolites.[a] *p-value for coefficient <0.05. Abbreviations are the same as given in table 1.

	all pre- and post-natal samples ($N = 242$)	pre- and post-natal samples taken within 365 days of one another ($N = 84$)	pre- and post-natal samples taken within 180 days of one another ($N = 28$)
MMP	0.17*	0.23*	0.44*
MCPP	0.17*	0.08	−0.04
MEHP	0.06	0.14	0.21
MEOHP	0.04	0.10	0.16
MEHHP	0.08	0.13	0.23
MEP	0.35*	0.31*	0.48*
MBP	0.18*	0.28*	0.10
MBzP	0.28*	0.42*	0.47*
MIBP	0.15*	0.26*	0.63*

[a]Pearson correlation coefficients between pre- and post-natal phthalate quartiles (unadjusted for creatinine).

are better correlated if samples are collected within three months of one another. In examining factors that predict the difference between pre- and post-natal phthalate concentrations, the difference in pre- and post-natal urinary creatinine concentration was consistently a significant factor in regression analyses. This probably represents differences in urinary dilution, metabolism, muscle breakdown and physical activity between pre- and post-natal measurements. Maternal age was a significant predictor for the difference in MEP, MBP and MIBP concentrations, which may reflect different lifestyle choices or physiologic changes due to pregnancy. Race was not a significant predictor of the difference in pre- and post-natal samples, which may reflect a generally homogeneous population, or perhaps suggest that racial factors that affect phthalate concentrations do not change from pre- to post-natal time periods.

Several studies have examined temporal variability in urine phthalate concentrations and show varying results, probably due to differences in study populations and design (Hoppin *et al.* 2002; Hauser *et al.* 2004; Fromme *et al.* 2007; Teitelbaum *et al.* 2008). In a study of 50 adult men and women over eight days, Fromme *et al.* (2007) found significant day-to-day variation, suggesting that several urine samples are needed to characterize exposures over a period of time. Hoppin *et al.* (2002) documented high correlation coefficients for adult female urine phthalate metabolite concentrations measured on two consecutive days, and, as discussed above, Hauser *et al.* (2004) found that a single urine measurement was predictive of a subject's exposure over a three-month period as estimated by tertile classifications determined from nine repeated urine samples. Our study is different in that urine samples were, on average, taken over a year apart with pregnancy and birth occurring between visits. There was also considerable variability in post-natal collection times between individuals. It may be that a single urine measurement is predictive of short-term exposures, as suggested by the current literature, but samples taken before and after major life/physiological events are not related, owing to individual characteristics such as changes in metabolism and lifestyle. Our data suggest that as the interval between samples decreases, agreement between the two increases, and within six months there were moderate correlations for several phthalate metabolites. However, post-natal urinary phthalate metabolite concentrations measured several months to a year following delivery alone would likely serve as a poor surrogate for exposure during pregnancy for most phthalates. We also found that post-natal concentrations exceeded pre-natal concentrations. However, whether this is the result of differing exposures (mother's daily routine and use of phthalate-laden products changing) or metabolism is unknown at this time. Limitations of our study include the single (as opposed to repeated) samples taken during the two different study periods and the inability to measure changes in lifestyle and metabolism in mothers between these periods. Additional research should document phthalate exposures and urine concentrations during and after pregnancy, as well as changes in metabolic processes to determine potential predictors of the difference in phthalate concentrations between pre- and post-natal urine samples.

(c) *Human developmental studies*

Human studies that have investigated associations between phthalate exposure and adverse pregnancy outcomes or developmental effects are presented in table 3. As compared with the large body of evidence in laboratory animals documenting phthalate reproductive or developmental toxicity, these associations have been immensely understudied in humans to date. In one Italian study of 84 newborns that examined gestational age in relation to phthalate exposure (Latini *et al.* 2003), MEHP in the cord blood of the newborns was associated with shorter gestational age at delivery (odds ratio for absence of detectable MEHP in cord blood associated with a 1-week increase in gestational age = 1.50, 95% CI 1.01–2.21). To date, most human studies investigating reproductive or developmental health outcomes associated with foetal or infant exposure to phthalates have been limited to males, and there is indirect evidence suggesting that certain phthalates may impart anti-androgenic effects in the perinatal period (see §2(d) below). In a study of phthalate concentrations in breast milk and serum hormone levels in three-month-old male offspring, Main *et al.* (2006*b*) identified a significant inverse association between MBP, the active metabolite of DBP and free testosterone levels (determined by total testosterone: sex hormone binding globulin ratio) in male offspring. They also reported positive associations between MEP, MMP and MBP with luteinizing hormone (LH): free testosterone ratio, which is a measure of Leydig cell function. There were also positive relationships between mono-isononyl phthalate (MiNP) and LH, and between MEP, MBP and sex hormone binding globulin (SHBG). However, phthalate levels were

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Table 3. Health outcomes in infants and children associated with phthalate concentrations in biological or environmental samples.[a] Adapted from Swan (2008). Abbreviations: BBzP, butyl benzyl phthalate; DBP, dibutyl phthalate; DEHP, di(2-ethylhexyl) phthalate; MMP, mono-methyl phthalate; MCPP, mono-(3-carboxypropyl) phthalate; MEHP, mono-(2-ethylhexyl) phthalate; MEOHP, mono-(2-ethyl-5-oxohexyl) phthalate; MEHHP, mono-(2-ethyl-5-hydroxylhexyl) phthalate; MEP, mono-ethyl phthalate; MBP, mono-butyl phthalate; MBzP, mono-benzyl phthalate; MiNP, mono-isononyl phthalate.

system	timing of exposure	sex	outcome	phthalate or metabolite (measured in urine unless otherwise noted)	references
reproductive	pre-natal	males/ females	shorter gestational age at birth	MEHP (in cord blood)	Latini *et al.* (2003)
	pre-natal (mean age 12.6 months at exam)	males	shorter anogenital distance	MEHP, MEOHP, MEHHP, MEP, MBP	Swan (2008)
			reduced penile size	MEHP	
			incomplete testicular descent	MEHP, MEHHP, MEOHP	
	lactation (mean age 3 months)	males	increased SHBG	MEP, MBP	Main (2006*b*)
			increased LH/free T	MMP, MEP, MBP	
			increased LH	MiNP	
			decreased free T	MBP	
	early childhood	females	premature thelarche	DEHP (in serum)	Colon *et al.* (2000)
respiratory, allergy and asthma	childhood	males/ females	rhinitis and eczema	BBzP (in house dust)	Bornehag *et al.* (2004)
			asthma	DEHP (in house dust)	Bornehag *et al.* (2004)
			wheezing, rhinitis and eczema	DEHP (in house dust)	Kolarik (2008)

[a]Associations significant at $p < 0.05$.

not associated with the occurrence of cryptorchidism in a nested case–control analysis.

Earlier evidence for DEHP and/or MEHP as an endocrine disruptor was reported in a study of 41 Puerto Rican girls (aged six months to 8 years; median age 20 months) with premature thelarche, which is defined as premature breast development in girls younger than 8 years of age with the absence of other physical signs of puberty onset. In the study, Colon *et al.* (2000) compared serum levels of phthalate diesters (and MEHP) between the thelarche cases and 35 paediatric controls (median control age 46 months). Among the phthalates measured, the largest differences between cases and controls were found for serum DEHP concentration, which were 450 and 70 ng ml^{-1}, respectively ($p < 0.05$). However, there are concerns regarding several aspects of the study design, including potential sample contamination by phthalate diesters during the collection, transport, storage, processing or analysis of the serum samples.

Other developmental effects of phthalate exposures may include pulmonary system effects: allergies, rhinitis, asthmatic reactions and direct toxicity. Bornehag *et al.* (2004) conducted a nested case–control study comparing phthalate concentrations in indoor dust in 198 children (ages 3–8 years) with allergic/asthmatic symptoms to 202 controls without symptoms in Sweden. They found significantly higher concentrations of butyl benzyl phthalate (BBzP) in the house dust of cases as compared with that of controls. Dust concentrations of BBzP were also associated with allergic and eczematous symptoms in a dose-dependent manner, as were DEHP dust concentrations and asthmatic symptoms (Bornehag *et al.* 2004). Another study in Bulgarian children produced similar results, where increased house dust DEHP concentrations were related to increased wheezing symptoms in a dose–response manner (Kolarik *et al.* 2008). These authors also found increased wheezing and rhinitis associated with BBzP in house dust, but these findings were not statistically significant (Kolarik *et al.* 2008). Because of the cross-sectional nature of these studies, we cannot determine if concentrations of phthalates in the home are also involved in the development of atopic diseases. In a study conducted in pre-term infants with polyvinyl chloride (PVC) respiratory tubing, Roth *et al.* (1988) found these infants to have clinically and radiologically confirmed hyaline membrane disease, which was hypothesized to be caused by the DEHP exposure. Di(2-ethylhexyl) phthalate was also found in the lung tissue of one of the patients who died of a pneumothorax soon after birth (Roth *et al.* 1988). These findings have not been replicated, but several authors have suggested that PVC medical tubing may be a likely source of respiratory health effects in young neonates (Hill *et al.* 2001, 2003; Tickner *et al.* 2001). One study of adults found an inverse relationship between urinary MEP and MBP and pulmonary function (Hoppin *et al.* 2004), but because urine markers reflect all routes of exposure and may not be a good measure of inhaled phthalate concentrations, it is difficult to interpret these results. Given these findings, further research needs to examine route- or target-specific phthalate biomarkers of exposure and potential pathophysiologic

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changes relating to the development of allergic and asthmatic diseases and pulmonary function in children.

(d) *Anogenital distance and the phthalate syndrome*

Numerous animal studies report that several phthalates (particularly DEHP and DBP) induce a marked reduction in foetal testosterone and insulin-like growth factor-3, resulting in a syndrome of male reproductive abnormalities (Foster 2006; Gray *et al.* 2006). In addition to shortened anogenital distance (AGD), which is a sensitive and non-invasive measure of potential androgen deficiency during foetal development in rodents and humans, other abnormalities include hypospadias, cryptorchidism and malformations of the epididymis, vas deferens, seminal vesicles and prostate; together they comprise the 'phthalate syndrome' (Gray *et al.* 2006). Prior to 2005, only a single study had evaluated AGD in human males (Salazar-Martinez *et al.* 2004), and none had examined AGD in relation to environmental chemical exposure. Swan *et al.* recently examined AGD and other genital parameters in male human infants in relation to their mother's phthalate exposure. The details of study methods are presented in a 2005 publication (Swan *et al.* 2005) that includes an analysis of the first 85 boys in this population, among whom genital measurements (including AGD) and concentrations of phthalate metabolites in maternal pre-natal urine samples were available. Follow-up analysis was subsequently conducted on a larger sample of boys, including 166 exams on 106 boys for whom genital measurements and pre-natal phthalate exposure measures were available (Swan 2008). We summarize those results here.

The male infants in the study were examined initially 2–24 months after birth (mean 12.8 months of age) using a detailed examination of breast and genitals developed specifically for this study. Boys' examinations included a description of the testes and scrotum, location of testes, measurement of the penis and AGD. AGD was measured in boys from the centre of the anus to the cephalad (towards the head) base of the penis and averaged 72.4 mm. For each infant, the expected weight for age (weight percentile) was determined by standard curves (http://www.cdc.gov/growthcharts/). AGD was then adjusted for age and weight percentile and examined in general linear models and categorical analyses. In these 106 mother–son pairs, maternal urinary concentrations of the three measured metabolites of DEHP were significantly and inversely related to AGD (p-values = 0.001, 0.002 and 0.017 for MEOHP, MEHHP and MEHP, respectively), as were MEP (p-value = 0.005) and MBP (p-value = 0.049).

For each boy, testicular descent was classified as incomplete if one or both testicles were not graded as normal (fully descended) or normal retractile. Each of the three measured DEHP metabolites were related to risk of incomplete testicular descent (p-values = 0.040, 0.054 and 0.048 for MEOHP, MEHHP and MEHP, respectively). AGD was correlated strongly with penile width and, to a lesser extent, with penile length ($p = 0.0003$ and 0.059, for width and length, respectively). In a mixed model controlling for age and weight percentile, penile width (but not length) was significantly associated with concentration of the sum of DEHP metabolites and the monoester metabolite (MEHP). No other metabolites were associated with penile size.

This study demonstrates inverse associations between AGD, penile width and testicular descent and one or more metabolites of DEHP, as well as between AGD and both MEP and MBP. These findings suggest that AGD is a marker for insufficient foetal androgenization and suggest that low-dose phthalate exposure may affect several markers of human male genital development.

(e) *Human studies in adults*

(i) *Semen quality*

Studies that have explored phthalate effects in adults are presented in table 4. A substantial proportion of the limited human studies on phthalates and reproductive effects in adults have been focused on investigating associations with altered semen quality. In an early study, Murature *et al.* (1987) measured DBP concentrations in the cellular fractions of ejaculates from 21 university students and found an inverse relationship with sperm concentration. The study had a number of limitations, including a small number of participants, the failure to adjust for potential confounders and measurement of the parent compound, which may be subject to sample contamination. In another small study, conducted in India, Rozati *et al.* (2002) studied 21 infertile men with poor semen quality and 32 'control' men with normal semen parameters. Phthalate esters were measured in seminal plasma and the results were reported as the sum of a mixture of dimethyl phthalate, DEP, DBP, BBzP, DEHP and di-*n*-octyl phthalate. The concentration of phthalates was inversely correlated with sperm morphology but not correlated with ejaculate volume, sperm concentration or motility. In this study, as in the Murature study, the measurement of phthalate diesters raises concern with sample contamination from the ubiquitous presence of the diester in the environment.

More recently, a larger study using urinary levels of phthalate metabolites was conducted by Duty *et al.* (2003*a*), with follow-up analysis reported by Hauser *et al.* (2006). Study subjects consisted of male partners of subfertile couples that presented to an infertility clinic in Massachusetts, USA. At the time of the clinic visit, one sample of semen and urine were collected. In the initial study, there were dose–response relationships (after adjusting for age, abstinence time and smoking status) between MBP and below World Health Organization (WHO) reference value sperm motility and sperm concentration among 168 men (Duty *et al.* 2003*a*). There was also a dose–response relationship between MBzP (the primary hydrolytic metabolite of BBzP) and below WHO reference value sperm concentration. In a recent follow-up study including these 168 men, plus an additional 295 men newly recruited into the study, Hauser *et al.* (2006) confirmed the associations between MBP and increased

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Table 4. Health outcomes in adults associated with phthalate concentrations in biological samples.[a] Adapted from Swan (2008). Abbreviations are the same as given in table 3.

system	sex	outcome	phthalate or metabolite (measured in urine unless otherwise noted)	references
reproductive	males	increased sperm DNA damage	MEP, MEHP	Hauser *et al.* (2007)
		increased sperm DNA damage	MEP	Duty *et al.* (2003*b*)
		decreased sperm motility	MBP	Duty *et al.* (2003*a*); Hauser *et al.* (2006)
		decreased sperm concentration	MBP, MBzP	Duty *et al.* (2003*a*); Hauser *et al.* (2006)[b]
	males	decreased sperm motility	DBP and DEHP in semen samples	Zhang (2006)
	males	decreased free T and increased LH/free T	MBP	Pan *et al.* (2006)
	males	decreased FSH	MBzP	Duty *et al.* (2005*b*)[c]
	males	decreased motility, reduced LH	MEP	Jonsson *et al.* (2005)
respiratory, allergy and asthma	males	decreased pulmonary function	MEP, MBP	Hoppin *et al.* (2004)
metabolic	males	increased waist circumference	MBzP, MEHHP MEOHP, MEP	Stahlhut *et al.* (2007)
		altered thyroid hormone	MBP, MBzP, MEP	
thyroid	males	levels (decreased T3 and T4)	MEHP	Meeker *et al.* (2007*b*)

[a]Associations significant at $p < 0.05$.
[b]In Hauser *et al.* (2006), p-value for MBzP association was 0.13.
[c]In Duty *et al.* (2005*b*), p-value for association between MEHP and testosterone was 0.10.

odds of below-reference sperm concentration and motility. The relationships appeared to follow dose-dependent patterns, where greater odds ratios were calculated among increasing phthalate metabolite quartiles. However, there was only a suggestive association between the highest MBzP quartile and low sperm concentration ($p = 0.13$), which was not fully consistent with the results of the preliminary analysis (Duty *et al.* 2003*a*).

In a recently published study from Sweden, Jonsson *et al.* (2005) recruited 234 young Swedish men at the time of their medical conscript examination. Each man provided a single urine sample used to measure concentrations of MEP, MEHP, MBzP, MBP and phthalic acid. In contrast to the US study, in the Swedish study there were no relationships of MBP or MBzP with any of the semen parameters. MEHP was also not associated with any of the semen parameters, but men in the highest quartile for MEP had fewer motile sperm and more immotile sperm than men in the lowest MEP quartile. Contrary to their hypothesis, phthalic acid was associated with improved function as measured by more motile sperm and fewer immotile sperm. Phthalic acid is a non-specific marker of phthalate exposure, formed as the result of the hydrolysis of any of the phthalates measured. Interactions between urinary phthalate levels and PCB 153 (measured previously in serum samples from these men) were assessed by including an interaction term in the models. There was no evidence of multiplicative interactions between PCB 153 and any of the phthalates with the reproductive markers (data were not shown). This is in contrast to a previous study by Hauser *et al.* (2005), where they found interactions of MBP and MBzP with congener PCB 153 in relation to sperm motility.

Although the Swedish study had some similarities to the US study, in that they were both cross-sectional studies in which a single urine and semen sample were collected, there were also many important differences. One of the primary differences was in the age of the study population and the method of recruitment. The Swedish study population consisted of young men (median age 18 years, range 18–21 years) undergoing a medical examination before military service. Since approximately 95 per cent of young men in Sweden undergo the conscript examination, these young men reflected the general population of young Swedish males. In contrast, in the US study, the median age of the men recruited from an infertility clinic was 35.5 years and ranged from 22 to 54 years. None of the men from the infertility clinic were 21 years of age or younger. The differences across studies in the ages and source of the men may account for some of the differences in results between studies. For instance, it is unclear whether men presenting to an infertility clinic are more 'susceptible' to reproductive toxicants, including phthalates, than men from the general population. Furthermore, it is also unclear whether middle-aged men, as compared to young men, are more susceptible to reproductive toxicants because of an age-related response to the toxicant. Other differences across studies include major differences in participation rates (14% in the Swedish

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study and 65% in the US study) and differences in the analytical methods used to measure urinary phthalate metabolites, where the method in the US study was much more sensitive than that used in the Swedish study.

In the Massachusetts (USA) study, semen samples from 379 men were also cryogenically frozen and sperm cells later analysed for DNA damage using the neutral comet assay (Hauser *et al.* 2007). Sperm DNA damage measurements included comet extent (CE), percentage of DNA in tail (tail%) and tail distributed moment (TDM). In multivariate linear regression models adjusted for age and smoking, significant positive associations were found for at least one of the three DNA damage measures with MEP (CE, TDM), MBP (tail%), MBzP (CE, TDM) and MEHP (tail%). For MEP, the significant association with CE and TDM confirmed previous findings among an earlier and smaller subset from the same study population (Duty *et al.* 2003*b*). Another interesting finding was that MEHP was strongly associated with all three DNA damage measures after adjustment for the oxidative DEHP metabolites, which may serve as phenotypic markers of DEHP metabolism to 'less toxic' metabolites and lower susceptibility to exposure-related effects compared with those individuals with low concentrations of oxidative DEHP metabolites relative to MEHP concentration (Hauser 2008). Metabolism of phthalates depends on the size and structure of the diester, and can occur via two steps: phase I (e.g. hydrolysis, oxidation) followed by phase II (conjugation) (Frederiksen *et al.* 2007). Since the monoester metabolite may be the more bioactive form of the phthalate, individuals who are predisposed to form and retain more monoester may have a heightened sensitivity to phthalate exposure.

In summary, the epidemiologic data on semen quality and/or sperm cell integrity in relation to phthalate exposure remain limited and inconsistent. Additional studies are critically needed to help elucidate possible explanations for differences across studies, and most importantly, to address whether phthalate exposure alters semen quality, sperm function and male fertility.

(ii) *Other reproductive/endocrine effects*

Several human studies have investigated associations between exposure to phthalates and circulating hormone levels. In a study of workers producing PVC flooring with high exposure to DEHP and DBP, urinary concentrations of metabolites of these phthalates were inversely associated with free testosterone levels (Pan *et al.* 2006). A report on 295 men from the Massachusetts (USA) infertility clinic study found a suggestive inverse association between MEHP and testosterone, along with a positive association between urinary MBP and inhibin B (a glycoprotein hormone produced by the gonads that has an inhibitory effect on pituitary FSH production), and an inverse association between urinary MBzP and FSH (Duty *et al.* 2005*b*). However, the significant results for MBP and MBzP and hormone levels were in patterns inconsistent with the authors' hypotheses. It is interesting to note that although MEHP concentrations in the Massachusetts study were several orders of magnitude lower than those measured in the exposed Chinese workers (Pan *et al.* 2006), the evidence for decreased testosterone in relation to DEHP/MEHP was consistent between the two studies. It is also interesting to note that the inverse association between MBP and testosterone in the study of exposed Chinese workers (Pan *et al.* 2006) appears to be consistent with the male infant studies described earlier, where MBP concentrations were inversely associated with anogenital index (a measure of androgen activity) and free testosterone (Swan *et al.* 2005; Main *et al.* 2006*b*). On the other hand, the study of 234 young Swedish men found an inverse association between urinary MEP and LH but no association between MEP, MBP, MEHP or other phthalate metabolites in urine and FSH, testosterone, oestradiol or inhibin B (Jonsson *et al.* 2005).

Owing to the documented anti-androgenic effects of certain phthalates in animal models, and recent observations that low testosterone in adult males may be associated with an increased prevalence of obesity and type 2 diabetes (Ding *et al.* 2006; Selvin *et al.* 2007), Stahlhut *et al.* (2007) explored the relationship between phthalate exposure and waist circumference in a large cross-sectional study carried out among a subset of adult male participants in the 1999–2002 US National Health and Nutrition Examination Survey (NHANES). The authors reported significant associations between urinary phthalate monoester concentrations (MBzP, MEHHP, MEOHP and MEP) and increased insulin resistance (measured through homeostatic model assessment), and positive associations between MBP, MBzP and MEP and waist circumference. These findings provide preliminary evidence of a potential contributing role for phthalates in the overall population burden of insulin resistance, obesity and related clinical conditions, but additional studies are needed.

The potential for phthalates to affect thyroid function has been demonstrated in animal studies, but human studies are limited to two recent investigations: one within the Massachusetts (USA) male infertility clinic study (Meeker *et al.* 2007*b*) and another among pregnant Taiwanese women in their second trimester (Huang *et al.* 2007). In the Massachusetts study, phthalate metabolite concentrations were measured in urine and thyroid hormones were measured in serum from 408 men. MEHP was inversely associated with free T4 and total T3, but was not associated with thyroid-stimulating hormone (TSH). The inverse association between MEHP and free T4 became stronger when also taking into account the concentrations of oxidative DEHP metabolites that were positively associated with free T4. As with the findings from the study of sperm DNA damage (Hauser *et al.* 2007), these results may reflect metabolic susceptibility to the adverse effects of MEHP among individuals who less efficiently oxidize DEHP and/or MEHP (Meeker *et al.* 2007*b*; Hauser 2008). Among 76 pregnant Taiwanese women, Huang *et al.* (2007) reported an inverse association between MBP and both total and free levels of T4. Unlike the study among US men, they did not find an inverse association with MEHP, but there were considerable

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differences between the design of the two studies. In addition to having a smaller study size and a vastly different study population, the Taiwanese study also did not take into account concentrations of oxidative DEHP metabolites, which served to strengthen the associations between MEHP and thyroid hormones in the US study. More study is needed on the association between phthalate exposure and thyroid function, which plays an important role in many human systems including reproduction and foetal neurodevelopment.

3. BISPHENOL A

(a) *Exposure*

BPA is used in a variety of consumer products, including epoxy resins that are used to line food cans (Kang *et al.* 2003), polyester–styrene (Factor 1998) and polycarbonate plastics used for baby bottles and other containers (Brede *et al.* 2003). BPA-containing resins and plastics are also used in some dental sealants (Sasaki *et al.* 2005) and fillings (Joskow *et al.* 2006), adhesives, protective coatings, flame retardants (Samuelsen *et al.* 2001) and water storage tanks and supply pipes (Bae *et al.* 2002). BPA is a polymer, but degrades into its monomeric form over time, a process that can be accelerated by heat exposure (Brotons *et al.* 1995). The monomeric form can leach from its source into adjacent materials such as into water or into the food products from the lining of the can. Several studies have demonstrated detectable levels in packaged foods that were contained in wrapping or cans coated with BPA (Lopez-Cervantes & Paseiro-Losada 2003). The majority of human exposure is via ingestion of food-containing BPA (Kang *et al.* 2006; Vandenberg *et al.* 2007).

Widespread exposure to BPA among the US population has recently been documented. In a subset of the 2003–2004 NHANES, 93 per cent of urine samples had detectable levels of BPA (Calafat *et al.* 2008). The subset included 2517 participants aged 6 years and older, with a mix of urban and rural residences. Human studies have also measured detectable levels of BPA in a variety of human body fluids and some tissues. Of potential concern to reproductive and developmental health endpoints is the presence of BPA in follicular fluid and amniotic fluid (Ikezuki *et al.* 2002), umbilical cord blood (Schonfelder *et al.* 2002) and breast milk (Sun *et al.* 2004).

Adult humans metabolize BPA via the hepatic glucuronidation pathway, and the biologic half-life of BPA is approximately 6 h, with nearly complete urinary excretion in 24 h (Volkel *et al.* 2002). Therefore, urinary BPA levels primarily reflect exposures that occurred ≤ 1 day preceding the collection of the urine specimen, and, like with phthalates, information on the ability of urinary biomarkers to predict individual exposure over longer periods of time is needed to properly design human studies. In a recent study investigating temporal variability of urinary BPA concentrations among 82 men and women, Mahalingaiah *et al.* (2008) reported high within-individual variability but concluded that a single sample may be relatively predictive of BPA concentration over a period of weeks or months in an epidemiologic study that categorizes subjects into broad BPA exposure groups (e.g. tertiles) (Mahalingaiah *et al.* 2008). However, if feasible, a second urine sample from individuals in a study would likely improve the sensitivity of exposure categories.

(b) *Human health effects*

BPA has been shown to have oestrogenic properties in studies published as early as 1936 (Dodds & Lawson 1936). These findings have been confirmed in a large number of subsequent *in vitro* and animal studies, along with evidence for other biological activities such as effects on thyroid function (Richter *et al.* 2007; Wetherill *et al.* 2007). Despite the high level of concern for health effects related to BPA exposure, and documented widespread human exposure to BPA, human epidemiologic studies of adverse health outcomes in association with BPA exposure are severely limited. BPA levels in blood have been associated with a variety of conditions in women, but studies have been small in size and have provided limited details on subject selection criteria. In one epidemiologic study, serum BPA levels were reported to be associated with recurrent miscarriage (Sugiura-Ogasawara *et al.* 2005). Compared with 32 non-parous women without fertility problems, the study reported that the mean serum BPA concentration was three times higher in 45 women with a history of three or more consecutive first-trimester miscarriages. Among 35 of the women who became pregnant there was some suggestive evidence of higher BPA concentrations among the women who subsequently miscarried again compared with women who went on to have a successful pregnancy, though the median BPA concentrations were the same between the two groups. Another Japanese study reported higher maternal serum BPA concentrations among 48 women carrying foetuses with an abnormal karyotype compared to 200 women carrying foetuses with a normal karyotype (Yamada *et al.* 2002).

Two small human studies have investigated exposure to BPA and hormone levels, where statistically significant positive correlations were found between BPA concentrations in serum and circulating total and free testosterone levels in both men and women (Takeuchi & Tsutsumi 2002; Takeuchi *et al.* 2004). These investigators also reported that women with polycystic ovary syndrome (PCOS) had higher serum levels compared with women without PCOS (Takeuchi & Tsutsumi 2002; Takeuchi *et al.* 2004). Another small study among men occupationally exposed to BPA during the application of epoxy resins found an inverse association between BPA and FSH (Hanaoka *et al.* 2002). Finally, a study among 172 adults reported a positive association between urinary BPA concentration and frequency of sister chromatid exchange measured in peripheral lymphocytes, but not between urinary BPA and self-diagnosed endocrine disorders (Yang *et al.* 2006). To summarize, suggestive but limited evidence exists for an association between BPA exposure and adverse human health endpoints, and human studies that have appeared in the literature thus far have suffered from

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a number of limitations. Given the widespread human exposure to BPA and the increasing concern for its potential to adversely affect health through various mechanisms, there is a clear need for more epidemiological research.

4. POLYBROMINATED DIPHENYL ETHERS

(a) *Exposure*

PBDEs are flame-retardant chemicals that are added to a variety of consumer products, including textiles, thermoplastics used in electronics (e.g. televisions, computers) and products containing polyurethane foam (e.g. mattresses, upholstered furniture) to make them difficult to burn (ATSDR 2004). Additive flame retardants like PBDEs are not chemically but physically combined with polymers at levels ranging from 5 to 30 per cent by weight, creating the possibility for them to leach out of the treated materials into the surrounding environment (EU 2001). In 2001, the worldwide demands for technical grade PBDEs were estimated to be 67 000 metric tonnes, with the Americas accounting for about half of that amount (ATSDR 2004). Despite bans or voluntary discontinuation of the production of certain commercial PBDE formulations in Europe and the USA, the general population continues to be exposed due to their persistence in the environment. PBDEs are highly hydrophobic and thus are stored in biological systems and bioaccumulate up the food chain, although bioaccumulation is thought to be inversely related to the degree of PBDE bromination (ATSDR 2004). The primary routes of exposure to humans are probably ingestion of contaminated foods and inhalation of indoor air or ingestion or dermal uptake of house dust containing PBDEs released from electrical appliances and furniture (ATSDR 2004; Webster *et al.* 2005; Allen *et al.* 2007; Lorber 2008). Recent evidence suggests that exposure to PBDE among women during pregnancy likely results in foetal exposure, though the rate of placental transport may vary by congener (Gomara *et al.* 2007; Schecter *et al.* 2007). For infants, consumption of breast milk is probably a primary source of exposure to the lower brominated PBDE congeners due to their lipophilicity (Schecter *et al.* 2003). Human biomonitoring studies of blood, breast milk or adipose tissue samples have shown high geographic variability in exposure levels, with concentrations among individuals in the US orders of magnitudes higher than those found among European and other populations studied throughout the world (ATSDR 2004; Schecter *et al.* 2005). In addition, there is much greater variability in PBDE levels measured in biological matrices compared with other persistent organic pollutants (Birnbaum & Cohen Hubal 2006), which may reflect continuing and direct exposures among subsets of the population.

For assessing human PBDE exposure in epidemiological studies, measuring the various PBDE congeners in blood is the preferred method (with the exception of certain study designs or populations, such as breast milk in studies of breast-feeding infants), and it provides a rather temporally stable exposure measure due to the persistent and bioaccumulation properties of PBDEs. Estimated half-lives for PBDEs in human serum range from weeks to months, depending on the congener, and half-lives in adipose tissue and for terminal elimination from the body are of the order of several years (Birnbaum & Cohen Hubal 2006). Thus, unlike urinary BPA and phthalate metabolite measures, a single blood PBDE measure is likely to be highly predictive of an individual's average PBDE level over a span of months or years.

(b) *Human health effects*

While a number of brominated flame retardants, including PBDE, are considered pervasive endocrine-disrupting environmental contaminants of concern, very few human studies have explored the associations between exposure and evidence for reproductive or developmental effects. Only one human study on data (Hagmar *et al.* 2001) has explored associations between non-occupational PBDE exposure and thyroid function or thyroid hormone levels despite a growing body of evidence for these effects in animal and *in vitro* studies (Zhou *et al.* 2002; Costa & Giordano 2007; Kuriyama *et al.* 2007; Tseng *et al.* 2008). In a study of 110 adult males from Sweden and Latvia, plasma concentrations of PBDE 47 were inversely and significantly associated with TSH, but not with T3 or T4, after adjusting for age (Hagmar *et al.* 2001). Conversely, a longitudinal study among a very small number of workers ($n = 11$) at an electronics recycling facility who were thought to be exposed to low levels or PBDEs reported no associations between exposure to specific congeners or the sum of PBDE congeners measured in the study and T3, T4 or TSH (Julander *et al.* 2005). However, PBDE 153 was suggestively associated with T4 (p-value $= 0.08$), and several statistically significant relationships between specific PBDE congeners and thyroid hormones were observed within individual workers over the study period. An earlier workplace study reported four cases of hypothyroidism among 35 exposed workers involved with the manufacture of PBDE 209 compared with no cases among 89 unexposed workers (Bahn *et al.* 1980).

A small number of recent human studies have assessed other reproductive and developmental outcomes in association with PBDE exposure, as animal studies have demonstrated PBDEs are anti-androgenic (Stoker *et al.* 2005) and that the foetal period may be one of high sensitivity to PBDE exposure (Lilienthal *et al.* 2006). Main *et al.* (2007) recently compared PBDE concentrations in placenta and breast milk between Danish and Finnish boys with and without cryptorchidism, and also explored associations between PBDE and reproductive hormone levels in the boys. They reported that PBDE concentrations in breast milk, but not placenta, were significantly higher in boys with cyrptorchism than in controls. They also reported a positive correlation between the sum of 14 PBDE congeners and LH. In a Swedish case–control study of testicular cancer among men and their mothers who were recruited in the years 1999–2000, there was an increased odds (odds

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ratio = 3.5; 95% CI 1.1–11) for testicular cancer among men whose mothers were above the 75th percentile for blood PBDE concentrations (sum of congeners 47, 99 and 153) (Hardell *et al.* 2006). However, this association was not found when basing exposure on PBDE concentrations measured in the men. This discrepancy suggests either a chance finding when using the mother's PBDE levels or perhaps a biological mechanism traced back to *in utero* exposures. Finally, despite a small study size ($n = 20$), Chao *et al.* (2007) recently reported statistically significant associations between concentrations of PBDE congeners in breast milk and adverse birth outcomes, including reduced birth weight, birth length, chest circumference and Quetelet's index (kg m^{-2}), as well as suggestive associations between PBDE levels and altered menstrual cycle characteristics. The PBDE levels measured in breast milk in this study and in the study of cryptorchidism (Main *et al.* 2007) were similar to levels reported in Europe and East Asia, but much lower than those measured in the USA (Schecter *et al.* 2003; Chao *et al.* 2007).

5. CONCLUSIONS AND FUTURE RESEARCH NEEDS

A number of chemicals used in plastics for property enhancement are emerging environmental contaminants of concern (see also the discussion in Oehlmann *et al.* (2009), Koch & Calafat (2009), Thompson *et al.* (2009*b*) and this paper). Although the epidemiological data on the plastic additives described here suggest that there may be associations with altered endocrine function and reproductive or developmental effects, the number of human studies is currently limited and the quantity and quality of the data available for the different compounds are varied. Also, for some of the more studied associations, such as between phthalates and semen quality, the data across studies are not consistent. This may be due to small study sizes and lack of statistical power or differences in study design, study populations, exposure assessment strategies, exposure levels, exposure sources, exposure routes, multiple/competing physiologic mechanisms, analytical approaches and potential confounding variables considered in the statistical analysis (e.g. age, BMI, season). The limited human data, and in certain instances inconsistent data across studies, highlight the need for further epidemiological research on these classes of chemicals. Most studies to date have been cross-sectional in nature. Future longitudinal studies are needed to explore the temporal relationship between exposure to plastic additives and adverse reproductive and developmental outcomes to provide more information on whether these relationships may be causal in nature. Owing to the complex nature of the endocrine system, studies should evaluate not only individual hormone levels but also the ratios between relevant hormones (e.g. LH : testosterone ratio in males as a marker for Leydig cell function) that may help provide clues to the biological mechanisms of xenobiotic activity in humans.

Researchers face a number of challenges that need to be addressed to further our understanding of the relationship between plastic additives and adverse human health effects. One future challenge includes the shifts in exposure levels among populations over time caused by the ever-changing patterns of production and use of these compounds. Another challenge is to understand how simultaneous co-exposures to these chemicals may affect endocrine function. It is well known that humans are exposed to all these compounds simultaneously, and to many other chemicals. However, most studies to date have only addressed single chemicals or classes of chemicals, and there are limited data on the interactions between chemicals within a class or across classes. Chemicals may interact additively, multiplicatively or antagonistically in what is commonly referred to as the 'cocktail effect'. The human health risks of exposure to chemical mixtures are much understudied. Despite these challenges, evolving and innovative technologies designed to improve the assessment of human exposure and intermediate biological markers of effect should provide enhanced opportunities for improving our understanding of the relationship between these environmental chemicals and reproductive and developmental health. Innovations include improved biomarkers of exposure, more sophisticated statistical methods that deal with multiple exposures simultaneously and sensitive new measures of intermediate alterations in human endocrine function, reproductive health and foetal/child development.

More information is required on biological mechanisms of plastic additives in humans as well as the clinical and public health consequences of changes of intermediate markers of effect observed in human studies. For example, to date, in most studies that have reported statistically significant hormone alterations attributed to environmental and occupational exposures, the actual degree of hormone alteration has been considered subclinical. However, much remains unknown as to whether hormone changes currently considered subclinical may be associated with increased risk of adverse systemic effects in the long term. Furthermore, although seemingly subtle, small changes in hormone levels resulting from exposure may be of public health importance when considering the prevalence of exposure to plastic additives and EDCs among entire populations. Finally, human research is needed on potential latent and transgenerational effects (e.g. epigenetic modifications) of exposure to plastic additives and other EDCs, including the possibility of environmentally linked foetal origins of adult diseases, as well as genetic, metabolic, demographic or environmental characteristics resulting in increased individual susceptibility to adverse health effects following exposure.

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A critical analysis of the biological impacts of plasticizers on wildlife

Jörg Oehlmann, Ulrike Schulte-Oehlmann, Werner Kloas, Oana Jagnytsch, Ilka Lutz, Kresten O. Kusk, Leah Wollenberger, Eduarda M. Santos, Gregory C. Paull, Katrien J. W. Van Look and Charles R. Tyler

Phil. Trans. R. Soc. B 2009 **364**, 2047-2062
doi: 10.1098/rstb.2008.0242

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PHILOSOPHICAL TRANSACTIONS OF THE ROYAL SOCIETY B

Phil. Trans. R. Soc. B (2009) **364**, 2047–2062
doi:10.1098/rstb.2008.0242

Review

A critical analysis of the biological impacts of plasticizers on wildlife

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This review provides a critical analysis of the biological effects of the most widely used plasticizers, including dibutyl phthalate, diethylhexyl phthalate, dimethyl phthalate, butyl benzyl phthalate and bisphenol A (BPA), on wildlife, with a focus on annelids (both aquatic and terrestrial), molluscs, crustaceans, insects, fish and amphibians. Moreover, the paper provides novel data on the biological effects of some of these plasticizers in invertebrates, fish and amphibians. Phthalates and BPA have been shown to affect reproduction in all studied animal groups, to impair development in crustaceans and amphibians and to induce genetic aberrations. Molluscs, crustaceans and amphibians appear to be especially sensitive to these compounds, and biological effects are observed at environmentally relevant exposures in the low $\mathrm{ng\,l^{-1}}$ to $\mu\mathrm{g\,l^{-1}}$ range. In contrast, most effects in fish (except for disturbance in spermatogenesis) occur at higher concentrations. Most plasticizers appear to act by interfering with the functioning of various hormone systems, but some phthalates have wider pathways of disruption. Effect concentrations of plasticizers in laboratory experiments coincide with measured environmental concentrations, and thus there is a very real potential for effects of these chemicals on some wildlife populations. The most striking gaps in our current knowledge on the impacts of plasticizers on wildlife are the lack of data for long-term exposures to environmentally relevant concentrations and their ecotoxicity when part of complex mixtures. Furthermore, the hazard of plasticizers has been investigated in annelids, molluscs and arthropods only, and given the sensitivity of some invertebrates, effects assessments are warranted in other invertebrate phyla.

Keywords: bisphenol A; dibutyl phthalate; diethylhexyl phthalate; dimethyl phthalate; butyl benzyl phthalate; endocrine disruption

1. INTRODUCTION

The history of phthalate-based plasticizers and bisphenol A (BPA) dates back to the 1920s and 1930s, respectively. Phthalates have been applied as polyvinyl chloride (PVC) additives since 1926, but were also used for healthcare purposes as insect repellents and cercaricides (Ferguson *et al.* 1946). The development of BPA had a very different genesis, driven primarily by the search for synthetic oestrogens for clinical use (Cook *et al.* 1933). Dodds & Lawson (1936, 1938) demonstrated the oestrogenic activity of BPA in ovariectomized rats, but the formulation of a more effective synthetic oestrogen, diethylstilboestrol, at around the same time rendered BPA redundant as a clinical oestrogen (Dodds *et al.* 1938). In the 1950s, BPA was rediscovered when a Bayer chemist, Hermann Schnell, reacted BPA with phosgene to produce polycarbonate plastic, and its use in plastics has subsequently become its primary commercial application.

The emollient properties of phthalates and polymerization of BPA have driven their widespread use in the production of plastics (Thompson *et al.*

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One contribution of 15 to a Theme Issue 'Plastics, the environment and human health'.

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2009*a*,*b*). Today, phthalates and BPA are found in many mass-produced products including medical devices, food packaging, perfumes, cosmetics, children's toys, flooring materials, computers, CDs, etc. (Andrady & Neal 2009). Phthalates may constitute up to 50 per cent of the total weight of PVC plastics, and their worldwide annual production is approximately 2.7 million metric tonnes (Bauer & Herrmann 1997). For BPA, worldwide annual production is 2.5 million metric tonnes (Staples *et al.* 2002). Plasticizers work by reducing the chemical affinity between molecules when embedded between chains of plastic raw materials (or act as monomers in polycarbonate plastic), but as plasticizers are not especially stable in these products, they can leach out and thus end up in the environment. Phthalates are not classified as persistent compounds (Staples *et al.* 1997), but their occurrence in the environment has been reported widely, possibly arguing against a rapid biodegradation in some environments (Fatoki & Vernon 1990; Bauer & Herrmann 1997; Heemken *et al.* 2001; Fromme *et al.* 2002; Beauchesne *et al.* 2007). Bisphenol A is easily degraded (Howard 1989), but nevertheless regularly detected in aquatic ecosystems owing to its continuous release into the environment.

The European Union Environmental Risk Assessment (ERA) reports provide predicted environmental concentrations (PECs) for phthalates and BPA using the European Union System for the Evaluation of Substances (EUSES). The reliability of some of these PECs, however, is disputed as monitoring data from surface waters have found significantly higher environmental concentrations of phthalates. While the PEC for dibutyl phthalate (DBP) is 0.4 $\mu g\,l^{-1}$ and 0.231 $mg\,kg^{-1}$ (dry wt.) for water and sediments, respectively (EU 2004), Fatoki & Vernon (1990) report a maximum measured concentration of 33.5 $\mu g\,l^{-1}$ in UK rivers. The ERA draft report calculates a PEC for diethylhexyl phthalate (DEHP) of 2.2 $\mu g\,l^{-1}$ for water and 34 $mg\,kg^{-1}$ (dry wt.) for sediments (EU 2006). This figure is exceeded in some German surface waters where measured concentrations range between 0.33 and 97.8 $\mu g\,l^{-1}$ (Fromme *et al.* 2002) and in Japanese freshwaters and sediments where peak concentrations have been measured at 58 $\mu g\,l^{-1}$ and 210 000 $\mu g\,kg^{-1}$ (dry wt.), respectively (average concentrations in Japanese sediments range between 46 and 326 $\mu g\,kg^{-1}$ (dry wt.; Naito *et al.* 2006). For di-isodecyl phthalate (DIDP), the EUSES-based calculations state PECs of 1.8 $\mu g\,l^{-1}$ in surface waters and 32 $mg\,kg^{-1}$ (dry wt.) in sediments (EU 2003*a*) and for di-isononyl phthalate (DINP), 0.7 $\mu g\,l^{-1}$ in surface waters and 18 $mg\,kg^{-1}$ (dry wt.) in sediments (EU 2003*b*). There is a lack of data on measured environmental concentrations of these compounds to confirm or refute these calculations.

Bisphenol A is present in many environmental compartments, including the aquatic environment, and enters water systems through point discharges, such as landfill leachates and sewage treatment plant effluents. Concentrations of BPA vary widely in the aquatic environment, but they may even reach concentrations up to 21 $\mu g\,l^{-1}$ in freshwater systems (Crain *et al.* 2007). Concentrations of BPA in sediments are generally several orders of magnitude higher than those in the water column. For example, in the River Elbe in Germany, BPA was measured at 0.776 $\mu g\,l^{-1}$ in the water compared with 343 $\mu g\,kg^{-1}$ (dry wt.) in the sediment (Heemken *et al.* 2001). These findings contrast with a PEC of 0.12 $\mu g\,l^{-1}$ for water and 1.6 $\mu g\,kg^{-1}$ (dry wt.) for sediments as indicated by the ERA report for BPA (EU 2003*c*).

Heightening our concern about possible health implications of exposure to plasticizers, phthalates and BPA have been shown to bioaccumulate in organisms. Bioconcentration factors (BCFs) for plasticizers, however, appear to differ widely between species. Some of these differences relate to the type and nature of the plasticizers, but others relate to the differences in the experimental designs employed, diet, life and species' history and the tissues investigated. Bioconcentration factors reported in invertebrates are generally higher than those in vertebrates. As an example, for DEHP, BCFs reported are between 42 and 842 in different fish species, but 3600 for the amphipod *Gammarus pseudolimnaueus*, 2500 in the mussel *Mytilus edulis* and up to 5380 in the copepod *Acartia tonsa* (EU 2003*d*).

In this review paper, we analyse critically the effects on wildlife of the most prominent plasticizers, with a focus on effects in annelids (both aquatic and terrestrial), molluscs, crustaceans, insects, fish and amphibians. The paper further presents novel data from the authors on the biological effects of some plasticizers on animal species. The paper concludes with an analysis on the level of harm (and the uncertainties) associated with the discharge of plasticizers into the environment and identifies key research needs to address some of the uncertainties.

2. MATERIAL AND METHODS

In this paper, a series of experiments are reported upon that provide novel data on the effects of butyl benzyl phthalate (BBP) on a fish (*Danio rerio*) and BPA on a crustacean (*A. tonsa*) and an amphibian (*Xenopus laevis*). These experiments are described briefly, as detailed information on study designs can be obtained from the references cited.

(a) Danio rerio (*zebrafish*)

A study was conducted to investigate the effects of BBP on reproduction in adult *D. rerio*. Fish were maintained in breeding colonies of six males and six females, according to Santos *et al.* (2007). The chemical exposure tanks were set up in duplicate and dosed with 0 (solvent controls with methanol $<0.01‰$), 3, 6 and 15 μg BBP l^{-1} and run for 20 days. The total number of eggs produced and the number of viable embryos at 8 h post fertilization (hpf) were determined according to Santos *et al.* (2007). At the end of the experiment, quantification of vitellogenin (VTG) in the plasma was performed according to Fenske *et al.* (2001), with an adjustment of the primary VTG antibody concentrations to 1 : 10 000 and a primary antibody incubation time of 30 min. One gonad from all parent males was dissected and immediately used for sperm quantity and motility assessments (as measures of sperm quality) using computer-assisted sperm

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analysis (CASA, v. 7V3B; Hobson Vision Ltd, UK) on a Hobson sperm tracker (Hobson Vision Ltd), according to the protocols and settings described in Santos *et al.* (2007).

Water samples were collected from each tank on days 1, 4, 7, 14 and 20, fixed with 5% methanol (Fisher, UK), extracted using Sep-Pack C18 cartridges (Waters Corporation, USA) and BBP concentrations of the extracted samples were determined by HPLC analysis (Shimadzu, USA). Samples were analysed through separation on a C18 column (Spherisorb ODS2) using an isocratic gradient of 70 per cent methanol in water, and the absorbance was monitored at 277 nm on a UV HPLC detector (Applied Biosystems, UK). Quantification was achieved by comparison of the absorbance of the BBP peak in each sample with the absorbance of serially diluted BBP standards (ranging from 0.0001 to 1 mg ml^{-1}).

The statistical comparisons on the VTG and fecundity data were performed using one-way ANOVA followed by Tukey's multiple comparison post-test analysis (using JANDEL Sigmastat 2.0 statistical software). The sperm data were arcsine transformed before a one-way ANOVA and Tukey's multiple comparison post-test analysis using Excel (Microsoft Excel 2002) and GraphPad Prism (v. 4.02, San Diego, CA, USA).

(b) Acartia tonsa (*calanoid copepod, Crustacea*)

Full life-cycle experiments were conducted with *A. tonsa*, starting with eggs and extending for three weeks to reproductively active adults, with BPA following an OECD guideline proposal (Kusk & Wollenberger 2007) with minor modifications. Animals were exposed in mass cultures (800 ml of medium/test solution with 400 copepods). Nominal BPA test concentrations were 10, 50, 100, 300, 1000 $\mu g\, l^{-1}$. Semi-static exposures were performed at $20 \pm 0.5°C$ with a photoperiod of 12 L : 12 D cycle using a synthetic seawater medium with a salinity of 18‰ (Kusk & Wollenberger 1999). On day 14, the number of eggs produced within a 24 h period was determined in isolated groups of 5 females (5 replicates per concentration, 10 control groups) and expressed as eggs per female. Eggs produced on day 18 were transferred to a clean medium (6 replicates per concentration, each containing 30 eggs) in order to observe the hatching rate and larval mortality of the second generation, determined on day 21. The larval development ratio (LDR), describing the proportion of animals completing metamorphosis from the nauplius to the copepodite morphology, was recorded on day 5. The LDR test (Wollenberger *et al.* 2005) was conducted in parallel to the exposure of the mass culture under the same exposure conditions, with nominal BPA concentrations of 30, 60, 125, 250, 500, 1000 $\mu g\, l^{-1}$, with 4 replicates per concentration, each containing 30 copepods and 12 control groups.

(c) Xenopus laevis (*African clawed frog*)

The impacts of BPA on the thyroid system of *X. laevis* were determined as described previously for tetrabromobisphenol A by Jagnytsch *et al.* (2006). Briefly, the levels of thyroid receptor β (TRβ) mRNA were measured in the brain as biomarker for the bioavailability of thyroid hormones (THs) (Opitz *et al.* 2006). Bisphenol A was applied at concentrations ranging between 2.5 and 500 $\mu g\, l^{-1}$ (2.5, 25, 250, 500 $\mu g\, l^{-1}$) with 2 replicates per concentration and 20 tadpoles per replicate in a long-term *Xenopus* metamorphosis assay (XEMA 21 d) and at BPA concentrations of 100, 250 and 500 $\mu g\, l^{-1}$ with and without 0.1 and 1.0 nM tri-iodothyronine (T3) in short-term exposures for 1, 2 and 3 days (starting with duplicate tanks, each containing 30 tadpoles/aquarium and subsampling 10 individuals in each replicate every day). The effects of exposure to BPA during larval development on sexual differentiation were also investigated, using a flow-through system, to the point of completion of metamorphosis (see Lutz *et al.* 2008 for details of protocol). Briefly, tadpoles were exposed to BPA at concentrations of 10^{-9}, 10^{-8}, 10^{-7} and 10^{-6} M (228 ng l^{-1} to 228 $\mu g\, l^{-1}$) and to dilution water without BPA (negative control). A positive oestrogen control was also included (0.2 μg 17β-oestradiol l^{-1}). In this study, there were 4 replicates for all treatments, and 25 individual tadpoles were exposed in each tank ($n = 100$ per treatment). The endpoints investigated were survival rate, time to completion of metamorphosis, weight, sex (based on gross morphology) and histology of testes. In addition, gene expression for hepatic insulin-like growth factor-I (IGF-I) and VTG mRNA was analysed in male *X. laevis* immediately at completion of metamorphosis using semi-quantitative RT–PCR, as described previously (Kloas *et al.* 1999).

3. RESULTS AND DISCUSSION

(a) *Effects of phthalates*

Assessing the biological effects of phthalates such as DEHP in aquatic environments presents a significant challenge owing to its low water solubility and its tendency to form colloidal dispersions above 3 $\mu g\, l^{-1}$. Many experiments exposing DEHP via the water at concentrations $>3\ \mu g\, l^{-1}$ have, nevertheless, clearly established concentration-related biological effects (Forget-Leray *et al.* 2005). Despite this, risk assessments of DEHP in Japan (Naito *et al.* 2006) and the EU (EU 2003*d*) have disregarded most studies adopting concentrations $>3\ \mu g\, l^{-1}$, and the Japanese assessment has set the threshold effect level (PNEC) to 3 μg DEHP l^{-1} (Naito *et al.* 2006). This omission needs addressing given the effects reported for this phthalate in this review. Working with other highly lipophilic phthalates presents similar solubility problems as for DEHP. Di-isononyl phthalate and DIDP have a $\log K_{ow}$ of 8.8 and a water solubility of only 0.2 $\mu g\, l^{-1}$ (EU 2003*a*,*b*), which makes them even more difficult to handle in aquatic media than DEHP.

(i) *Annelids and molluscs*

Despite their ecological importance, annelids and molluscs have received relatively little attention in assessing the effects of plasticizers compared with vertebrates or other invertebrates (especially crustaceans and insects; Weltje & Schulte-Oehlmann 2007). Early investigations on annelids and molluscs

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focused on bioaccumulation and acute toxicity of phthalates, but more recently, wider biological effects have been shown, including mitotic inhibition, induction of chromosomal aberrations and effects on larval development.

Considering soil-dwelling organisms, studies on the uptake and toxicity of the phthalates dimethyl phthalate (DMP), diethyl phthalate (DEP), DBP and DEHP (at a concentration of 5 mg kg^{-1} for 25 days) in the worm *Eisenia fetida* have shown that high molecular weight phthalates (DBP and DEHP) were incorporated but with relatively low accumulation factors (0.242–0.307 and 0.073–0.244, respectively), and DMP and DEP were apparently not taken up (Xiao-yu *et al.* 2005). It is possible that the lack of any uptake of DMP and DEP may be as a consequence of their higher microbial degradation rates and a faster metabolic degradation (Albro *et al.* 1993). Neuhauser *et al.* (1986) investigated the acute toxicity of phthalates and found similar LC_{50} values (1064–3335 mg kg^{-1} for a 14 day soil exposure) for DMP in four soil-dwelling annelid species (*Allolobophora tuberculata, E. fetida, Eudrilus eugeniae, Perionyx excavatus*). Together, these studies demonstrate that phthalates enter terrestrial worms, but they have a low acute toxicity. It is worth emphasizing that soils can receive very high inputs of phthalates, and the total amount of DEHP deposited in US landfills to 1992 has been estimated at 5 million metric tonnes.

Comparisons of the acute toxicity of several phthalate esters [DMP, DEP, DBP, BBP, di-*n*-hexyl-phthalate (DHP) and DEHP] in various aquatic organisms including the annelid *Lumbriculus variegatus* have shown that the higher molecular weight phthalates DHP and DEHP were not acutely toxic for the concentration range tested (<2.3–47.8 μg DHP l^{-1}, 10.2–69.1 μg DEHP l^{-1}; Call *et al.* 2001). The toxicity of lower molecular weight phthalates was positively correlated with their K_{ow}, indicating non-polar narcosis as the underlying mode of action. The same study found *L. variegatus* was the species least sensitive to DMP; the 10 day LC_{50} (10 day) was 246, compared with 28.1 and 68.2 mg l^{-1} for the crustacean *Hyalella azteca* and the insect *Chironomus tentans*, respectively. In adult marine tubeworms *Pomatoceros lamarckii*, exposure to DMP was shown to decrease fertilization success at a threshold concentration of 1×10^{-5} M DMP (1.94 mg l^{-1}) and induce a significant increase in the number of aberrations in chromosome separations in oocytes at anaphase at concentrations $\geq1\times10^{-7}$ M (19.4 μg l^{-1}) (Dixon *et al.* 1999; Wilson *et al.* 2002). DMP has been shown to affect larval development in the polychaete worm at a threshold concentration $\geq1\times10^{-5}$ M (1.94 mg l^{-1}).

The effects of phthalates have also been studied on various biochemical pathways such as antioxidant and peroxisomal marker enzymes in marine mussels (*Mytilus galloprovincialis*, Orbea *et al.* 2002; *M. edulis*, Cajaraville & Ortiz-Zarragoitia 2006). Exposure to 500 μg DEHP l^{-1} for 21 days was shown to result in a significant increase in the catalase and acyl-CoA oxidase (AOX) activity and an inhibition of the superoxide and manganate superoxide dismutase in *M. galloprovincialis*. The peroxisomal volume density in the digestive gland of *M. edulis* was significantly enhanced at 50 μg diallylphthalate (DAP) per litre after three weeks exposure, whereas the chemical had no significant effect on AOX activity in this species (Cajaraville & Ortiz-Zarragoitia 2006). DAP was able to decrease the phospho-protein level (a mussel VTG-like protein) but did not impair ovarian follicles, oocytes and spermatogenesis in *M. edulis* when exposed to 50 μg DAP l^{-1} for three weeks (Aarab *et al.* 2006). In another 21 day study, DAP was also shown to increase micronuclei frequency, a marker of genotoxicity, and fragmented-apoptotic cells in the gills of *M. edulis* at 50 μg DAP l^{-1} (Baršienė *et al.* 2006).

Very few studies have considered the effects of phthalates on annelid or mollusc populations, but Tagatz *et al.* (1986) found that exposure to DBP altered the community structure and colonization profile of the zoomacrobenthos under laboratory and field conditions. The densities of molluscs and echinoderms were reduced by 49 and 97 per cent, while no effects were seen in annelids, arthropods, chordates and other groups (at concentrations of 10–1000 mg kg^{-1} over eight weeks).

(ii) *Crustaceans and insects*

Lower molecular weight phthalates that are relatively water soluble (Howard *et al.* 1985) have been shown to have a lower acute and chronic toxicity in *Daphnia magna*. For example, 48 h immobilization tests with *D. magna* have shown IC_{50} values of 284, 22.0 and 6.78 mg l^{-1}, for DMP, DEP and DBP, respectively (Jonsson & Baun 2003). Acute toxicity for phthalates, however, differs significantly for different species of crustaceans. *Hyalella azteca* is 10–20-fold more sensitive than *D. magna*, although the higher molecular weight phthalates DHP and DEHP did not exhibit toxic effects in *H. azteca* even at concentrations up to 22.5 μg DHP l^{-1} and 59.0 μg DEHP l^{-1} (Call *et al.* 2001). Other effects of DBP reported in crustaceans include decreased locomotor activity in *Gammarus pulex* at an exposure concentration of 500 μg l^{-1} (Thuren & Woin 1991).

In the *Drosophila melanogaster* BII cell line, DEP exhibited a weak ecdysteroid antagonistic activity (Dinan *et al.* 2001). However, the concentrations required to cause an effect were several orders of magnitude higher than that for 20-hydroxy ecdysone, thus classifying DEP as a very weak environmental anti-ecdysteroid only.

DEHP has been shown to induce sublethal effects in *Chironomus riparius*, and the exposure of larvae resulted in an increased female body volume at exposure concentrations of 0.3 μg l^{-1} (Kwak & Lee 2005). It also increased the expression of heat shock protein (HSP) and haemoglobin genes in exposed larvae of *C. tentans* at a concentration of 0.5 μg l^{-1} (Lee *et al.* 2006*a*), but the same compound was not acutely toxic to this species at concentrations up to 69.1 μg DEHP l^{-1} (Call *et al.* 2001). The significance of such effects on gene expression is not known, but in vertebrates, HSPs are involved in steroid hormone receptor stabilization and activation (Gillesby & Zacharewski 1998) and if they act similarly in

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arthropods, a change in the HSP70 titre might result in a disruption of the ecdysone axis by influencing the ecdysone receptor.

(iii) *Fish*

Given the widespread occurrence of phthalates in the aquatic environment, fish are also likely to be exposed to phthalates via the water column, food and/or via the sediments, depending on their ecological niche. In support of this, a study in wild fish in The Netherlands measured median concentrations of DEHP and DBP at 1.7 and 1.0 $\mu g\,kg^{-1}$ (wet wt.), respectively, with 95th percentile values reaching up to 141 μg DEHP kg^{-1} and 26 μg DBP kg^{-1} (wet wt.) (Peijnenburg & Struijs 2006). Laboratory studies have further shown that phthalates bioconcentrate in fish. For example, the BCF for DEHP was 120 in the carp and for BBP, 9.4 in bluegill sunfish (Staples *et al.* 1997). The presence of phthalates in wild fish and their ability to concentrate in body tissues demonstrate their potential to induce adverse effects on the health of wild populations, but there has been little bearing on this expectation.

Very few studies have comprehensively addressed the biological effects of phthalates on fish, and *in vivo* studies have reported mixed findings. BBP has been reported to be oestrogenic and induces VTG synthesis in male fathead minnow (*Pimephales promelas*) at a concentration of 100 $\mu g\,l^{-1}$ via the water (Harries *et al.* 2000), and, similarly, to induce VTG in carp (*Cyprinus carpio*) exposed to 1–20 mg DEP l^{-1} over a period of four weeks (Barse *et al.* 2007). In salmon (*Salmo salar*), exposure to 1500 mg DEHP kg^{-1} in the food for a prolonged period (four months) during early life resulted in a small incidence of intersex (Norman *et al.* 2007). These *in vivo* effects are consistent with the mode of action of BBP and DEHP established *in vitro* where they have been shown to bind to the oestrogen receptor (ER) and elicit a weakly oestrogenic response in fish cell cultures (Jobling *et al.* 1995) and in the yeast oestrogen screen (YES; containing the human ER) (Harris *et al.* 1997). In female Japanese medaka (*Oryzias latipes*), exposure to concentrations in excess of 1 μg DEHP l^{-1} caused a marked reduction in both VTG and in the percentage of females containing mature oocytes, suggesting an anti-oestrogenic mode of action (Kim *et al.* 2002). This anti-oestrogenic effect presumably arises through competition with the endogenous oestrogens for the receptor. *In vitro* receptor interaction studies have also shown that BBP and DEHP possess anti-androgenic activity (Sohoni & Sumpter 1998). Other *in vivo* effects of phthalates in fish include effects on the metabolic pathways involved in steroid biosynthesis and metabolism. In a study by Thibaut & Porte (2004), DEHP and DBP altered the activity of enzymes involved in the synthesis of endogenous steroid hormones and their metabolism in carp, effects that can lead to the alteration of the balance between endogenous oestrogens and androgens, and thus alterations in their reproductive physiology. These findings are supported by studies in mammals in which phthalates have also been shown to alter the expression and enzymatic activity of enzymes involved in the synthesis and metabolism of sex steroids in the testis (Liu *et al.* 2005). In fish, exposures to low $\mu g\,l^{-1}$ concentrations of DEP have been shown to induce alterations in a number of enzymes in the liver and muscle, including phosphatases and transaminases, showing a disruptive effect on general metabolic functions (Barse *et al.* 2007). Other studies have also reported that, potentially, the most relevant mechanism of action of phthalates is via binding to the peroxisome proliferator-activated receptors (PPARs) and modulation of the transcriptional events coordinated by this molecular pathway, which includes steroid hormone metabolism (Fan *et al.* 2004).

Exposures to phthalates have also been shown to alter behaviour in fish. Exposure to 100 μg BBP l^{-1}, via the water, caused alterations in shoaling and feeding behaviour in three-spined stickleback (*Gasterosteus aculeatus*) (Wibe *et al.* 2002, 2004), and exposure to 5 mg DEP l^{-1} caused alterations in the general behaviour of common carp (Barse *et al.* 2007). It should be noted that these studies have employed very high exposure concentrations, and these are unlikely to occur in the water column in part owing to their low solubility. Exceptions to this may be for fish living in/closely associated with the sediments in heavily contaminated environments. Exposures of fish to lower levels of phthalates have generally found no adverse effects. As an example, exposure of Japanese medaka to 21.9 and 19.3 $\mu g\,g^{-1}$ DINP and DIDP, respectively, via the diet, with a daily feeding regime of 5 per cent of the body weight, failed to identify any effects on reproduction (gonad somatic index, egg production, embryo survival and sex ratios), growth or survival in a multi-generational study (14 days post fertilization (dpf) to 140 dpf of the F1 generation) (Patyna *et al.* 2006). Similarly, exposure of medaka to concentrations up to 5 mg DEHP l^{-1} for 90 days post-hatch did not affect germ cell development (Metcalfe *et al.* 2001).

In our exposure of adult breeding zebrafish to BBP, measured concentrations in the tank water were 3, 6 and 15 $\mu g\,l^{-1}$ over the exposure period. Exposure to BBP did not affect the concentration of VTG in the plasma at any of the concentrations tested in males (VTG concentrations in the plasma were 1310 ± 255; 1500 ± 358; 948 ± 141; 1300 ± 318; 1640 ± 220 ng ml^{-1} for the water control, solvent control, 3, 6 and 15 $\mu g\,l^{-1}$ BBP, respectively; $n = 9–12$ males per treatment group). Similarly, there was no evidence of any chemical effect on the number of eggs spawned; the total number of eggs spawned per colony over the assessment period ranged between 4109 and 7578 with a maximum of 727 eggs spawned by a single colony on a single day, with a mean number of eggs spawned per female per day (including days with no eggs) of 22.2 ± 1.4; 21.3 ± 1.4; 18.4 ± 0.8; 21.6 ± 1.7; 28.5 ± 1.4 prior to exposure and 19.5 ± 1.6; 27.0 ± 1.7; 21.1 ± 1.2; 25.0 ± 2.1; 32.5 ± 1.4 during the exposure for the water control, solvent control, 3, 6 and 15 μg BBP l^{-1}, respectively. Furthermore, there were no effects of BBP on the mean percentage viability of embryos at 8 hpf (figure 1). These results are in agreement with reports in the literature, where VTG

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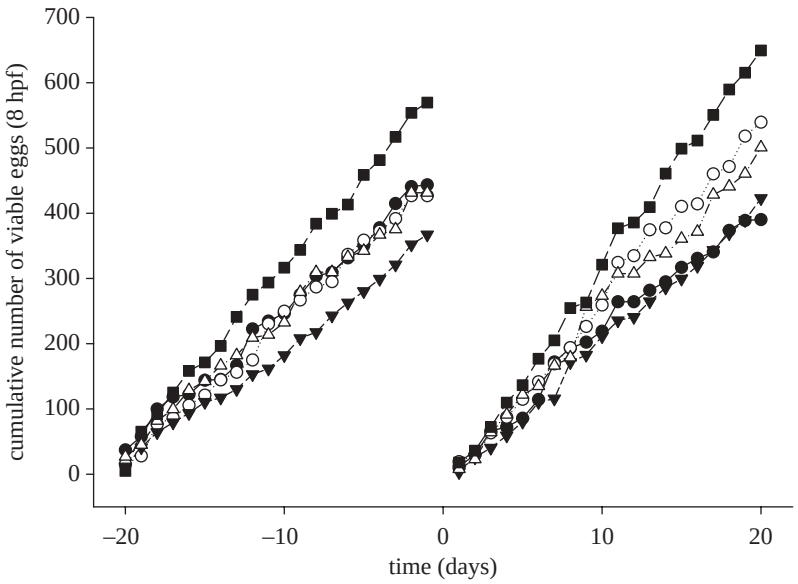

Figure 1. *Danio rerio.* Cumulative number of viable eggs spawned in colonies during a 20 day period prior to BBP exposure and a 20 day period during BBP exposure. Symbols represent average results for two colonies per treatment: filled circle, water control; open circle, solvent control; inverted filled triangle, 0.003 mg l^{-1} BBP; open triangle, 0.006 mg l^{-1} BBP; filled square, 0.015 mg l^{-1} BBP.

induction in males, or reproductive effects on females, has been induced only at exposure concentrations that far exceed the measured concentrations in our study. In contrast, however, we found consistent changes in parameters of sperm quality in BBP-exposed male zebrafish. The solvent adopted (methanol) had a negative effect on zebrafish sperm motility compared with the water control after 20 days of exposure (mean curvilinear velocity of 14 μm s^{-1} (mean) and 25 μm s^{-1}, respectively, $p < 0.05$, figure 2*a*), and BBP exposure resulted in a concentration-dependent enhancement in curvilinear velocity of the solvent-affected sperm. At the highest concentration tested (15 μg BBP l^{-1}), the average curvilinear velocity of the sperm was restored to levels similar to those found in the water control tanks (31 μm s^{-1}; figure 2*a*). Both the number of sperm tracked and the linearity of the sperm movement showed patterns similar to those described for the curvilinear velocity (figure 2*b*,*c*).

The ability of BBP to reverse the adverse effect of the methanol on sperm number and motility is a novel finding, and we hypothesize that this may have resulted from an interaction of BBP with the PPAR receptor, which leads to increased oxidative stress in the testis, as demonstrated in mammals for phthalates (reviewed in Corton & Lapinskas 2005). In the final stages of sperm development, maturation and activation are crucially dependent on the oxidative balance within the seminiferous tubes and in the sperm itself. Exposure to methanol may have reduced the free oxygen radicals required to promote sperm maturation and activation, and this effect may have been reversed by the increase in free radicals caused by the BBP exposure. The ethanol metabolite, acetaldehyde, has been shown to block the action of PPARs *in vitro* (Galli *et al.* 2001), suggesting a mechanism by which sperm motility might be reduced in fish exposed to a solvent. Exposure to BBP in the solvent carrier may counterbalance this effect, restoring the normal motility of the sperm. The increase in oxidative stress within the testis as a result of phthalate exposure has been previously observed in mammalian systems, including in rats (Kasahara *et al.* 2002), and this effect was shown to be mediated through activation of PPAR pathways.

(iv) *Amphibians*

Studies on phthalates in amphibians (reviewed in Kloas 2002) are less extensive than those in fish and some invertebrates. The effects of plasticizers in amphibians have been focused on their toxicity, their potential to induce teratogenic effects and their endocrine-disrupting activities, most notably effects on reproduction, sexual differentiation and the thyroid system. Focusing on toxicity, Larsson & Thurén (1987) found that exposure of *Rana arvalis* eggs to DEHP via sediment decreased successful hatchings with increasing concentrations and deduced a no observed effect concentration (NOEC) of 10 mg kg^{-1} fresh weight. Studies with DBP using the frog embryo teratogenesis assay *Xenopus* (FETAX) have shown developmental malformations at 500 μg l^{-1} and enhanced mortality rates at concentrations of 10 and 15 mg l^{-1} (Lee *et al.* 2005).

DBP has also been shown to affect sexual differentiation in male *Rana rugosa* tadpoles, delaying gonadal development (Ohtani *et al.* 2000) and altering the spermatogenesis process in eight-month-old *X. laevis* after exposure during larval development (at 1–10 mg DBP l^{-1} from stage 52 to completion of metamorphosis; Lee & Veeramachaneni 2005). Studies in *Xenopus* investigating the potential for phthalates to bind to the ER via competitive displacement experiments with cytosolic liver preparations have found that DEP has only a moderate affinity for the ER (Lutz & Kloas 1999). Parallel *in vitro* studies using *X. laevis* primary hepatocytes, however, have shown that

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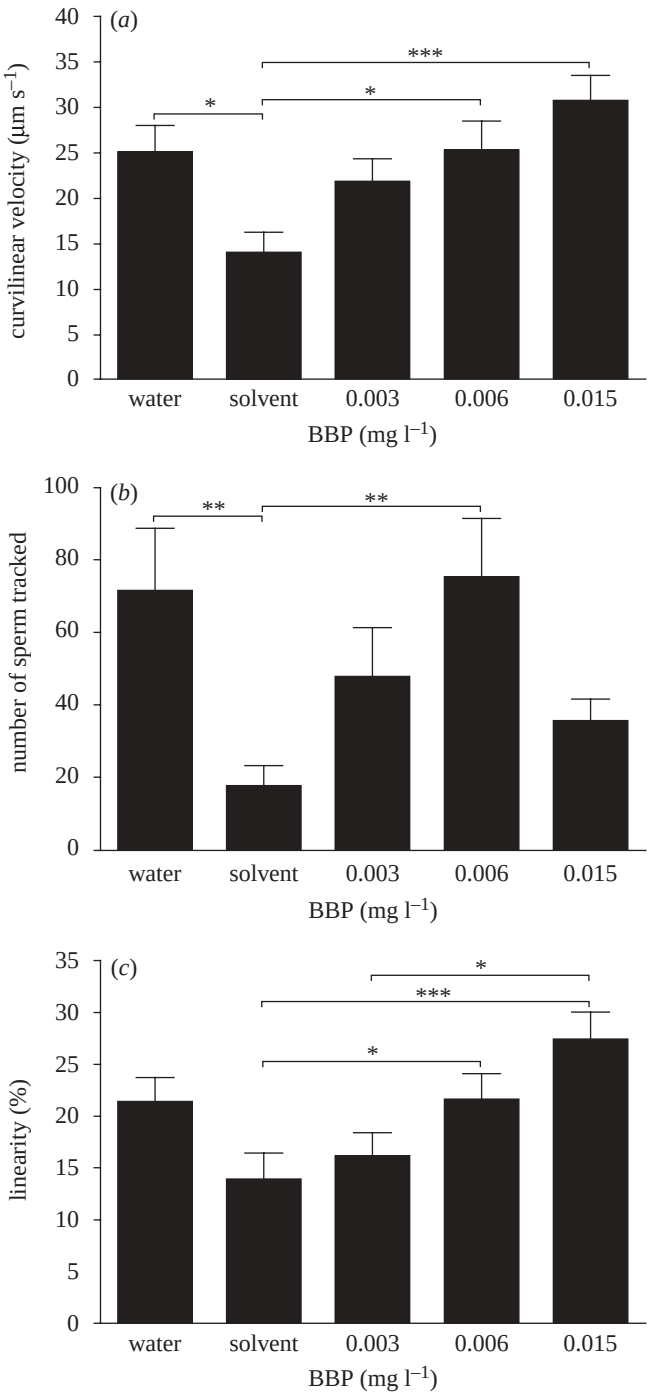

Figure 2. *Danio rerio*. Effects of exposure to BBP for 20 days on sperm quality: (*a*) the curvilinear velocity, (*b*) the number of sperm tracked and (*c*) sperm linearity. Bars represent mean ± SEM and are given for the first four intervals of tracking (i.e. sperm motility): 0–15, 16–30, 31–45 and 46–60 s. Chemical concentrations are measured in the tank water. Data were arcsine transformed before statistical analysis (one-way ANOVA and Tukey's multiple comparison post-test; $*p < 0.05$; $**p < 0.01$; $***p < 0.001$).

DBP, BBP and butylphthalyl-butylglycolate had no oestrogenic activity, deduced by measuring VTG gene expression (Kloas *et al.* 1999; Lutz *et al.* 2005) and VTG synthesis (Nomura *et al.* 2006) for a concentration range between 4×10^{-7} and 10^{-4} M (110 μg l^{-1} to 33.6 mg l^{-1}).

(b) *Effects of bisphenol A*

(i) *Annelids and molluscs*

There is only one study reporting effects of BPA in annelids (Biggers & Laufer 2004), where BPA induced metamorphosis and settlement in 2-day-old *Capitella capitata* metatrochophore larvae (a marine polychaete), following a 1 h exposure at 11.4 μg l^{-1}. In contrast, there has been considerably more research investigating the effects of BPA in molluscs, where most has been focused on gastropods and bivalves. Oehlmann *et al.* (2000) have reported that exposure of ramshorn snails (*Marisa cornuarietis*) to BPA induced a superfeminization syndrome at all nominal concentrations tested (1, 5, 25, 100 μg BPA l^{-1}). Superfemales are characterized by additional sex organs, enlarged accessory sex glands, gross malformations of the pallial oviduct, enhanced

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egg production outside the main spawning season and increased female mortality (13.3–15.7% in exposed snails compared with control mortality of 3.8%). Effects were concentration dependent (except for mortality) and statistically significant at every test concentration. Limitations on this first study included the lack of analytical confirmation of the exposure regimes, a lack of replication and the absence of a positive control. However, two follow-up exposures employing these features confirmed the original findings (Schulte-Oehlmann *et al.* 2001; Oehlmann *et al.* 2006). Schulte-Oehlmann *et al.* (2001) investigated the effects of BPA at different phases of the reproductive cycle at concentrations of 0.05, 0.1, 0.25, 0.5 and 1.0 $\mu g\,l^{-1}$ over 180 days. Before and after the main spawning season, superfemales were observed with oviduct malformations and a significantly increased egg production in all treated groups except the lowest test concentration. During the peak spawning period, no effect of BPA was seen on spawning levels, probably because spawning activity in the controls increased six-fold at this time, masking any effect seen for pre- and post-spawning periods. Calculated NOEC and EC_{10} values in this experiment were 7.9 and 13.9 ng BPA l^{-1} for egg production, respectively, based on measured concentrations in the tanks.

Oehlmann *et al.* (2006) demonstrated that the effects of BPA on reproductive output and oviduct malformations in the ramshorn snail were temperature dependent. Sexually mature snails (2 replicates of 30) were exposed to 0.25, 0.5, 1, and 5 μg BPA l^{-1} for five months outside the main spawning season at either 20 or 27°C. As was seen previously, exposure to BPA at 20°C produced significantly more eggs compared with controls. The calculated EC_{10} value was 14.8 ng l^{-1} for egg production, supporting the results from Schulte-Oehlmann *et al.* (2001). In contrast, at 27°C, none of the treatment groups produced more eggs per female compared with controls. At this temperature, however, reproductive output in controls was considerably higher, probably masking the stimulatory effects of BPA that occurred at lower temperatures. In contrast with these findings, Forbes *et al.* (2007) found no effects of BPA on fecundity, mortality and hatchability in *M. cornuarietis* when exposed to 0, 0.1, 1.0, 16, 160 or 640 μg BPA l^{-1} for 12 weeks. However, in that study, snails were exposed at temperatures of 25 ± 1°C, potentially masking the effects of BPA. Furthermore, the study also lacked a positive control, making accurate interpretations on these data difficult.

The mud snail *Potamopyrgus antipodarum* has also been shown to be sensitive to BPA when exposed via sediments with a stimulation in embryo production at an EC_{50} of 5.67 $\mu g\,kg^{-1}$ and an EC_{10} of 0.19 $\mu g\,kg^{-1}$ after four weeks of exposure (Duft *et al.* 2003). In another study on mud snails, exposure via the water to 1, 5, 25 and 100 μg BPA l^{-1} also stimulated embryo production (and additionally for the ethinyloestradiol-positive controls) with significant effects at a concentration of 5 μg BPA l^{-1} (NOEC 1 μg BPA l^{-1}; Jobling *et al.* 2004).

Bisphenol A effects have also been studied in bivalves, notably *Mytilus* species. In *M. edulis*, BPA (at 50 $\mu g\,l^{-1}$ for three weeks) has been shown to increase the phospho-protein levels, induce spawning in both sexes, damage oocytes and ovarian follicles (Aarab *et al.* 2006) and induce increased micronuclei frequency in gill cells (Baršienė *et al.* 2006). Canesi *et al.* (2005) showed that BPA at a nominal concentration of 25 nM (5.7 μg BPA l^{-1}) in the haemolymph caused a significant lysosomal membrane destabilization in haemocytes in *M. galloprovincialis*. Canesi *et al.* (2007*a*) evaluated the lysosomal membrane destabilization in extracted haemocytes of the same species and reported an NOEC of 1 μM and a corresponding EC_{50} value of 34.5 μM (7.87 mg l^{-1}). Furthermore, BPA induced significant changes in the phosphorylation of MAP kinases and STAT factors, resulting in a decrease in phosphorylation of the stress-activated p38 MAPK and CREB-like transcription factor (cAMP-responsive element binding protein). The MAP-kinase pathway is important in embryogenesis, cell growth, differentiation and apoptosis, and disturbances of STAT signalling favour tumour development by compromising the immune surveillance. Thus the findings of Canesi *et al.* (2007*a*) may have significant implications for mussel health. Canesi *et al.* (2007*b*) also reported altered patterns of gene expression and activity of enzymes involved in the redox balance following an injection of BPA into the posterior adductor muscle (at nominal doses of 3, 15 and 60 ng BPA g^{-1} dry wt.). In the digestive gland, an increased expression of the *Mytilus* ER (MeER2) was observed at the lowest test concentration 24 h after injection, whereas results for the higher test concentrations were not significant. The positive control 17β-oestradiol had the same effect on both receptors (MeER1, MeER2). Metallothionein MT20 gene expression was significantly downregulated; catalase, GSH transferase and GSSG reductase exhibited a changed activity pattern and total glutathione content was enhanced at all test concentrations. Unlike for the phthalate DAP, a study by Cajaraville & Ortiz-Zarragoitia (2006) found no effects of BPA on AOX activity or peroxisomal volume density.

(ii) *Crustaceans and insects*

Bisphenol A has been shown to be acutely toxic in the high $\mu g\,l^{-1}$ range in crustaceans and aquatic insects. In *D. magna*, nominal IC/LC_{50} values vary widely, at between 240 and 12 800 $\mu g\,l^{-1}$ (Chen *et al.* 2002; Hirano *et al.* 2004; Park & Choi 2007). The lowest acute value, an IC_{50} for *D. magna* of 240 $\mu g\,l^{-1}$, for 24 h (Park & Choi 2007), is remarkably low compared with other studies and also compared with studies for effects on reproduction in *D. magna* (Brennan *et al.* 2006). Differences in chemical sensitivity have been shown to occur between strains of *D. magna* (Baird *et al.* 1991); therefore, Park & Choi (2007) may have been working with an especially sensitive strain. Applying the Comet assay to this sensitive strain, DNA breakage occurred at 2.1 μg BPA l^{-1} (Park & Choi 2007).

Generally in *D. magna*, chronic BPA effects are observed at concentrations slightly below those that

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are acutely toxic. Bisphenol A inhibits reproduction and development rates with an NOEC of 1000 $\mu g\,l^{-1}$ (Brennan *et al.* 2006) and affected intermolt duration at 9000 $\mu g\,l^{-1}$ (Mu *et al.* 2005). In the marine copepods *A. tonsa* and *Tigriopus japonicus*, BPA has been shown to inhibit development at 100 and 0.1 $\mu g\,l^{-1}$, respectively (Andersen *et al.* 2001; Marcial *et al.* 2003). A significant stimulation of larval development in *A. tonsa* was, however, observed at 12.5–50 $\mu g\,l^{-1}$ (Mariager 2001), which was also seen in our own experiments at 30 and 60 $\mu g\,l^{-1}$, reported in this paper. In 10-day-old *A. tonsa*, there was an initial increased egg production at 20 $\mu g\,l^{-1}$ BPA (Andersen *et al.* 1999), possibly indicating an accelerated rate of maturation owing to a higher development rate of the early life stages at this exposure concentration. In our experiments with fully mature *A. tonsa* (14 days old), egg production was reduced at a lowest observed effect concentration (LOEC) of 300 μg BPA l^{-1}. Furthermore, in subsequent female offspring derived from exposed parents (to all test concentrations of BPA down to and including 50 $\mu g\,l^{-1}$), the hatching rate was significantly reduced. Mortality in second-generation larvae was also significantly higher compared with controls. These results indicate a maternal transfer of BPA into the developing eggs and/or a disrupting effect during oogenesis. In conclusion, these datasets show that concentrations of BPA between 10 and 60 $\mu g\,l^{-1}$ in the water stimulated development and growth of *A. tonsa*, while concentrations of 100 $\mu g\,l^{-1}$ and higher inhibited development, growth and egg production.

In studies on *C. riparius* exposed to BPA via spiked sediments, emergence of second-generation individuals was delayed (at a water phase concentration of 78 ng BPA l^{-1} and higher, Watts *et al.* 2001). This observation may have resulted from effects of BPA on the levels of HSP, which has been observed in *Gammarus fossarum* exposed to BPA (Schirling *et al.* 2006). In this study, a 34 day exposure to 50 and 500 μg BPA l^{-1} accelerated maturation of oocytes, reduced the number of offspring produced and resulted in reduced levels of HSP. However, in *C. tentans*, the expression of HSP70 genes has been shown to be stimulated at concentrations down to 8 μg BPA l^{-1} (the lowest concentration applied; Lee *et al.* 2006*a*). In the *D. melanogaster* BII cell line, BPA was shown to act as an ecdysteroid antagonist, but only at a high exposure concentration of 22.8 $mg\,l^{-1}$ (Dinan *et al.* 2001). Studies combining BPA with other chemicals have shown interactive effects, affecting the number of embryo abnormalities and offspring emergence (Mu *et al.* 2005; Wang *et al.* 2005). In summary, for studies in aquatic arthropods, BPA appears to interfere with their hormone systems—possibly by affecting the synthesis of HSPs and/or by interfering with the ecdysone receptor itself—and causes adverse effects at concentrations in the low $\mu g\,l^{-1}$ range.

(iii) *Fish*

Studies *in vitro* using the YES have clearly established that BPA has the ability to act as an oestrogenic substance in vertebrates by directly binding and activating the ER (Harris *et al.* 1997). In parallel, studies using a human breast cancer cell line (MCF7) have demonstrated that BPA is able to elicit an oestrogenic response similar to that induced by natural oestrogens (Olea *et al.* 1996). In these studies, the concentrations required to cause an effect, however, were several orders of magnitude higher than that for oestradiol, thus classifying BPA as a weak environmental oestrogen in vertebrates.

The presence of BPA in rivers and streams at relatively high concentrations (up to 21 $\mu g\,l^{-1}$, Crain *et al.* 2007) has led to significant research efforts investigating its biological effects in fish. Consistent with these findings, BPA has been shown to have feminizing effects *in vivo* and to induce VTG and/or zona radiata proteins (ZRPs) synthesis in a diverse range of fish species (carp, 100 $\mu g\,l^{-1}$, Mandich *et al.* 2007; fathead minnow, 160 $\mu g\,l^{-1}$, Brian *et al.* 2005; Sohoni *et al.* 2001; cod, 50 $\mu g\,l^{-1}$, Larsen *et al.* 2006; medaka, 1000 $\mu g\,l^{-1}$, Ishibashi *et al.* 2005; rainbow trout, 500 $\mu g\,l^{-1}$, Lindholst *et al.* 2001). These concentrations, however, have not been reported in the aquatic environment. *In vivo* studies have shown that BPA affects numerous other biological processes and in many cases at concentrations that have been found in the aquatic environment. These effects include disruptions in androgen and oestrogen synthesis and metabolism. Studies in carp have reported that exposure to low concentrations of BPA (1–10 $\mu g\,l^{-1}$) decreases the ratio of oestrogen to androgen in the plasma, whereas exposure to high concentrations (1000 $\mu g\,l^{-1}$) increases it (Mandich *et al.* 2007). Alterations in concentrations of circulating steroids have also been reported in juvenile turbot (*Psetta maxima*), exposed to 59 μg BPA l^{-1} (Labadie & Budzinski 2006). This effect is likely to result from alterations in the levels of steroidogenic enzymes, or their enzymatic activity catalysing the synthesis and/or metabolism of these hormones and/or displacement of oestradiol from sex steroid-binding globulins (reviewed in Crain *et al.* 2007). The biological consequences of induced changes in the ratio between oestrogens and androgens are wide ranging and may include masculinization or feminization of organisms, and/or alterations in other processes regulated by these hormones (including growth, bone morphogenesis, insulin signalling, neural development, cell division and apoptosis).

Studies in fish, as in some invertebrates, have also provided evidence for differential species sensitivities for exposure to BPA. For example, in exposures of Atlantic cod (*Gadus morhua*) and turbot (*Scophthalmus maximus*) to 59 μg BPA l^{-1} via the water, VTG and ZRP were more highly induced in the cod than in turbot (Larsen *et al.* 2006). A possible reason for the differences in species sensitivity is in the rate of metabolic transformation of BPA. Supporting this argument, BPA has been shown to be metabolized and eliminated much more rapidly in the zebrafish (*D. rerio*) than in the rainbow trout (*Oncorhynchus mykiss*) (Lindholst *et al.* 2003).

BPA has also been shown to induce alterations in gonadal development and gamete quality in fish, including at concentrations found in the environment. These effects include alterations in the progression of

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germ cell development in fathead minnow (1 μg BPA l^{-1}, Sohoni *et al.* 2001), alterations in the gonadal structure in male carp (from 1 μg BPA l^{-1}) and an enhancement of atresia in oocytes in female carp (1 μg BPA l^{-1}, Mandich *et al.* 2007). In addition, in brown trout (*Salmo trutta f. fario*), exposures to BPA has been shown to cause reduced sperm quality (1.75–2.40 μg BPA l^{-1}), delayed ovulation in females (1.75 μg BPA l^{-1}) and inhibition of ovulation in females (5 μg BPA l^{-1}, Lahnsteiner *et al.* 2005). The effects observed for brown trout may have population-level implications as they would lead to a delay in breeding, with the risk that offspring would be produced at less favourable seasonal periods. In carp, exposure to very high levels of BPA (1 mg l^{-1}) has also been shown to induce intersex (Mandich *et al.* 2007). Another reported physiological effect in turbot exposed to BPA (at 50 μg BPA l^{-1} for three weeks) is an increased micronuclei frequency in erythrocytes, demonstrating the capability of this compound to induce DNA damage in fish (Bolognesi *et al.* 2006).

Gene expression studies have been employed with good success for identifying the molecular mechanisms leading to the biological effects of BPA in fish. Studies in *Kryptolebias marmoratus* (formerly *Rivulus m.*) have shown that exposure to BPA induces changes in the expression of genes associated with oestrogen signalling, including cytochrome P450 aromatase A and B and ERα but not ERβ or androgen receptor (Lee *et al.* 2006*b*; Seo *et al.* 2006). Similarly, BPA has been shown to induce the expression of ERα in the anal fin of Japanese medaka (*O. latipes*) (Hayashi *et al.* 2007). Studies on carp (*C. carpio*) exposed to BPA employing gene arrays have reported alterations in the expression of many of the same genes induced by exposure to natural and synthetic oestrogens (oestradiol and ethinyloestradiol), but also some unique genes too (Moens *et al.* 2006, 2007). Studies on *K. marmoratus* following exposure to BPA also demonstrated differential expression of large sets of genes and, in particular, genes belonging to the Gene Ontologies representing catalytic activity and binding (Lee *et al.* 2007). These studies however, have generally employed high BPA exposure concentrations. BPA has also been shown to modulate the somatostatin system in the brain of teleosts, resulting in deregulation of neuro-hormonal functions linked to the reproductive system (Alo *et al.* 2005). Taken together, these molecular mechanistic studies highlight that BPA appears to act predominantly, but not exclusively, through oestrogen signalling pathways in exerting its biological effects on fish.

The biological effects induced in laboratory exposures to BPA are consistent with some phenotypes observed in wild fish living in waters receiving high levels of effluent discharges and containing relatively high concentrations of BPA. Although BPA does not appear to bioconcentrate in fish and can degrade rapidly in the environment (Staples *et al.* 1998), it can be metabolized into biologically more active compounds (Ishibashi *et al.* 2005). Overall, the available datasets support the contention that BPA may contribute to adverse reproductive health effects in wild fish populations inhabiting contaminated sites.

(iv) *Amphibians*

BPA has been shown to induce a wide range of adverse effects in amphibians. Teratogenic effects have been induced in *X. laevis* embryos exposed from stage 6 onwards (Oka *et al.* 2003), and they include crooked vertebrae, abnormal development of head and abdomen to stage 40 (at an exposure of 2×10^{-5} M (4.6 mg l^{-1})), as well as apoptosis in the central nervous system. When exposures are started later than stage 10, however, no adverse impacts of BPA on survival or development occur (Iwamuro *et al.* 2003; Pickford *et al.* 2003; Levy *et al.* 2004). Exposure of *Rana temporaria* to BPA in combination with another stressor, ultraviolet-B (UVB) radiation, has been shown to induce an enhanced effect on tadpole mortality (Koponen & Kukkonen 2002). The mechanism for this is unknown, but it is not due to an effect of UVB radiation on the accumulation or depuration kinetics of BPA (Koponen *et al.* 2007).

BPA has been reported to affect sexual development, causing sex reversal at 10^{-7} M (22.8 μg l^{-1}) in semi-static exposures (Kloas *et al.* 1999; Levy *et al.* 2004), but under flow-through conditions (at an exposure to BPA up to 500 μg l^{-1}), no such feminizing effects were found (Pickford *et al.* 2003). In the new data presented here, no effects of exposure to BPA (using a flow-through system and spanning concentrations from 228 ng l^{-1} to 228 μg l^{-1}) were seen on gross morphological sex; in the controls and BPA treatments there were between 45 and 50 per cent males and 50 and 55 per cent females, and in the oestradiol (E2)-positive control (0.2 μg l^{-1}) there were 19 per cent males, 72 per cent females and 9 per cent mixed sex. However, detailed histological analysis of the testes (n varies between 19 in the E2 treatment and 47 in 10^{-6} M BPA treatment)—examining the complete gonad by serial sections—identified effects for the BPA treatments. At 10^{-9}, 10^{-8}, 10^{-7} and 10^{-6} M (228 ng l^{-1} and 2.28, 22.8 and 228 μg l^{-1}), BPA testicular lacunae occurred at frequencies of 15, 20, 10 and 25 per cent and oogonia with frequencies of 40, 60, 55 and 60 per cent, demonstrating oestrogenic effects even at 10^{-9} M BPA (228 ng l^{-1}). In addition, BPA also affected growth and body weight; these were significantly increased in males exposed to concentrations of 10^{-8} M BPA (2.28 μg l^{-1}) and above ($p < 0.05$; Mann–Whitney U-test). This effect on growth was supported by the induction of hepatic IGF-I expression at 10^{-8} M BPA (2.28 μg l^{-1}) and above. In these treatments, there was a significant induction of VTG gene expression in liver at 10^{-6} M BPA (228 μg l^{-1}).

In competitive displacement experiments with cytosolic liver extracts of *X. laevis*, BPA was shown to bind only moderately to amphibian ER (Lutz & Kloas 1999). Similarly in VTG- and ER-mRNA induction studies with primary hepatocytes from male *X. laevis*, BPA was stimulatory at 10^{-7} M (22.8 μg l^{-1}) compared with 10^{-9} M (272 ng l^{-1}) for E2 (Kloas *et al.* 1999; Lutz *et al.* 2005). In another, longer term *in vitro* study, induction of VTG occurred only at a BPA concentration of 1.1×10^{-5} M (2.5 mg l^{-1}) (compared with 4.1×10^{-11} M (11.1 ng l^{-1}) in the E2 positive control; Nomura *et al.* 2006). Both

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experimental approaches indicate that BPA is moderately to weakly oestrogenic for VTG and ER induction. BPA has also been shown to significantly reduce basal water absorption of male pelvic patches in the Japanese tree frog, *Hyla japonica*, a known oestrogenic response (Kohno *et al.* 2004).

BPA also affects the thyroid system in amphibians. Exposure to 10^{-5} M BPA (2.28 mg l^{-1}) has been shown to inhibit spontaneous and TH-induced metamorphic changes in *X. laevis* and suppress TRβ gene expression both *in vivo* and *in vitro* (Iwamuro *et al.* 2003). More recently, *in vitro* tail cultures of *X. laevis* have shown to respond to BPA, reducing T3-induced gene expression of TRα and TRβ at exposures as low as 10^{-7} M (22.8 μg l^{-1}). In contrast, the T3-induced suppression of RXRγ was only moderately affected by exposure to BPA alone (Iwamuro *et al.* 2006). Similar results for effects of BPA on T3-induced metamorphic changes were found by Goto *et al.* (2006) studying *Silurana tropicalis* and *R. rugosa*. In the skin of *Rana tagoi*, expression of the antimicrobial peptides preprotemporins (normally upregulated by TH) has been shown to be reduced markedly following BPA treatment at 10^{-8} M (2.28 μg l^{-1}) (Ohnuma *et al.* 2006), and using transgenic *X. laevis* embryos bearing a TH/bZIP-green fluorescent protein construct, T3-induced fluorescence was shown to be inhibited by BPA at a concentration of 10^{-5} M (2.28 mg l^{-1}; Fini *et al.* 2007).

4. CONCLUSION

The datasets reported upon here show that phthalates and BPA can affect both development and reproduction in a wide range of wildlife species. Reproductive and developmental disturbances include alterations in the number of offspring produced and/or reduced hatching success, disruption of larval development and, in insects, delayed emergence. Interphyla differences in responses to plasticizers occur, and as an example, in amphibians, phthalates and BPA disrupt thyroid function impacting on larval development, but no such effects have been reported thus far for fish, even though TH is deposited into fish oocytes and subsequently promotes and regulates larval metamorphosis. Similarly in molluscs, plasticizers have not been found to affect embryo development. There are clear differences in species sensitivity to the effects of plasticizers too. Molluscs, crustaceans and amphibians appear more responsive, with most effects induced in exposures via the water in the low ng l^{-1} to μg l^{-1} concentration range. In contrast, effects in fish are generally induced in the higher μg l^{-1} and mg l^{-1} range, with the exception of disruption of spermatogenesis by BPA, that occurs in the low μg l^{-1} range. These taxon-specific effects hamper meaningful comparisons of the relative sensitivities of different phyla to plasticizers.

Given that the biological effect concentrations for plasticizers seen in the laboratory coincide with environmental concentrations, some wildlife populations are probably impacted. To date, however, studies investigating population effects have not been reported, and this constitutes a significant knowledge gap. Furthermore, for invertebrate phyla most highly sensitive to the biological effects of specific plasticizers, long-term exposures should be considered as a research priority. However, only a very limited number of invertebrate phyla have been tested for the effects of phthalates and BPA, and representatives from other, presently neglected, taxa should be considered in the future, including terrestrial species. Mixture effects of plasticizers, as of many other environmental pollutants, are also a much neglected area of study, and given that there are likely to be additive effects, this warrants investigations to assess more accurately their impacts in the environment (see discussion in Koch & Calafat 2009; Teuten *et al.* 2009; Thompson *et al.* 2009*b*). As a final note, increasingly, investigations into the mechanisms of action of plasticizers are finding that some (e.g. phthalates) can have multiple interaction sites in the body, affecting a wide range of biological processes (see discussion in Meeker *et al.* 2009; Talsness *et al.* 2009). Thus, a more thorough understanding of the modes of action of these chemicals will help in developing a more comprehensive understanding of their potential for harm and in the identification of the species most vulnerable to their biological effects.

This work was financially supported by the European Union (COMPRENDO project, contract EVK1-CT-2002-00129). E.M.S. and G.C.P. were funded by the Natural Environment Research Council (NERC), grant ref NE/C507661/1, under the Post-genomics and Proteomics research programme and under the Environmental Genomics Thematic (grant ref: NER/T/S/2002/00182), and by the Biotechnology and Biological Sciences Research Council (BBSRC; grant ref: 9/S15001), UK to C.R.T.

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Phil. Trans. R. Soc. B 2009 **364**, 1999-2012
doi: 10.1098/rstb.2008.0207

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Phil. Trans. R. Soc. B (2009) **364**, 1999–2012
doi:10.1098/rstb.2008.0207

Review

Monitoring the abundance of plastic debris in the marine environment

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Plastic debris has significant environmental and economic impacts in marine systems. Monitoring is crucial to assess the efficacy of measures implemented to reduce the abundance of plastic debris, but it is complicated by large spatial and temporal heterogeneity in the amounts of plastic debris and by our limited understanding of the pathways followed by plastic debris and its long-term fate. To date, most monitoring has focused on beach surveys of stranded plastics and other litter. Infrequent surveys of the standing stock of litter on beaches provide crude estimates of debris types and abundance, but are biased by differential removal of litter items by beachcombing, cleanups and beach dynamics. Monitoring the accumulation of stranded debris provides an index of debris trends in adjacent waters, but is costly to undertake. At-sea sampling requires large sample sizes for statistical power to detect changes in abundance, given the high spatial and temporal heterogeneity. Another approach is to monitor the impacts of plastics. Seabirds and other marine organisms that accumulate plastics in their stomachs offer a cost-effective way to monitor the abundance and composition of small plastic litter. Changes in entanglement rates are harder to interpret, as they are sensitive to changes in population sizes of affected species. Monitoring waste disposal on ships and plastic debris levels in rivers and storm-water runoff is useful because it identifies the main sources of plastic debris entering the sea and can direct mitigation efforts. Different monitoring approaches are required to answer different questions, but attempts should be made to standardize approaches internationally.

Keywords: marine debris; mitigation; monitoring; plastic; seabirds; virgin pellets

1. INTRODUCTION

Two of the key characteristics that make plastics so useful—their light weight and durability—also make inappropriately handled waste plastics a significant environmental threat. Plastics are readily transported long distances from source areas and accumulate in sinks, mainly in the oceans, where they have a variety of significant environmental and economic impacts (Coe & Rogers 1997; Thompson *et al.* 2009*a*,*b*; UNEP 2005). Discarded plastics also affect terrestrial and fresh-water systems, including ingestion by and entanglement of animals, blocked drainage systems and aesthetic impacts. However, the literature on plastic pollution largely focuses on marine systems.

Most plastics break down slowly through a combination of photodegradation, oxidation and mechanical abrasion (Andrady 2003). Thick plastic items persist for decades, even when subject to direct sunlight, and survive even longer when shielded from UV radiation under water or in sediments. Except for expanded polystyrene, plastics take much longer to degrade in water than they do on land, mainly owing to the reduced UV exposure and lower temperatures found in aquatic habitats (Gregory & Andrady 2003). There has been a rapid increase in the amount of plastic litter in the marine environment (Ryan & Moloney 1993), linked to increases in the use of plastics. Even the development of the so-called 'biodegradable' plastics is not a long-term solution to the plastic litter problem, because many such materials contain only a proportion of biodegradable material such as starch, leaving behind microscopic plastic fragments (Klemchuk 1990).

Given the impacts of plastic litter (Gregory 2009), considerable effort has been made to remove waste plastic and other persistent debris from the environment. This removal may occur before it enters the sea, through litter collection and screening waste water systems (e.g. Marais & Armitage 2004) or, thereafter, through collections of litter from beaches (e.g. Ryan & Swanepoel 1996; Ocean Conservancy 2007), the seabed (e.g. Donohue *et al.* 2001) or at sea (Pichel *et al.* 2007). However, the most efficient and cost-effective solution is to reduce the release of plastics into the environment in the first place. Measures taken to achieve this goal include education, both of the

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One contribution of 15 to a Theme Issue 'Plastics, the environment and human health'.

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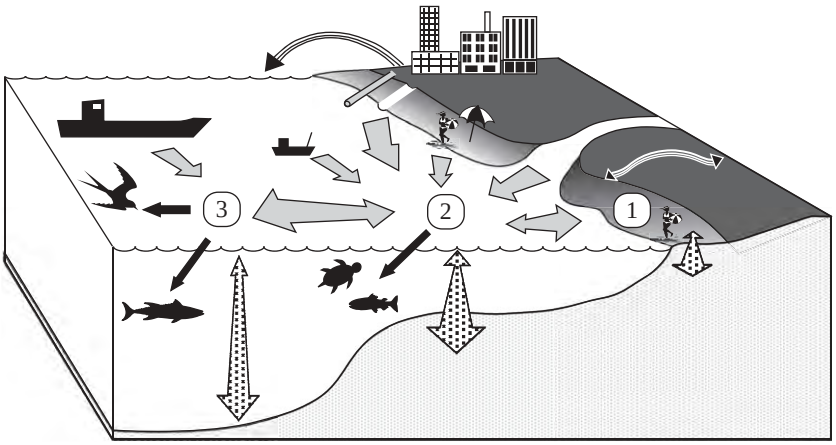

Figure 1. Schematic diagram showing the main sources and movement pathways for plastics in the marine environment, with sinks occurring (1) on beaches, (2) in coastal waters and their sediments and (3) in the open ocean. Curved arrows depict wind-blown litter, grey arrows water-borne litter, stippled arrows vertical movement through the water column (including burial in sediments) and black arrows ingestion by marine organisms.

general public and specific user groups (e.g. the plastics industry's Operation Clean Sweep to prevent loss of virgin pellets), and legislation (e.g. Annex V of MARPOL banning the dumping of plastics at sea). Assessing the efficacy of these measures requires monitoring both the amounts of plastic in the environment and the rates at which plastic litter enters the environment (Sheavly 2007).

Monitoring is a series of measures made to detect change in the state of a system (Goldsmith 1991). It is goal dependent, so the protocols used need to be tailored to the questions being asked. The main questions regarding plastic litter in the environment are:

(i) What is the abundance, distribution and composition of plastic litter, and are these attributes changing over time?
(ii) What are the main sources of plastic litter, and are they changing over time?
(iii) What are the impacts of plastic litter (environmental and economic) and are they changing over time?

For all three questions, targets may be linked to specific mitigation measures and may operate at a range of spatial and temporal scales. For example, at a local scale, monitoring may assess whether the implementation of a screening system in an urban catchment reduces litter loads to target levels. At a larger scale, we might test whether initiatives to reduce the loss of virgin pellets have reduced the abundance of pellets at sea. Similarly, monitoring may track a broad suite of litter types or specific items that have particular impacts (e.g. high risk of entangling animals) or are indicators of specific sources of plastic litter.

This paper summarizes monitoring protocols used to measure changes in plastic debris with a focus on the marine environment because accumulation and impacts of plastic litter appear to be most serious in marine systems. We highlight the strengths and weaknesses of the various approaches and provide a set of best-practice guidelines for monitoring the abundance and impacts of plastics. Our paper links with that of Barnes *et al.* (2009) summarizing trends in plastics in the environment.

2. PLASTICS IN MARINE ECOSYSTEMS

Plastics dominate marine debris (Coe & Rogers 1997; UNEP 2005). The proportion of plastic articles among litter increases with distance from source areas because they transport more easily than do more dense materials such as glass or metal and because they last longer than other low-density materials such as paper. Most plastics are less dense than water, but some are more dense (e.g. polyamide, polyterephthalate, polyvinyl chloride). Floating plastic debris has become a global problem because it is carried across ocean basins, contaminating even the most remote islands (Barnes 2002). Plastics in the marine environment derive from two main sources: rubbish dumped from ships at sea and land-based sources such as runoff from rivers, waste water systems, wind-blown litter and recreational litter left on beaches (Coe & Rogers 1997). To monitor plastic litter, we need to understand the dynamic linkages between litter sources and sinks (figure 1). As one moves offshore, there usually is an increase in the proportion of ship-based litter and a decrease in total litter loads, although aggregations may occur in mid-ocean gyres (Pichel *et al.* 2007). In addition, the suite of organisms exposed to plastic litter may differ between coastal and oceanic waters. Unfortunately, the rates at which plastic cycles through various pathways are largely unknown. The combination of multiple diffuse and point-source inputs and the non-random transportation of debris by winds and currents results in great temporal and spatial variability in litter loads. Such variability tends to mask long-term trends, requiring a non-confounded sampling design with sufficiently large replication in time and space. Superimposed on the dynamic system of plastic flux is the gradual fragmentation of large plastic articles over time. Plastic litter can be broadly divided into macrodebris (>20 mm diameter), meso-debris (2–20 mm) and micro-debris (<2 mm), although different authorities recommend subtly different size limits (e.g. Cheshire *et al.* 2009).

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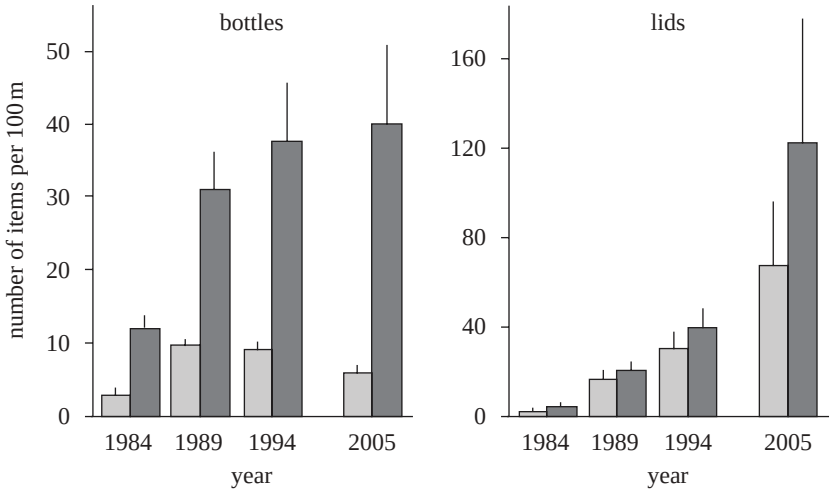

Figure 2. Scoring litter collected from a 50 m stretch of beach (top) and trends in the abundance of plastic bottles and lids (mean and s.e.) at South African beaches sampled in 1984, 1989, 1994 and 2005. Light grey bars, 36 beaches with regular, municipal cleaning programmes; dark grey bars, 14 beaches with no formal cleaning programmes (P. G. Ryan & C. L. Moloney, unpublished data).

There is little information on the rates at which different plastics degrade and fragment under different conditions (e.g. Pritchard 1997; Andrady 2003), nor is it clear what is the fate of all the plastic fragments (Thompson *et al.* 2004). We thus are challenged with trying to monitor a highly dynamic system about which our understanding is incomplete, and where the items being monitored appear and disappear in response to various societal, technological, environmental and political pressures.

Rees & Pond (1995) recognized three approaches to marine litter monitoring: (i) beach surveys, (ii) at-sea surveys and (iii) estimates of the amounts entering the sea. We summarize each of these approaches, starting with beach surveys, because they are often regarded as the simplest and most cost-effective way to monitor large-scale trends in marine litter (Dixon & Dixon 1981; Ribic *et al.* 1992; Rees & Pond 1995). We discuss the strengths and weaknesses of each approach and add a fourth one: evaluating trends in interactions between wildlife and plastic litter.

3. BEACH SURVEYS

Much of what we know about the abundance, distribution and origin of plastic debris in the marine environment comes from surveys of litter stranded on beaches (Coe & Rogers 1997). Initial studies were baseline surveys that summarized the abundance, distribution and composition of litter in various regions. We cannot provide a comprehensive review of the more than 100 published surveys of beach macro-debris (see reviews by Pruter 1987; Derraik 2002), but there are some consistent patterns: plastics dominate beach litter in terms of numbers of items (Derraik 2002), and litter loads are greater close to urban areas (e.g. Garrity & Levings 1993; Willoughby *et al.* 1997), increasing with numbers of visitors to beaches (e.g. Frost & Cullen 1997). However, many authors note the difficulty in comparing data among studies. This difficulty is largely owing to differences in sampling protocols and the type of data recorded. Some studies record the numbers of items, some the mass of litter and some do both. Litter is categorized by the type of material, function or both. Most studies record only fairly large items, although the lower limit varies from 10 to 100 mm and is often not reported. Even more fundamentally, most studies record all litter between the sea and the highest strandline on the upper shore, whereas some measure litter within fixed areas, and others only sample specific strandlines (e.g. Velander & Mocogni 1999). Only a few studies have sampled buried litter, even though it may account for a substantial

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Table 1. Comparison of survey protocols for monitoring the accumulation of beached litter in the USA (US Marine Debris Monitoring Program (USMDMP), Sheavly 2007) and Europe (Beach Litter Monitoring Programme, OSPAR Commission 2007*a*).

	USMDMP	OSPAR
type of beach	sand/gravel	sand/gravel
beach slope	15–45° (not steep)	—
beach length (m)	>500	>1000
length of beach surveyed (m)	500	100 (all items) 1000 (items >0.5 m across)
sample frequency (days)	28 ± 3	90 (approx.)[a]
type of litter recorded	31 indicator items	all debris (111 categories)
other criteria	no regular cleaning no impact on threatened species	distant from sources (rivers) visually/frequently littered

[a]Litter is not removed at some OSPAR sites and some sites are also cleaned by local municipalities (Barbara Wenneker *in litt.*).

proportion of beach litter loads (Kusui & Noda 2003). In addition, the width of beach sampled varies greatly, from a few metres (e.g. Madzena & Lasiak 1997) to entire beaches >20 km long (e.g. Edyvane *et al.* 2004). Finally, most studies report standing stocks of litter, whereas others assess the rate of accumulation following removal of existing debris. This is a critical distinction that requires detailed discussion.

(a) *Standing-stock surveys*

Standing-stock surveys can show gross changes in the abundance and distribution of plastic litter (e.g. Ryan & Moloney 1990; Willoughby *et al.* 1997; figure 2), but there are significant problems with the interpretation of results. The amount of litter on a beach is determined by several factors in addition to the abundance of litter in adjacent coastal waters. These include local currents and circulation patterns, beach structure (slope, particle size, etc.), recent weather conditions and associated beach dynamics (burial or exposure of litter, especially on sandy beaches), local land-based sources (e.g. beach recreation, proximity to poorly managed landfill sites) and, at least for macro-debris, any formal or informal cleanup efforts (OSPAR Commission 2007*a*; Cheshire *et al.* 2009). Standing stocks of beach litter reflect the long-term balance between inputs (both local, land-based sources and strandings) and removal (through export, burial, degradation and cleanups). The few studies of beach-litter dynamics indicate that beach structure influences the rate of litter turnover, with fairly rapid turnover rates (3–12 months), although the fate of most items is unknown (Garrity & Levings 1993; Bowman *et al.* 1998). However, persistent accumulations probably occur at some beaches. Repeated measures of standing stocks at such beaches may reflect the gradual accumulation of long-lasting debris rather than provide an index of changes in the abundance of debris at sea.

Some of the factors affecting litter inputs and removal are fairly constant and thus have little influence on monitoring programmes, but episodic events can mask long-term trends (e.g. storms carry large loads of land-based litter to sea and re-suspend buried litter). Of greater concern is that human activities typically change over time. Patterns of beach use tend to increase with growing human populations, coastal development and improved access. For example, many beaches categorized as 'rural' with little human influence in an initial survey of 50 South African beaches in 1984 (Ryan & Moloney 1990) have become resort beaches over the following two decades. There has been a concomitant increase in formal beach-cleaning efforts over this period (Ryan & Swanepoel 1996). As a result, the numbers of plastic bottles on beaches have stabilized at regularly cleaned beaches, but have continued to increase at seldom-cleaned beaches (figure 2). In comparison, small items such as lids that are often overlooked by cleaning teams have continued to increase at all beaches (figure 2). Beach cleanups increasingly are used as an educational tool (Storrier & McGlashan 2006; Ocean Conservancy 2007), altering litter loads on beaches, and thus need to be considered when monitoring changes in beach litter. Beachcombing also selectively removes items from all but the most remote beaches. For example, the abundance of fishing floats on uninhabited Inaccessible Island, central South Atlantic, is two orders of magnitude greater than similar beaches in South Africa (Ryan & Watkins 1988). Thus although standing-stock surveys can track changes in the composition of beach litter (Rees & Pond 1995), they are not sufficiently sensitive to monitor changes in macro-litter abundance (Escardó-Boomsma *et al.* 1995).

(b) *Accumulation and loading rates*

Many of the problems associated with monitoring trends in litter with standing-stock surveys are avoided by recording the rate at which litter accumulates on beaches. This requires an initial cleanup to remove all existing debris, followed by regular surveys that record and remove all newly arrived debris. Such surveys form the basis of major monitoring programmes established in the USA and western Europe (table 1). Such fine-scale studies can reveal surprising linkages between long-term patterns in litter accumulation rates and large-scale climatic cycles (Morishige *et al.* 2007). However, they require much more effort than do surveys of standing stocks, and substantial investment to conduct them routinely over a sufficiently large number of sites to track trends in debris abundance accurately (Sheavly 2007).

Escardó-Boomsma *et al.* (1995) made the distinction between loading rate (the amount of litter arriving on a beach) and net accumulation rate (the amount of litter that accumulates per unit of time). Turnover of plastic

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on beaches can happen rapidly. Scoring the amount accumulated at a predetermined interval will influence the estimate of accumulation rate. Most studies to date have sampled at monthly or quarterly intervals (e.g. Garrity & Levings 1993; Madzena & Lasiak 1997; table 1), but some more frequent surveys have been conducted (two weeks, Morishige *et al.* 2007; 3 days, Vauk & Schrey 1987; daily, Swanepoel 1995). There have been few attempts to assess the impact of sampling interval on estimates of accumulation rate. The US National Marine Debris Monitoring Program (USMDMP) found no difference in accumulation rate for samples collected at different intervals (Sheavly 2007), but failed to report the range of intervals tested. Walker *et al.* (1997) reported greater accumulation rates in winter, when sampling was monthly, than in summer, when litter could be sampled only at the end of the season; however, it was not possible to discriminate whether this was a sampling artefact or a seasonal difference in the rate of litter accumulation. The daily accumulation rate of all debris at two sites near Cape Town, South Africa, was 50–60% greater by mass and 100–600% greater by number compared with weekly sampling (Swanepoel 1995). This suggests that the loading rate of small items, in particular, is grossly underestimated by weekly sampling. The actual magnitude of this effect is dependent on turnover rates of debris, which are influenced by local conditions (Bowman *et al.* 1998) and the type of debris (in the Cape Town study, lighter debris such as foamed polystyrene turned over more rapidly, probably because it was blown away by the wind). Of course, frequent surveys fail to measure the rate of loss of stranded litter, but they provide the best proxy for at-sea abundance of litter.

Additional issues need to be considered when planning accumulation surveys. The initial cleanup is unlikely to locate all debris, so several collections may be necessary before the data reflect actual accumulation rates. In addition, there remain the problems of lateral drift from adjacent, uncleaned areas and exhumation of buried debris. Lateral drift may be minor (Garrity & Levings 1993) and can be addressed by cleaning buffer zones on either side of the monitoring area. Pilot studies tracking the movement of marked litter items should help determine the appropriate buffer width, which is likely to be site and season specific, linked to local current and wind conditions. Exhumation of buried debris is harder to tackle. Probably the best that can be achieved is to record major changes in beach profile. Such records might explain anomalies in the data series during subsequent analysis.

Accumulation studies are more labour-intensive than standing-stock surveys and require substantial resources to be conducted at multiple sites. Well-trained volunteers can assist with the process (Sheavly 2007). A less demanding alternative is to sample accumulation rates sporadically, recording accumulation on a daily or weekly basis for a period, then repeating the exercise several years hence (e.g. Velander & Mocogni 1998). However, consideration needs to be given to the variability in daily or weekly accumulation rates when deciding the appropriate duration for each bout of sampling. Daily sampling is more variable (coefficient of variation 47–60% by number, 61–86% by mass) than weekly sampling (CV 28–42% by number, 19–38% by mass; Swanepoel 1995; P. G. Ryan, unpublished data), suggesting that longer intervals between surveys buffer some of the short-term variability in accumulation rates linked to local conditions (e.g. wind direction and sea state).

(c) *Sampling meso- and micro-plastics*

Meso- and micro-debris on beaches differ from macro-debris, in that these categories of litter are less likely to derive directly from beach users and are not targeted by most cleanup efforts (floatation systems that separate out low-density debris are costly and not widely used). Accumulation studies are not feasible because they require a thorough initial cleanup. It is possible that changes in beach-cleaning regimes may cause a bias if a substantial proportion of small plastic fragments derive from the fragmentation of large litter items *in situ*, but this problem does not apply to virgin pellets. Repeated surveys of beach meso-debris can detect changes in their abundance (e.g. Ryan & Moloney 1990), but the results are hard to interpret if the goal is to detect changes in the amount of debris at sea. Further, without an understanding of the dynamics of meso-debris on beaches, it is hard to predict how a change in the at-sea abundance of debris will affect beach loads. If turnover in beach litter is rapid, one would expect a decrease in the amount of litter at sea to result in a decrease in beach load. However, if beaches accumulate meso-debris, a decrease in litter at sea would manifest in only a slowing in the rate of increase in beach load. Changes in the proportions of different debris types presumably indicate changes in their relative abundance at sea.

Another problem with sampling meso- and micro-debris on beaches is the design of representative sampling protocols. Sampling of micro-debris is still in its infancy, with baseline data largely confined to the presence/absence of different polymers (Thompson *et al.* 2004; Ng & Obbard 2006). Initial studies of meso-debris were also largely qualitative, sufficient only to detect gross changes in abundance (e.g. Gregory & Ryan 1997). Quantitative estimates of meso-debris have been obtained by sieving beach samples, typically to a depth of approximately 50 mm, and sorting samples including the use of floatation in sea water (e.g. Ryan & Moloney 1990; McDermid & McMullen 2004). However, little attention has been paid to sampling design and statistical power. Although debris is concentrated in strandlines, point sampling (Moore, S. L. *et al.* 2001; McDermid & McMullen 2004) is likely to miss old, buried lines. Sieving a strip transect from the most recent strandline to the back of the beach is a more reliable way to characterize meso-debris loads (e.g. Ryan & Moloney 1990).

(d) *Best practice for beach surveys*

The lack of consistency among surveys of beached plastics to date results in part from different goals, which include the need to assess the amount and composition of litter, to identify the sources of litter and to monitor changes in litter loads. Although these goals may favour different sampling approaches, there is

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need for greater standardization. The United Nations Environment Programme (UNEP) is currently developing a set of guidelines to standardize beach survey methods (Cheshire *et al.* 2009). The following recommendations are based on our best understanding of beach-litter dynamics and conform to the most commonly employed practice where possible.

For standing-stock and accumulation studies, the best approach is to record all litter from the sea edge to the highest strandline (in most cases, the edge of terrestrial vegetation). Ideally, both the numbers and mass of plastic items should be recorded, but counts may be sufficient for specific types of litter. Items should be identified as accurately as possible, allowing them to be categorized according to both composition and function. Methodologies should explicitly state the size range of litter items sampled. The minimum length of beach sampled is determined by the abundance of litter (de Araújo *et al.* 2006) and, for monitoring purposes, it should be determined by a power analysis based on estimates of variability in accumulation data from pilot studies and on considerations of the minimum rate of change to be detected (Ribic & Ganio 1996). Ideally, sample widths should be at least 50 m for standing stocks and 500 m for accumulation studies. Site selection is likely to be determined by the monitoring question (e.g. remote beaches track litter from ships and long-distance drift litter, whereas urban beaches track local inputs). However, it is important that sites be described adequately by recording substratum, slope, exposure to the open ocean, proximity to local litter sources, cleaning history, etc. (table 1). For most studies, it is best if no beach-cleaning takes place, because cleaning significantly alters the abundance and composition of debris (Moore, S. L. *et al.* 2001; Somerville *et al.* 2003). It is also crucial that the site can be relocated accurately for repeated sampling.

Comparisons of standing stocks have shown marked increases in some litter types (e.g. Ryan & Moloney 1990; Willoughby *et al.* 1997; figure 2), but this may reflect long-term accumulation rather than absolute increases in the amount of debris at sea. Accumulation studies are preferred because they demonstrate unambiguous trends in macro-debris abundance at sea (Ribic *et al.* 1992, 1997). Despite failing to demonstrate major changes in debris accumulation rates (Sheavly 2007), the USMDMP provides the best model for large-scale beach monitoring (table 1). However, it would be valuable to monitor a broader spectrum of litter types. Focusing on specific indicator groups can be useful (e.g. Józwiak 2005; Shigeru *et al.* 2006), but it may fail to detect important changes in non-target categories (Ribic 1998), including changes in plastic products in the waste stream resulting from novel packaging applications. Ideally, monitoring should take place at a network of sites, with analysis testing for common trends across sites.

Meso-debris should be sampled by a combination of sieving, dry picking and floatation to locate the greatest proportion of plastic litter. Fourier-transform infrared spectroscopy could be used to identify the polymer type of fragments collected. Surveys should sample the entire beach profile from the most recent strandline to the back of the shore to a depth of 50 mm. Minimum transect width is again determined by the abundance of litter, but should not be <0.5 m. Macro-debris sampled in this way can give an indication of buried litter loads compared with surface macro-debris (Kusui & Noda 2003).

4. SURVEYS AT SEA

If the primary goal is to monitor changes in the amount and composition of plastic debris at sea, direct surveys avoid many of the complications of beach dynamics and contamination by beach users. However, at-sea surveys are complicated by ocean current dynamics, shipboard disposal and accidental loss and are more costly and more challenging logistically, given the intensive sampling needed to detect subtle changes. Surveys at sea are also limited to assessing standing stocks rather than accumulation rates. Changes detected in the amounts of debris are the balance between inputs and losses and do not necessarily reflect the efficacy of mitigation measures to reduce losses of plastics into the environment.

(a) *Floating and suspended debris*

The abundance of floating plastics at sea can be estimated either by direct observation of large debris items (e.g. Day *et al.* 1990*a*; Matsumura & Nasu 1997; Thiel *et al.* 2003; Pichel *et al.* 2007) or by net trawls for smaller items (e.g. Carpenter & Smith 1972; Day & Shaw 1987; Ryan 1988*a*; Day *et al.* 1990*b*; Ogi *et al.* 1999; Moore, C. J. *et al.* 2001; Yamashita & Tanimura 2007). Direct observations rely on competent, motivated observers. Studies comparing detection ability show marked differences among observers (e.g. Ryan & Cooper 1989), which needs to be addressed if multiple observers are used to monitor debris at sea. Counts of litter at sea can be used to provide an index of abundance (number of items per unit distance) or an estimate of abundance based on fixed-width or line transects. Fixed-width transects assume that all debris is detected, which is unlikely unless transects are very narrow (e.g. Willoughby *et al.* 1997). For line transects, the perpendicular distance to each item has to be estimated to compensate for decreasing detection rate with distance from the observer (Buckland *et al.* 1993). This method assumes that the probability of detection on the transect line is 1, and there are problems with variable detection rates depending on sea state, light conditions and the size, colour and height above water of plastic objects. Observations should be conducted only on that side of the ship with the best viewing conditions. Separate detection curves should be estimated for different sea states, and studies should state the smallest size of items recorded.

Most surveys are conducted from ships or small boats, but aerial surveys also have been used to estimate the abundance of plastic litter at sea (Lecke-Mitchell & Mullin 1992) and to locate major aggregations of litter (Pichel *et al.* 2007). Aerial surveys cover large areas and are less prone to changes in litter detectability linked to wind strength and sea state, but they only detect large litter items. As with

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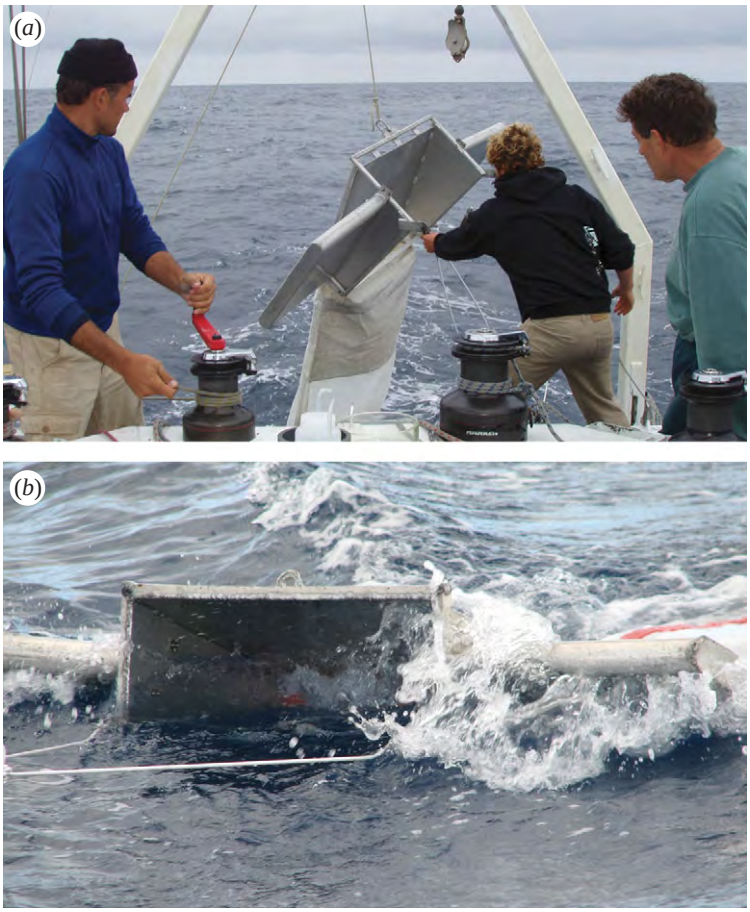

Figure 3. A manta trawl (*a*) being deployed from a research vessel and (*b*) being towed at sea to sample floating plastics.

ship-based surveys, unless inter-observer effects can be strictly controlled, aerial surveys are more valuable for detecting spatial differences in abundance than for monitoring changes over time.

Net-based surveys are less subjective than direct observations but are limited regarding the area that can be sampled (net apertures 1–2 m and ships typically have to slow down to deploy nets, requiring dedicated ship's time). The plastic debris sampled is determined by net mesh size, with similar mesh sizes required to make meaningful comparisons among studies. Floating debris typically is sampled with a neuston or manta trawl net lined with 0.33 mm mesh (figure 3). Given the very high level of spatial clumping in marine litter (e.g. Ryan 1988*a*; Pichel *et al.* 2007), large numbers of net tows are required to adequately characterize the average abundance of litter at sea. Long-term changes in plastic meso-litter have been reported using surface net tows: in the North Pacific Subtropical Gyre in 1999, plastic abundance was 335 000 items km^{-2} and 5.1 kg km^{-2} (Moore, C. J. *et al.* 2001), roughly an order of magnitude greater than samples collected in the 1980s (Day *et al.* 1990*a*,*b*). Similar dramatic increases in plastic debris have been reported off Japan (Ogi *et al.* 1999). However, caution is needed in interpreting such findings, because of the problems of extreme spatial heterogeneity, and the need to compare samples from equivalent water masses.

To date, most studies have sampled floating plastic debris, but some plastics are more dense than seawater, making it important to sample mid-water and bottom loads of plastic debris. Suspended debris can be sampled with bongo nets with a 0.33 mm mesh (Lattin *et al.* 2004). Few such surveys have been conducted, but data from the eastern North Pacific suggest that the abundance of suspended plastic within 10–30 m of the sea surface averages two orders of magnitude less than that of surface plastics (AMRF, unpublished data). All subsurface net tows should be deployed with a flowmeter to assess the volume of water sampled. The continuous plankton recorder (CPR) offers a valuable subsurface tool to track changes in the distribution and composition of micro-plastic particles at sea, both spatially and temporally (Thompson *et al.* 2004).

(b) *Litter on the seabed*

Surveys of macro-debris loads on the seabed have been conducted with divers (e.g. Donohue *et al.* 2001; Nagelkerken *et al.* 2001), submersibles and remote-operated vehicles (Galgani *et al.* 2000) and trawl surveys (e.g. Galil *et al.* 1995; Galgani *et al.* 2000; Moore & Allen 2000; Lattin *et al.* 2004; OSPAR Commission 2007*b*). Perhaps somewhat surprisingly, plastics dominate macro-debris on the sea floor to an extent similar to which they dominate floating litter and beach debris. Just like stranded debris, plastic on the seabed aggregates locally in response to local sources and bottom topography (Galgani *et al.* 2000; Moore & Allen 2000). The amount of plastic litter is

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so great in some areas with large amounts of shipping traffic that initiatives have been started to clean the seabed with trawls (OSPAR Commission 2007*b*), despite concerns about the ecological impacts of trawling. To date, most studies have measured standing stocks of macro-debris, but some accumulation data have been obtained following cleanups of shallow reefs in Hawaii (Boland & Donohue 2003; Dameron *et al.* 2007). The rate of litter accumulation on these reefs is correlated with initial standing stock and is a function of reef exposure and depth (Dameron *et al.* 2007). There has been little attention to the abundance of meso- and micro-debris on the seabed. Epibenthic trawls have found substantial plastic loads just above the seabed in shallow coastal waters off southern California (Lattin *et al.* 2004). Bottom sediments in deeper waters can be sampled with a Van Veen grab or similar device. Micro-plastics have been found in subtidal sediments around the UK and Singapore (Thompson *et al.* 2004; Ng & Obbard 2006).

(c) *Best practice for at-sea surveys*

Effective monitoring of floating plastics at-sea requires huge sample sizes to overcome the very large spatial heterogeneity in plastic litter. Stratified random sampling can help with this issue, but it requires *a priori* categorization of water masses into the relevant sampling strata. If resources are available, probably the best tool is to sample with neuston nets with a 0.33 mm mesh. Direct observations, often using vessels of opportunity, are less resource-intensive, but are fraught with potential biases linked to differences in litter detectability. Such surveys provide only a crude index of the abundance of floating litter. Much less is known about the distribution and abundance of mid-water plastics, but they probably suffer the same sampling problems, with the added complication of even lower abundances. The CPR is a useful tool for long-term subsurface monitoring of micro-particles.

Monitoring changes in benthic plastic litter is functionally similar to beach surveys, with the added complication of working underwater. Divers can replicate beach sampling protocols in shallow water, but in deeper waters there are greater issues with quantitatively robust sampling owing to variation in trawl and grab efficiency (linked to substratum type and other local conditions). Trawl nets also become clogged, reducing their efficiency and thus underestimating actual plastic abundance. Remote cameras may provide a more objective sampling strategy for benthic litter.

5. MONITORING AFFECTED SPECIES

One of the major concerns about the accumulation of plastic debris in the ocean is the impact on the biota. Consequently, it is sensible to monitor the rates at which impacts occur, although it only provides information on specific types of plastic litter. The two main impacts are entanglement and ingestion, but monitoring other interactions with wildlife also can detect useful trends, such as the amount of litter incorporated in seabird nests (Hartwig *et al.* 2007).

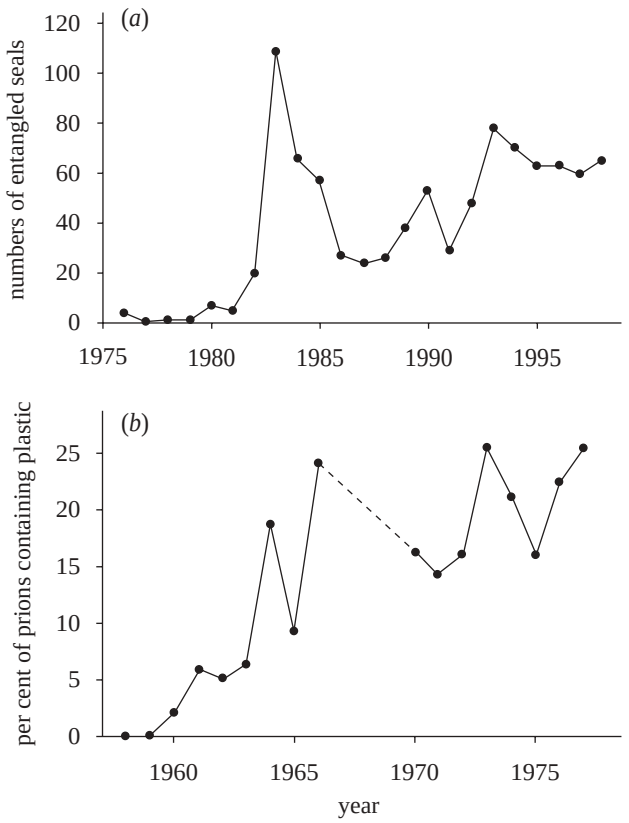

Figure 4. Long-term trends in the impacts of plastics on marine animals: (*a*) numbers of entangled seals recorded annually on SE Farallon Island, California, 1976–1998 (reproduced with permission from Hanni & Pyle 2000); (*b*) proportions of prions *Pachyptila* spp. stranded on New Zealand beaches that had plastics in their stomachs, 1958–1977 (reproduced with permission from Harper & Fowler 1987).

(a) *Entanglement*

Entanglement is one of the more visible impacts of plastic debris, affecting a large number of marine and fresh-water species (Laist 1997). Monitoring the number of entangled organisms can indicate changes in the abundance of debris responsible for entanglements. For example, entanglement records for three species of seals from the Farallon Islands show a steady increase since the 1970s, but with a marked peak from 1983 to 1985 (Hanni & Pyle 2000; figure 4*a*). Camphuysen (2001) reported increased entanglement rates of Northern Gannets *Morus bassanus* in the North Sea from 1997 to 2000. Such data can be used to assess whether mitigation measures have been effective. There was no decrease in entanglement rates among Hawaiian Monk Seals *Monachus schauinslandi* following the introduction of MARPOL Annex V banning the disposal of plastics at sea (Henderson 2001), but most entanglements are from fishing gear lost at sea (not discarded). However, there was no decrease in entanglement of two seal species in Australia, despite efforts to reduce the amount of fishery debris being lost at sea (Page *et al.* 2004). At South Georgia, the proportion of entangled Antarctic fur seals *Arctocephalus gazella* halved following active promotion of MARPOL Annex V regulations, but the population of seals

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roughly doubled over the same period, suggesting that there was no decrease in the amount of litter at sea and that the total number of seals affected may have increased (Arnould & Croxall 1995). This example illustrates the need to interpret results with caution. Entanglement tends to be quite rare (Laist 1997), making it hard to obtain sufficient data to detect a significant change in the rate of entanglement. It is only really feasible to use as an index of plastic litter if there is constant search effort.

(b) *Plastic ingestion*

Ingestion of plastic debris occurs much more frequently than entanglement, with almost all individuals of some species containing ingested plastic (e.g. Ryan 1987; Laist 1997; Robards *et al.* 1997). Indeed, the first indication that plastics were a problem in marine systems came when plastic fragments were found in the stomachs of seabirds in the 1960s. Prions *Pachyptila* spp. in New Zealand showed a steady increase in the incidence of plastic ingestion from the 1960s to the 1970s (Harper & Fowler 1987; figure 4*b*), and 74 per cent of albatross *Phoebastria* spp. chicks found dead on Laysan Island, Hawaii, contained plastics in 1965 (Kenyon & Kridler 1969). Seabirds that accumulate plastics in their stomachs (especially petrels and storm-petrels, Procellariiformes) are useful indicators of changes in the amount and composition of plastic debris at sea. They collect debris over large areas and can be sampled with little cost by examining the stomach contents of beached birds (Harper & Fowler 1987), birds killed accidentally by fishing activities (Mallory *et al.* 2006; Ryan 2008) or by examining regurgitated pellets of predators that feed on seabirds (Ryan 2008).

Plastic loads in birds reflect regional differences in the abundance of marine debris (Day *et al.* 1985; van Franeker 1985; van Franeker & Bell 1985; Spear *et al.* 1995). For example, the amount of ingested plastic in Northern Fulmars *Fulmarus glacialis* in the North Atlantic is greatest in highly contaminated waters of the North Sea, where almost all birds contain some plastic (van Franeker *et al.* 2005), decreasing to only 36 per cent of birds in arctic Canada (Mallory *et al.* 2006). Even within the North Sea, regional differences are apparent, with average plastic loads doubling from the Faroe Islands to Scotland and doubling again from Scotland to the southern North Sea (van Franeker *et al.* 2005). These regional differences suggest that there are marked differences in the abundance of meso-debris between these regions, provided that the foraging areas of birds are largely confined to these regions.

For monitoring purposes, it is important to understand the factors that influence the amount of plastic in birds' stomachs (Ryan 2008). Seabirds select the types of plastic fragments they ingest (Day *et al.* 1985; Ryan 1987), but comparisons within species should have a consistent bias. Plastic loads in Northern Fulmars found dead on the Dutch coast are affected by age but not by sex, season, level of starvation or cause of death (van Franeker *et al.* 2005). Young birds typically contain more plastic than adults, probably because of the transfer of plastic from parents to their offspring (Ryan 1988*b*) exacerbated by poor discrimination of suitable food items by naive birds (Day *et al.* 1985). Another issue is how ingested plastic loads change with increases in the abundance of plastic at sea. Is there a linear relationship, or do birds reach a point where they become saturated? This question has not been resolved (Ryan 2008).

Monitoring of plastic loads in seabirds initially showed increases in plastic ingestion from the 1960s to the 1980s (figure 4*b*; Day *et al.* 1985; Moser & Lee 1992), but plastic loads have stabilized or decreased more recently, with significant changes in the composition of ingested plastic. In the North Pacific, plastic loads in Short-tailed Shearwaters *Puffinus tenuirostris* did not change significantly from the 1970s to the late-1990s, but virgin pellets were replaced by fragments of user plastics, decreasing from 55–73% in the 1970s to 33 per cent in the 1990s (Vlietstra & Parga 2002). Similar decreases in the proportions of virgin pellets have been found in five other seabirds foraging in the Atlantic and southern Indian Oceans from the 1980s to 2000s (Ryan 2008). Among Northern Fulmars stranded on Dutch beaches, there has been a long-term decrease in the mass of virgin pellets over the last 20 yr and, after peaking in the 1990s, the total mass of ingested plastic has decreased over the last 10 yr (figure 5). There thus appears to have been a global decrease in the abundance of virgin pellets at sea over the last two decades, which, with the exception of fulmars in the North Sea, has been offset to a degree by increases in user plastics. These insights have been obtained at a fraction of the cost of ship-based surveys of meso-debris abundance at sea.

6. MONITORING INPUTS

The most direct measure of success in the campaign against plastic pollution is to monitor the amounts of waste plastic entering the marine environment. This is no trivial undertaking, given the wide range of sources of plastic debris. Two main sources of plastic debris can be considered separately: ship- and land-based (Coe & Rogers 1997).

(a) *Ship-based sources*

Initial concerns about debris in marine environments focused largely on ship-based sources. Early attempts to assess the amount of waste disposed of by vessels at sea (Dixon & Dixon 1981; Pruter 1987) provided crude estimates of the amount dumped (Rees & Pond 1995), but there have been no formal estimates since MARPOL Annex V came into force in 1988. Estimates of compliance with the ban on disposal at sea imposed on signatories of MARPOL Annex V have been obtained from the use of port reception facilities (Carpenter & Macgill 2005). In addition, independent fishery observers can be tasked to report on disposal practices on fishing vessels (Jones 1995; Walker *et al.* 1997).

(b) *Land-based sources*

Although some plastic debris is transported by wind, most land-based litter is carried by water via rivers

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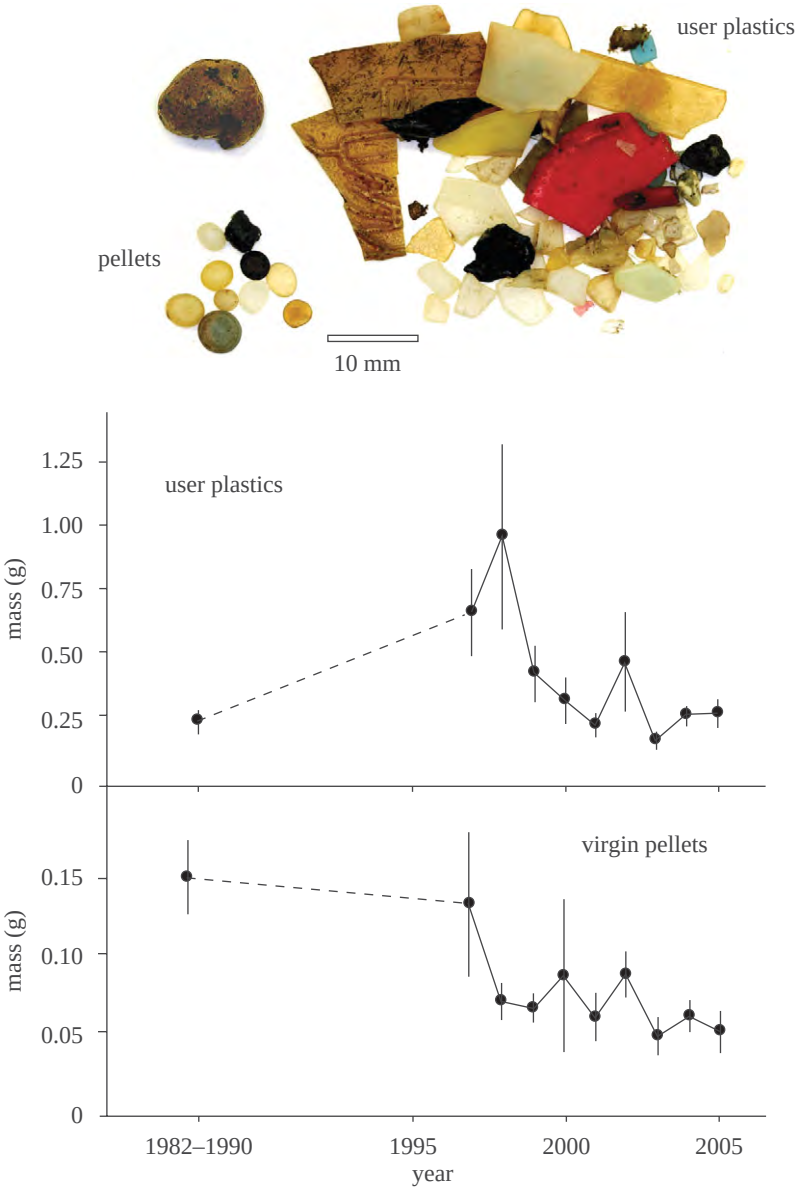

Figure 5. Typical plastics from a Northern Fulmar stomach (top panel) and trends in the average mass (±s.e.) of user plastic and virgin plastic pellets in Northern Fulmars stranded on Dutch beaches in 1982–1990 ($n = 69$) and 1997–2005 ($n = 580$).

Table 2. Sampling protocols used to assess plastic debris loads in urban runoff by the Algalita Marine Research Foundation. Water-based samples are collected upstream of debris booms, above the influence of the tidal prism.

habitat sampled	collection device	net aperture (m)	mesh size (mm)
surface; mid-stream	manta trawl	0.9 × 0.15	0.33
surface, edge	hand net	0.46 × 0.25	0.5 or 0.8
mid-water	weighted net	0.46 × 0.25	0.33
bed load	streambed sampler	0.15 × 0.15	0.33
bottom sediments	scoop	15 l bucket[a]	—

[a]Fifteen litres of bottom sediments are collected to a depth of 100 mm; in cement-lined canals, samples are collected where the concrete bottom terminates and natural bottom begins.

and storm-water. Few attempts to quantify the magnitude of litter in runoff have been published in the primary literature (see papers in Coe & Rogers 1997), but numerous studies have been conducted, often as part of programmes to educate the public to the dangers of inappropriate disposal of plastics. Most studies quantify plastic loads in runoff by sampling litter captured in a variety of filtering systems (e.g. Durrum 1997; Armitage & Roseboom 2000) or in customized nets (table 2). However, one study in Melbourne, Australia, released marked litter items in street-side storm-water drains and then used schoolchildren to locate them on surrounding beaches, thus establishing the link between street litter and beach litter.

To date, most studies of litter in urban runoff have focused on macro-debris (e.g. Marais & Armitage

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2004; Marais *et al.* 2004). The most comprehensive survey of small plastic litter has been conducted by the Algalita Marine Research Foundation (AMRF) in southern California (www.algalita.org; table 2), which recorded up to 81 g m^{-3} of small plastic items in storm-water discharges. A key challenge is to cope with the great temporal heterogeneity in plastic loads linked to rainfall events. Litter loads build up between rain events and are then flushed into the receiving water body. Sampling high-flow events is complicated by clogging of sampling nets or grids. Passive litter traps risk blocking drains or losing samples through the tearing of nets. Surveys of litter on land can also provide useful information on debris abundance. The AMRF conducted site inspections of plastic converters in the Los Angeles basin to assess pellet spillages, then re-surveyed these businesses following the implementation of best management practices. This intervention reduced pellet spillage by 80 per cent.

7. CONCLUSIONS

Just as multiple initiatives are needed to tackle the marine litter problem (Coe & Rogers 1997), diverse approaches are required to monitor the abundance of plastics in marine environments. For any monitoring programme, the objectives must be clearly stated, the methodology clearly defined and quality control implemented to ensure quality data. Sampling design needs to be cognizant of the dynamics of plastic in the environment. Debris monitoring is complicated by large spatial and temporal heterogeneity in the amounts of plastic debris. Pilot studies should be used to estimate variability in sample data, and then power analysis should assess the numbers of samples necessary to detect a predetermined change.

By selecting beaches at varying distances from major litter sources, beach surveys can provide useful insights into the origins of plastic debris. Monitoring of stranded litter should concentrate on estimating the accumulation rate of debris on beaches, because this gives a measure of the amount of litter at sea. However, this effort adds considerably to the cost of beach surveys. Accumulation rates are sensitive to sampling interval; frequent sampling reduces biases owing to rapid debris turnover, but estimates are more variable than longer sampling intervals, requiring greater sampling effort. At-sea sampling is also extremely expensive, but net-based samples suggest that there have been marked increases in plastic litter in accumulation zones in oceanic gyres. Seabirds that accumulate plastic in their stomachs provide an inexpensive, powerful tool to monitor changes in the abundance and composition of small plastic debris at regional scales. Monitoring the incidence of entanglement also is useful, provided that search effort is constant and that interpretation acknowledges changes in the population size of affected organisms.

Assessing the magnitude of debris sources is complicated by the large number of point and diffuse sources of plastic debris (figure 1). However, this approach has the advantage of assessing directly whether specific mitigation efforts are having the desired effect. Dumping of ship-based litter can be assessed by independent observers on vessels and by monitoring the use of port reception facilities. Land-based sources can be monitored by sampling rivers, storm-water run-off and other key sources. In tandem with appropriate education programmes, measurement of sources can be effective in promoting changes in disposal practices that ultimately reduce the amounts of plastic entering the environment.

We thank our colleagues and many volunteers who assisted with data collection and tolerated the sometimes smelly business of collecting beached birds and sorting litter. P.G.R. and C.L.M. received financial support from the National Research Foundation and the Plastics Federation of South Africa. C.J.M. and the Algalita Foundation received financial support for ocean sampling aboard ORV *Alguita* from the Will J. Reid Foundation. Fulmar monitoring in The Netherlands was commissioned by the Ministry of Transport, Public Works and Water Management (VenW) and is possible thanks to volunteer assistance from the Dutch Seabird Group (NZG). North Sea wide monitoring was established by EU Interreg support and is continued with funds from NYK Group Europe Ltd and Chevron Upstream Europe.

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Phil. Trans. R. Soc. B 2009 **364**, 2141-2151
doi: 10.1098/rstb.2008.0283

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PHILOSOPHICAL TRANSACTIONS OF THE ROYAL SOCIETY B

Phil. Trans. R. Soc. B (2009) **364**, 2141–2151
doi:10.1098/rstb.2008.0283

Structuring policy problems for plastics, the environment and human health: reflections from the UK

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How can we strengthen the science–policy interface for plastics, the environment and human health? In a complex policy area with multiple stakeholders, it is important to clarify the nature of the particular plastics-related issue before trying to understand how to reconcile the supply and demand for evidence in policy. This article proposes a simple problem typology to assess the fundamental characteristics of a policy issue and thus identify appropriate processes for science–policy interactions. This is illustrated with two case studies from one UK Government Department, showing how policy and science meet over the environmental problems of plastics waste in the marine environment and on land. A problem-structuring methodology helps us understand why some policy issues can be addressed through relatively linear flows of science from experts to policymakers but why others demand a more reflexive approach to brokering the knowledge between science and policy. Suggestions are given at the end of the article for practical actions that can be taken on both sides.

Keywords: science–policy interface; knowledge brokering; problem structuring; plastics policy

1. INTRODUCTION

In a previous Theme Issue of this journal, Robert Watson outlined many of the challenges at the interface between science and policy.[1] Discussing the role of national and international assessments, Watson (2005) outlines how science can—with appropriate participation from all the key stakeholders—provide the evidence that forms the basis for policy discussions. International assessments such as those on stratospheric ozone depletion, climate change, biodiversity and science and technology for agriculture make important contributions to the global store of knowledge. They set the high-level agenda, bring the topic to a wide audience and focus expert discussions on the main issues.

Equally important, however, are the day-to-day operations that characterize the policy process, from strategy formulation through policy development to policy delivery. In spite of the publication of guidelines on the use of science in policy (HM Government 2000, 2005), Nutley *et al.* (2007) note that it is a source of frustration to both researchers and policymakers that robust evidence, presented clearly, does not always have the desired effect on policy processes (see also Scott *et al.* 2005; ERFF 2007; McNie 2007; Bielak *et al.* 2008). This article focuses on the ongoing interactions between scientists and policymakers, drawing in particular on my experience at the interface between scientific research and environmental policy in the UK.[2]

How policies are developed will vary from country to country, depending on the politico-administrative system that has emerged and any supra-national agreements that may be in force (Albæk 1995; Jasanoff 2005). A useful analytical framework is provided by Sarewitz & Pielke (2007) who focus on reconciling the supply of, and demand for, scientific evidence in policy. In a thorough review of the challenges associated with this, McNie (2007, p. 18) notes that this reconciliation is not a single event, but it is '. . . about process: facilitating our exploration and assessment of the question: are we doing the 'right' science to better respond to society's needs?' The challenge in this article is to unpack these processes in detail by looking at some of the ways science and policy meet inside a government department. The first section outlines some of the complexities of plastics policy-making in the UK. The second limits this complexity using an issue typology to characterize policy issues. Using this problem-structuring approach helps explain why there is no one 'science–policy interface': instead, there are many different ways in which policy and science interact. The third section describes two of these in short case studies that analyse the organizational processes put in place to reconcile the supply and demand for evidence relating to plastics waste. Finally, the implications of applying a problem-structuring approach are summarized in some practical suggestions for both science and plastics policy.

2. THE POLICY ENVIRONMENT FOR PLASTICS, THE ENVIRONMENT AND HUMAN HEALTH

As an employer of around 220 000 in the UK across 6000 businesses, the plastics industry is a major

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One contribution of 15 to a Theme Issue 'Plastics, the environment and human health'.

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player in the UK economy: the total annual value of the plastics processing sector alone is estimated to be £12.4 billion (HSE 2008). Plastics is one of the seven key waste materials identified in the UK's Waste Strategy 2007: the total amount of plastic waste arising is estimated to be 5.9 million tonnes (mt) per annum: 2.3 mt from household and other municipal sources, 2.5 mt arising in the commercial waste stream (1.9 mt from packaging) and 0.8 mt and 0.1 mt from industrial and agricultural waste streams, respectively (Defra 2007*a*, Annex D).

In spite of the size of the industry and the other contributions plastics make to the British economy, there is no single plastics policy. Instead, it is shared between at least five government departments and driven by both international obligations and European directives. The Department for Business, Enterprise and Regulatory Reform (BERR) seeks to ensure the success of UK business, focusing on industrial productivity and the contribution of the plastics industry to the British economy. BERR sponsors the Polymer Innovation Network, one of the Materials Knowledge Transfer Networks designed to bring together '...the views of all in business, design, research and technology organisations, trade associations, the financial market, academia and others...' (Materials KTN 2008). The Department for Environment, Food and Rural Affairs (Defra), whose overarching goal is the delivery of a low-carbon economy (Defra 2008*a*), develops policy around the environmental impacts of plastics arising from plastic waste on land or at sea. Defra also leads UK policymaking under two major EU directives affecting the plastics industry. The Energy Using Products Directive covers eco-design and end-of-life waste management requirements for all energy-using products except motor vehicles (http://ec.europa.eu/energy/demand/legislation/eco_design_en.htm). REACH, the European directive on the Regulation, Evaluation, Authorisation and Restriction of Chemicals, makes specific provisions for monomers, polymers and some of the additives used in plastics manufacture (http://reach.jrc.it/docs/guidance_document/polymers_en.pdf). Human health issues relating to plastics are addressed by the Department of Health (public health), the Medicines and Healthcare Products Regulatory Agency (a wide range of medical devices containing plastics), the Health and Safety Executive (health and safety within industries producing and handling plastics) and the Food Standards Agency, which is charged with implementing the Materials and Articles in Contact with Food Regulation 1987 and subsequent amendments (HM Government 1987).

Each organization works closely with bodies that deliver policies such as implementing regulations about water quality, reducing the health impacts of chemical hazards, imposing charges for waste and negotiating voluntary energy efficiency standards. Both individually and jointly they sponsor a variety of groups designed to bring together industry, policymakers, academic research and non-governmental organizations around particular issues. Within this complex organizational environment, different types of policy are formulated, appraised, implemented and evaluated. The next section examines these using a problem typology, which describes the nature of the policy problem and thus helps understand how best to link science and policy to deal with it.

3. DIFFERENT POLICY TYPES GIVE RISE TO DIFFERENT INTERACTIONS BETWEEN SCIENCE AND POLICY

A policy can be defined as '...a set of patterns of related decisions to which many circumstances and ...influences have contributed' (Hogwood & Gunn 1984, p. 24). Public policy can result in concrete plans, which themselves may result in specific proposals for action including regulation, economic instruments, such as subsidies or taxes, or programmes of legislation with accompanying organizations and resources. But it does not only encompass these sorts of legislated actions. 'Policy' may also result in voluntary negotiated standards such as the tripartite negotiations between government, retailers and industry to limit the use of plastic carrier bags. It also covers risk governance such as understanding the impacts and potential hazards of novel plastics, decisions about the allocation of public funds via research prioritization, the provision of information via (for example) ecolabelling strategies and marketing to 'win hearts and minds' (Collins *et al.* 2003, p. 3). However, it is made, policy cannot be characterized by a single decision point. Instead, it is a series of decisions—one of which may be crucial in determining the ultimate direction of the policy—but all of which contribute to how it is planned and implemented.

Different policymaking processes require different tools. Hogwood & Gunn (1984, p. 62) point out that '...different policy issues will require different policy approaches. Some...will always require a highly political, pluralist, bargaining and incrementalist approach...(while)...other issues—probably only a small minority—will both require and lend themselves to a much more planned or analytical approach'. However, UK Government guidelines on the use of scientific advice in policy do not pick up on this. Instead, they state simply: 'Individual departments should ensure that their procedures can anticipate as early as possible those issues for which scientific advice will be needed, particularly those which are potentially sensitive. They should also ensure that research is commissioned as early as possible into what are known or likely to be key areas of uncertainty' (HM Government 2000). They give little indication of *how* to involve the relevant experts and other stakeholders and to determine which approach is appropriate.

Hisschemöller & Hoppé (2001) proposed a typology for analysing policy problems, depending on whether there is consensus on the questions policy is addressing and certainty about the relevant knowledge (table 1). It is pertinent to the issue addressed in this article—how to reconcile supply and demand for science in policy. Here, policy demand is expressed via the questions it asks of science about an issue. Science supply is expressed through the knowledge it provides in answer to the question. With this framework, we can analyse how different types of policy problems give rise to different relationships between science and policy.

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Table 1. Structuring policy problems. Adapted from Hisschemöller & Hoppé (2001), Snowden & Boone (2007) and Shaxson (2008).

		is there consensus and clarity in the policy question . . .	
		no	yes
. . . and clarity about the relevant knowledge	no	unstructured problem policy issues are 'complex' domain of emergence	moderately structured problem policy issues are 'knowable' domain of experts
	yes	badly structured problem policy issues are 'chaotic' domain of rapid or symbolic responses	well-structured problem policy issues are 'known' domain of best practice

(a) *Well-structured problems*

A well-structured policy problem is characterized by '. . . clear cause and effect relationships that are easily discernable by everyone . . . the right answer is self-evident and undisputed' (Snowden & Boone 2007, p. 2). The questions for policy are well known, and the evidence is well understood. This is the domain of best practice and published guidance, which policy does through regulation or by agreeing voluntary negotiated standards. For example, once legislation on water quality is in place, the policy question of whether or not a particular industry should be penalized for a release of chemicals into a water body is well structured—it can be resolved by the yes/no answer of whether the chemical load in the water has breached agreed limits. Because the question is clear and the answer can be known with certainty ('Is the chemical load above regulated limits'?), science has a clear and leading role to play in policy's decision-making process.

However, only a few policy problems are likely to be classifiable as well structured: those where the question is unambiguous and the relevant evidence is clear. For many issues, the evidence may be contested by different interest groups who have reservations about its robustness or challenge the assumptions on which it is based. In our water quality example, questions might be raised about whether water sampling techniques lead to false-positive results. If this is the case, we need to reassess how well the problem is structured. This leads us into the second domain: that of moderately structured problems.

(b) *Moderately structured problems*

A moderately structured problem occurs when the policy questions are agreed, but either there is a degree of uncertainty over the best way to answer them, or a piece of knowledge that should form the basis for a particular policy process is missing or ambiguous (Vesely 2007). The problem can be described as knowable (Kurtz & Snowden 2003) because while a cause-and-effect relationship may exist, it is not visible to everyone. However, once the relevant specialists have been gathered and set a clear question, their analysis helps policymakers converge on answers: this is the domain of experts. Our policymakers concerned with the chemical leak will need science to use its expertise to choose the most appropriate indicator species to represent the health of our putative water body. However, if evidence emerges suggesting that low-dose effects of the leaked chemical are cumulative in the food chain, then policymakers will need to ask whether—given the goal of maintaining water quality—they are still basing their decisions on the best available evidence.

Science plays a prominent role in this domain, but can only do so because policy has set clear questions that are amenable to solution through research-based and other analytical techniques. Having said that, a clear question does not mean that there is only one solution. Contributions made by different disciplines such as the natural sciences, economics, law, statistics and social science all need to be taken into account, even when their conclusions may oppose each other. Where value for money is the over-riding consideration, the ultimate arbiter will be cost–benefit analyses of the possible solutions. In other cases, decisions may give greater weight to evidence from research programmes or expert advisory groups. Where different actors have different interests, policymakers will be faced with '. . . different solutions for the problem (which) may have far going implications for the distribution of costs and benefits' (Turnhout *et al.* 2007, p. 224).

The danger in this domain is that evidence is sought from a narrow range of sources, leaving both policymakers and experts blind to novel solutions and unaware that the wider context of the problem has changed. Both may continue to assume a degree of structure for a problem that has become either badly structured or unstructured.

(c) *Badly structured problems*

Badly structured problems arise where divergent views are expressed about a variety of issues in a seemingly chaotic way (Shaxson 2009). There may be agreement on what the evidence is, but the different values held by different interest groups will cause them to disagree over what the evidence means or even what question it is really answering. Searching for cause and effect becomes meaningless because new evidence causes confusion rather than clarity. Science may be prominent in this domain, but instead of providing answers, it brings attention to new possibilities that society may find very challenging. The early debates around nanotechnology could be described as badly structured: rapidly emerging and sometimes conflicting scientific information about nanomaterials provoked a value-laden public debate about 'grey goo' fuelled by diverse sources such as the Prince of Wales (Highfield 2003) and science fiction (Phoenix & Drexler 2004). In this domain, politicians may need to respond rapidly if they perceive

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there to be a crisis (Snowden & Boone 2007): alternatively, they may respond by defining vague or symbolic goals such as 'sustainability' (Turnhout *et al.* 2007) without necessarily describing them in detail.[3] Different horizon scanning and futures techniques can be used in this domain, playing an important role in examining what might happen and what the preferred outcomes might be and describing the possible future contexts within which policy can plan to deliver the needed changes (Bochel & Shaxson 2007).

(d) *Unstructured problems*

With unstructured problems, there is agreement neither on what the real policy questions are nor the state of the knowledge on which policies are based. 'Conflicting values and facts are interwoven, and many actors become involved in the policy process' (Hisschemöller & Hoppé 2001, p. 51). This results in a domain, where cause and effect can only be seen in retrospect, once the different points of consensus have emerged (Snowden & Boone 2007).

Scientists may be frustrated that a strong message, clearly communicated, does not promote consensus about an issue, which they believe to be 'closed'. This indicates a misperception of the problem structure: many plastics issues such as plasticizers and brominated flame retardants, for example, are unstructured. Scientific information on phthalates and bisphenol A, for example, can never form the basis of a societal consensus about how they should be used and whether that use should be regulated. Their widespread use in everyday products (Koch & Calafat 2009) combined with information on low-dose effects of the chemical to both human health (vom Saal & Welshons 2006) and wildlife (Oehlmann *et al.* 2009) means that the context of the policy question is not 'what levels are safe'? but 'what levels are safe *enough*'? Ethical issues are involved in determining what 'enough' means to different parts of society, implying that however much research is done, the definition of 'safe enough' is not a question that can ever be answered by plastics science alone. There is no clear tradeoff between the benefits to society of using plastics and the hazards to human health and the health of the natural environment of phthalates or bisphenol A (but see Andrady & Neal (2009) and Thompson *et al.* (2009*a*,*b*)). Nor can we predict the 'right' method for deciding what that tradeoff might be.

Cost–benefit analysis and other forms of research—standard tools for dealing with moderately structured problems—cannot be used to deal with unstructured problems because of disagreements about the assumptions on which they are based. In the unstructured domain, policymakers cannot impose a solution based only on science, even interdisciplinary science. Instead, scientific research is one voice that policymakers listen to as they negotiate different values and rival interests, diverse views and rival claims about whose knowledge counts, until consensus emerges or they arrive at a practical solution (RCEP 1998).

(e) *Problem structuring: summary*

Table 2 summarizes how different problem structures will give rise to different relationships between science and policy. Misunderstanding the structure of the problem will mean that scientific advice is unlikely to be translated into policy in the way initially envisaged. Both scientists and policymakers may consider a problem to be well structured if there is a clear gap in one part of the evidence, which could be rectified by better communication across the divide—often referred to as the 'deficit model' of science–policy interactions. However, if others consider that the problems are broader, ethically complex or controversial, then the policy environment becomes unstructured. Science is only one of many voices competing to be heard, and if it is the only voice that policymakers listen to, then they will limit themselves to a small part of the evidence base. While a part of the overall policy problem may be moderately structured and thus potentially amenable to solution through the supply of scientific expertise, it is likely to be nested within a wider set of problems with different structures and demands for different types of evidence. The boundaries of the problem will be diffuse, and it will be difficult to separate it from other problems (Hisschemöller & Hoppé 2001). The organizational arrangements put in place to reconcile the supply and demand for evidence in policy need to recognize this fact. If not, the processes of making policy will limit the range of evidence policymakers can listen to, limiting their range of potential choices.

The next section looks at two groups of people who collect evidence to inform policymaking around plastic waste in the UK. The first case study assesses the processes for developing and implementing indicators of the effects of plastic litter on the marine ecosystem: crossing the boundary between well- and moderately structured problems. The second shows how the move to a life-cycle approach to policymaking around land-based plastics waste has led to the development of an organizational structure, which is flexible enough to deal simultaneously with moderately and unstructured problems. This involves moving to a 'knowledge brokering' approach, which is outlined briefly prior to the case studies.

4. BROKERING THE INTERACTIONS BETWEEN SCIENCE AND POLICY: TWO CASE STUDIES OF THE SCIENCE–POLICY INTERFACE

In his book *The Honest Broker*, Pielke (2007) describes four idealized roles of science in policy and politics, which can be filled either by people or by organizations. Pure scientists have no interest in the decision-making process and simply share fundamental information about an issue. Science arbiters serve as an expert resource, providing the answers to clearly specified questions. Issue advocates make the case for one alternative over all others, either transparently or by stealth. The fourth role (examined here) is the Honest Broker of Policy Alternatives, who engages at the interface of science and policy to clarify the scope of choice for decision-making (Pielke 2007, p. 2) and help shed light on what the science means for policy.[4]

Many external organizations perform this translation function—from international assessments to think-tanks, research institutes and consultancies.

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Table 2. Different types of policy problems give rise to different relationships between science and policy.

typology	key issues	examples	relationship between science and policy
problem is well-structured: policy issues are known	policy has already set clear questions to be answered. Cause and effect are clearly defined and can be anticipated	using indicators of water quality to determine regulatory breaches by a plastics manufacturer	science leads policy by establishing best practice guidelines, giving policy a clear steer on what to do
problem is moderately structured: policy issues are knowable	policy has set clear questions, but cause and effect have not yet been firmly established so there is no agreement on the best way to answer them	developing indicators of marine pollution by man-made plastics such as the OSPAR Ecological Quality Objective relating to plastic debris in the stomachs of fulmars (Noordzeeloket 2007) developing resource-efficient methods for cracking low-quality plastics into monomers that can subsequently be reprocessed	policy identifies a clear need for science, which uses research and expert advice to provide answers. It may lead to the formulation of best practice guidance and regulations, or to an understanding that the problem is in fact more complex than anticipated
problem is unstructured. Policy issues are complex	policy has set the question (how can we make plastics more sustainable?) but there is no consensus around what that actually means. Cause and effect are only visible in retrospect, once consensus has emerged	plasticizers—there is no policy consensus about how to weigh up the economic, environmental and human health impacts of plasticizers improving household recycling of plastics, which relies on knowledge of personal consumption and disposal behaviours and how to influence them	science is one of many voices in the emerging debate: over-reliance on science will lead to poor policy choices. Policymakers need to negotiate competing values and interests of different stakeholders. Consensus may emerge, or the issue may die away of its own accord. It is only really possible to fully define the problem in retrospect
problem is badly structured. Policy issues are chaotic	cause and effect cannot be determined: information emerges rapidly from many sources. Goals are largely symbolic or set in response to a crisis	novel materials such as nanomaterials: there may be a great deal of science, but social and ethical debates have not had time to catch up with science, so the wider implications are poorly understood.	science challenges policy, but the future direction of the technology is unclear, its social and ethical impacts are unexplored and public attitudes are unfixed (Gavelin *et al.* 2007). Policy can help coalesce debate around key questions or values

Much has been done to understand how to use them to improve the supply of evidence (McNie 2007), but relatively little attention has been paid as to how the demand for evidence in policy is conditioned by organizational arrangements within government departments. To see how this demand is formed and expressed, we need to look in detail at the work of groups led from within the policy environment which advise on commissioning and using evidence.

(a) *Case study 1: developing indicators for the health of the marine ecosystem: plastic litter in the marine environment*

Marine policy in the UK is driven by several EU directives such as the Marine Framework Strategy Directive (MSFD: http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2008:164:0019:0040:EN:PDF), international conventions such as OSPAR (http://www.ospar.org) and the Convention on Biological Diversity (http://www.cbd.int) and the UK-specific legislation coming into force under the Marine Bill. The publication of *Safeguarding our Seas* (Defra 2002) committed the UK to adopting an ecosystem management approach to the seas and oceans and to an integrated assessment of the state of the seas (Defra 2005*a*), including the development and use of indicators of good environmental status.

No single body is charged with developing indicators or proposing new ones. The UK Marine Monitoring and Assessment Strategy (UKMMAS) was set up in 2006 with the goal of ensuring the cost-effective provision of the information needed to support policy and management decisions to develop the UK marine vision. It is delivered through a number of groups working together at the science–policy interface as shown in figure 1.

The Marine Assessment Policy Committee (MAPC) provides policy direction to the body charged with implementing the UKMMAS: the Marine Assessment and Reporting Group (MARG). MARG

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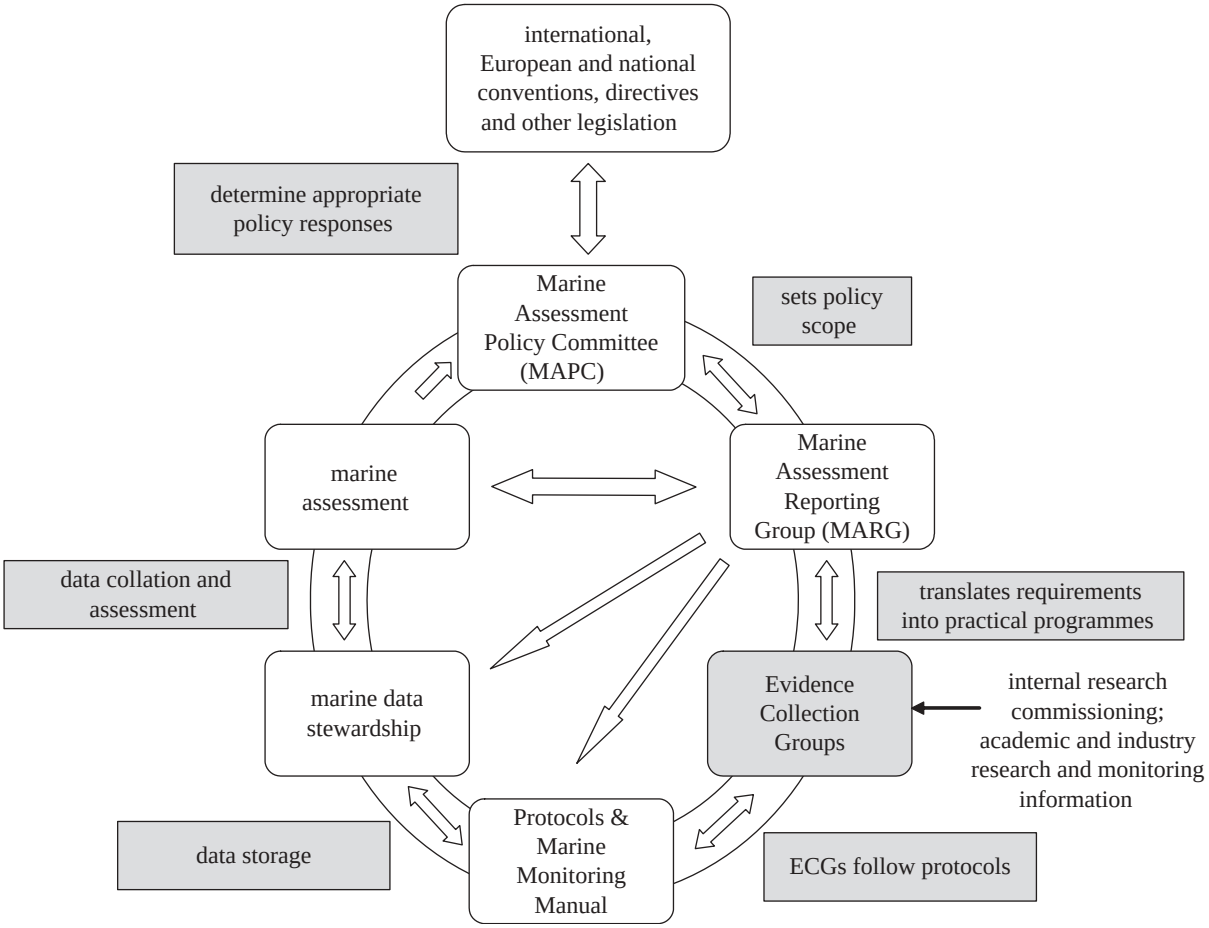

Figure 1. The science–policy interface for the development of marine indicators in the UK. Adapted with permission from MARG (2006).

identifies how assessments will be carried out to meet policy needs; directing programmes, reviewing outcomes, quality assessing protocols for data collection and storage and suggesting changes to monitoring programmes. Several Evidence Collection Groups (ECGs) report to MARG: these are drawn from academic science and marine monitoring organizations involved in data collection, analysis and storage. Information is generated from collating the data, and this contributes to the evidence base from which policy decisions are made. Part of their terms of reference includes being informed by emerging research and technology, promoting awareness of new research and development and identifying new requirements (see TOR3 at http://www.defra.gov.uk/environment/water/marine/uk/science/pdf/csseg-tier3tor.pdf). Although marine policymakers will use their informal networks to commission ad hoc pieces of research to answer pressing policy questions as they arise, these are more likely to be based on secondary analysis and interdisciplinary synthesis than on new primary data. In terms of a more formal system for feeding 'new' science into policy, the ECGs are the route via which science can press to refine existing indicators or develop new ones.

Given that the goals for the marine environment have been clearly specified, the problem of indicator development must have a degree of structure to it. An existing, though still draft, OSPAR indicator relates to the presence of plastic pellets in the stomachs of fulmars as a proxy indicator for the presence of plastic litter in the marine environment (Ryan *et al.* 2009; Noordzeeloket 2007). But what if—as is often the case in policy—a new piece of evidence emerges about the long-term effects of plastic litter (e.g. Thompson *et al.* 2009*b*; Barnes *et al.* 2009) or leached chemicals from plastic debris on other organisms in the marine environment (e.g. Teuten *et al.* 2009)? Whether this should be added into the list of indicators that policymakers use to assess the state of the seas is a moderately structured problem because it can be answered using research and expert advice to answer the questions: should the indicator relate to presence or harm? To which species? Why is that particular organism suitable as an indicator species for that particular ecosystem? Our new piece of science about the long-term effects of plastic litter on the marine environment will be picked up by the ECGs, but this does not guarantee that it will emerge as an indicator. Instead, it will feed into discussions about the full range of indicators needed to inform marine policy, how cost-effective it is to monitor it, how the data will be stored and how it will be weighed up against all the other indicators being considered. In addition, it is not only academic science that can bring evidence to these groups. Initiatives such as Fishing for Litter (www.kimointernational.org) may be examined to see whether this is an effective way to inform judgements about the extent and types of plastic debris in the marine environment, though as discussed by Ryan *et al.* (2009), these opportunistic techniques suffer

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from sampling problems and the resulting evidence needs to be treated with caution.

The ECGs may be the point at which science can apply pressure, but they do not have a direct influence on policy. Instead, the entire UKMMAS needs to be seen as the science–policy interface, maintaining the relationship via forums at different levels. The ECGs are made up of those responsible for the technical aspects of monitoring programmes. Their representatives feed into MARG, which is made up of people in both policy and technical roles. These people in turn feed into MAPC, which is composed of senior policy leads with budgetary control. However, in contrast to the next case study, the feedback loop is fairly closed, reflecting the moderately structured nature of the problem of developing indicators.

(b) *Case study 2: making policy for land-based plastics waste: the move to a life-cycle approach*

Policymaking around land-based plastics has shifted direction over the past decade. From seeing waste as an 'end of pipe' problem to be dealt with once the waste has been created, policymaking has moved to an approach that considers how to promote sustainable consumption and production (SCP) of products and materials (Defra 2005*b*). This integrated product policy approach to plastic-containing products (http://ec.europa.eu/environment/ipp/home.htm) places more weight on waste prevention, recycling and resource efficiency; focusing on priority materials, products and sectors under the overarching goal of a low-carbon economy (Defra 2007*a*,*b*). This involves measures to change the way products and services are designed, produced, used and disposed of—and how any waste is subsequently prepared for reuse, recycled, subjected to energy recovery or otherwise treated prior to landfill.

Policy processes relating to plastics are driven by several European Directives such as REACH (case study 1), the 2002 European Directive on Waste Electrical and Electronic Equipment (WEEE: http://ec.europa.eu/environment/waste/weee/index_en.htm), the 2004 Packaging Directive 94/62/EC on packaging waste (http://europa.eu/scadplus/leg/en/lvb/l21207.htm) and by international agreements such as the Basel Convention on the transhipment of waste (http://www.basel.int/). Plastics policies also need to contribute to innovation and economic performance (Defra 2008*a*), informed by research into improved resource efficiency and the evidence emerging from social science research on understanding and influencing pro-environmental behaviour (http://www.defra.gov.uk/evidence/social/behaviour/index.htm). The list of policy's interest groups contains plastics scientists from a variety of different disciplines in industry and academia, product designers and manufacturers, the British Plastics Federation, the Polymer Innovation Network, retailers of plastic-containing products, users of plastic packaging such as supermarkets, waste management companies, environmental and human health interest groups and local authorities who are responsible for waste collection. It also contains the policymakers dealing with plastic litter in the marine environment in case study 1. Some of the technical issues may have a degree of problem structure to them, such as how to reduce emissions from energy from waste plants, monitoring leaching from landfill (see Teuten *et al.* 2009; Oehlmann *et al.* 2009) or how to use the presence of plastic debris as an indicator of the health of the marine environment. However, the introduction of new technologies such as bioplastics, combined with emerging evidence on human health and environmental effects of conventional plastics, means that the range of interest groups in this debate is wide and constantly changing. Given their competing values, interests and priorities, the overall policy problem of plastics waste must, therefore, be described as unstructured. Realizing this, the team that deals with evidence for land-based plastics policy has developed a far more open structure than that dealing with marine indicators. The next few paragraphs outline its history and approach.

Sustainable Consumption and Production was elevated to a strategic priority for Defra in 2005. This instigated the creation of a small group to deal specifically with SCP policymakers' need to understand what evidence would inform a life-cycle approach and pull together diverse strands of existing evidence. Early work by this evidence base team supported the idea that SCP policy should focus on a products and materials approach—resulting in the formation in early 2008 of the Sustainable Products and Materials Programme. The evidence teams that had previously worked separately on the evidence for SCP and Waste issues were brought together to support this approach (Shaxson 2009).

Previous work elsewhere in the department described the context within which this new team would operate. Drawing on a new understanding of what we mean by robustness in the evidence base for policy (Shaxson 2005; Defra 2008), Defra set out the four functions of evidence-based policymaking: scoping, assembling, procuring and interpreting evidence for policy (Defra 2006). These are shown in figure 2.

A report by Ashridge Centre for Business and Society (ACBAS 2006) encouraged the team to take a more relational approach to the provision of evidence, focusing on building multi-stakeholder networks rather than relying solely on expert academic advice. Key aspects of the team's approach are described in Shaxson (2009): how these are linked together within a revised governance structure is currently being actively addressed.

(i) Working closely with policy colleagues to scope the questions, rather than relying on expert-driven research programmes (Defra 2007*b*; Wilson *et al.* 2007). To do this, the evidence team needed to develop new knowledge management techniques including a suite of workshop-based tools, which encouraged policymakers to engage stakeholders in developing and managing the evidence base (Bielak *et al.* 2008).

(ii) Restructuring research reports to ensure that policy-relevant issues are clear and accessible to non-specialist policymakers; separating these from the technical information and detailed analysis required for peer review and quality assurance. The

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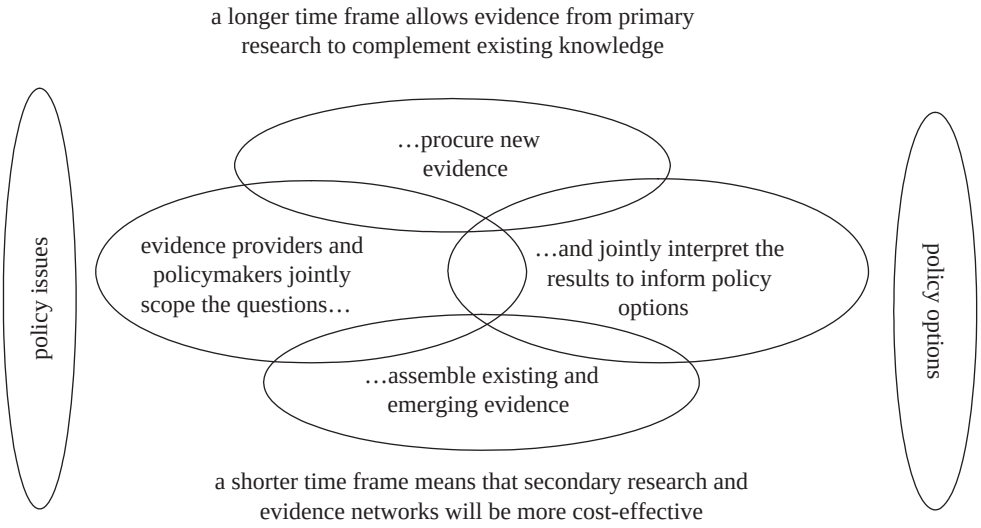

Figure 2. The four functions of evidence-based policymaking. Adapted with permission from Defra (2006).

'enlightenment function' described by Weiss (1977) will not happen to any great extent if reports are not read. The team is also helping produce even shorter (one-page) analyses of the policy implications from the reports. These do not repeat researchers' recommendations, but represent the best available collective knowledge about what a particular piece of research means for a particular aspect of policy at a particular time. Revisiting previously completed research to synthesize and understand its relevance in a changed policy environment is an important part of the team's work (http://www.sd-research.org.uk/post.php?p=816).

(iii) An approach to 'sensing' emerging evidence by taking a more distributed approach to the provision of expert advice. Instead of relying solely on formally constituted programme-level expert advisory groups, the team is experimenting with two new approaches. First, internal research managers and technical experts present informally to university researchers and interdisciplinary research networks, explaining current policy goals and listening to what is emerging from academia. This information may not yet be peer-reviewed, but it stimulates policy to think further into the future and broadens its horizons. Second, the strategic overview normally offered by a formal advisory body will instead be done by bringing project steering groups and other advisors together on an infrequent but regular basis, ensuring that the entire SCP&W agenda is informed by a representational rather than a more traditional task-based approach.

(iv) Explicitly taking a knowledge-brokering approach to managing the SCP and Waste evidence bases; creating and managing multi-stakeholder networks, developing cost-effective tools to improve knowledge management in a resource-constrained policy environment and looking across evidence bases within Defra and elsewhere. Instead of relying solely on external organizations, the team felt that driving the knowledge-brokering approach from within Defra could help improve policy's receptivity to new and emerging evidence.

The team's structure is described in figure 3.

5. CONCLUSIONS AND SUGGESTIONS

> Science Departments must engage with diverse audiences . . . in ways tailored for each audience. This means paying greater attention to the changing contexts in which information is received and used, and consequently the mechanisms required to produce and transfer scientific information. For policy audiences in particular, the relevance of the science to the issues of the day, and the crucial importance of timing, underline the need for interactive knowledge brokering approaches that can deliver synergistic combinations of 'science push' and 'policy pull'.
>
> (Bielak *et al.* 2008, p. 201)

What can we conclude for policymaking around plastics, the environment and human health? My conclusions all stem from the proposition that many plastics-related policies fall into the category of unstructured or badly structured problems. Individual components of a particular policy may be researchable (moderately structured) or relate to the implementation of best practice (well structured). However, policymakers need to reconcile the economic and social benefits plastics bring to society as well as their potential hazards to human and environmental health (Thompson *et al.* 2009*a*,*b*). Doing this in a debate that is heavily value laden and ethically charged means that policy cannot rely on scientific research alone to provide unambiguous answers. Instead, plastics policymaking demands the pluralist, bargaining and incrementalist approaches applicable in the unstructured and badly structured domains and mentioned at the outset of this paper.

My first conclusion is that policymaking around plastics demands a knowledge-brokering approach and that it is possible to change the way evidence teams operate

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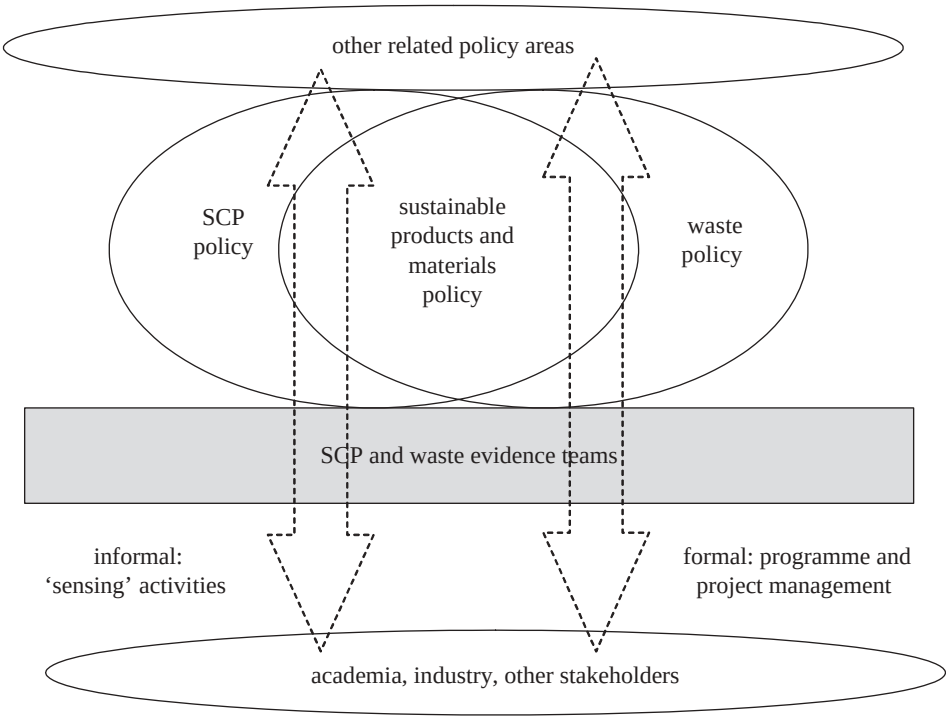

Figure 3. Knowledge brokering by the SCP and Waste evidence team.

within government to become more responsive to the needs of unstructured problems. Table 2 showed how different problem structures give rise to a diverse set of relationships between science and policy. The case studies build on this to illustrate the adage that form follows function: the structure of the brokering process must reflect the structure of the problem. A structured policy problem can be addressed by a relatively linear (though certainly two-way) flow of information, but an unstructured problem has multiple interfaces between multiple stakeholders. The teams in case study 2 have moved from being conventional research managers to devising new methods for scoping, assembling, procuring and interpreting evidence for SCP and Waste policy. However, implementing a knowledge-brokering approach within a government department requires organizational structures, knowledge management tools, governance and budgeting arrangements whose characteristics we are only just beginning to understand.

My second conclusion is that science can continue to improve the way it interacts with policy by becoming more involved in the processes of interpreting what a piece of scientific evidence means for current policy discussions. This does not mean losing objectivity: it is about understanding how the evidence is likely to be used in the policy environment, using it to challenge received wisdom in policy circles, to enrich policy's understanding and to help scope opportunities for change. There is a real role for *short* reports aimed directly at a policy audience, and for knowledge brokers sitting within departments to help researchers and policy teams jointly interpret these into even shorter policy-relevant summaries (Shaxson 2009), being clear where disagreements arise and why. There is also a need for scientists to take value judgements on broad issues, using evidence that may not yet have been through peer review, but which is nevertheless potentially useful to policy. The Marine Climate Change Information Partnership's Annual Report Card (http://www.mccip.org.uk/arc/2007/PDF/ARC2007.pdf) is a good example of this. Many marine policymakers have a scientific degree, but are operating in an environment where rapid decisions need to be made. The Annual Report Card, only four pages long, offers judgements based on a combination of peer-reviewed evidence, emerging evidence and expert opinion. It presents these as degrees of confidence in a statement about the current and likely future effects of climate change on different aspects of the marine environment. Based on reviews done by marine scientists from 18 organizations, it highlights important developments and explores issues that may be new to policymakers. It is possible to click through the website version to the peer-reviewed reports for clarification and find out which organization to turn to for a rapid update. However, the heavily summarized nature of the printed card itself is welcomed by marine policymakers who use it as a decision support tool while being well aware that the information it contains will be updated in future.

Third, I believe we need to be more imaginative about where this brokering function can take place. A Theme Issue such as this can perform two roles in the policy process. One it performs almost by default, but the other needs more thought before it will be effective. There is its traditional role—contributing to the ongoing enlightenment of policymakers about the science of plastics, the environment and human health. The science in this Theme Issue will continue to inform policy for years to come as scientists who read these articles serve on departmental expert advisory committees, research policy issues, review

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others' work or teach students who go on to careers in industry, government or academia. Having said that, it may be possible to use such a Theme Issue in a way which is less traditional but which has a more immediate effect on policy. In the early stages of commissioning this Theme Issue, I suggested that it would be useful to turn the launch of the electronic version into a knowledge-brokering event with heavily summarized reports, workshops, seminars and other activities designed to bring authors together with policymakers to interpret the science contained in these pages and create a basis for more coherent policy action. However, as a proposal that meshed research, science communication, knowledge management and consultancy, it sat outside the remit of every funding body which the editors and I approached.

Finally, there is an opportunity to address many of these issues simultaneously by using the science in this issue to help develop an enhanced Road Map for policy around plastics, the environment and human health in the UK. The Milk and Dairy Road Map (Defra 2008*b*), the first of 10 to be published, used life-cycle analysis to build a picture of the environmental effects of liquid milk 'from cradle to grave'. It is an example of good evidence-based policymaking in practice—using the best available knowledge to build credible commitment from a wide range of interest groups, accompanied by an obligation to update the evidence base and an associated budget. A Plastics Road Map would deal with a policy issue that is inherently less structured than that of liquid milk: life-cycle analysis alone would be insufficient to analyse all of the issues. We would need to think hard about the science–policy relationships, which oversee production of a Plastics Road Map, ensuring that we do not rely only on a limited group of experts whose approach presupposes that the problem is structured. On the other hand, we would need to be pragmatic about how to work in the unstructured domain; developing cost-effective techniques to mesh life-cycle analysis with approaches more appropriate to considering health, social and ethical issues. We would need to build on our emerging understanding about how to access the best available knowledge around complex, unstructured issues and broker it into policy. A Plastics Road Map would not attempt to 'close down' discussion about plastics policy using expert advice alone. Instead, it should be seen as the start of a process involving a wide range of stakeholders including the cutting edge of plastics science (see Thompson *et al.* (2009*a*,*b*), and other papers in this Theme Issue). The process would open up a broad debate that would, in the long term, help a robust dialogue emerge about how to reconcile the relative costs and benefits of plastics to society, the economy, the environment and human health.

I am grateful to Shealagh Pope, Richard Thompson, Angela Coulton and three anonymous reviewers for insightful comments on earlier versions of this article. I would also like to acknowledge many other colleagues at Defra, Environment Canada and the Delta Partnership for information, advice and ideas which I have drawn on in this personal viewpoint.

ENDNOTES

[1]'Science' is taken to include the breadth of natural and physical sciences, social sciences, economics and statistics and the arts and humanities. 'Evidence' includes all of these as well as information from public opinion and other sources.

[2]For the past 5 years, I have worked as a consultant to policy teams in several UK Government departments in the area of science policy and strategy. My focus has been on the practical and organizational aspects of implementing an evidence-based approach to policymaking.

[3]A crisis response effectively 'moves' the problem into the well-structured domain: a symbolic response moves the problem into the unstructured domain. For a detailed discussion of this, see Kurtz & Snowden (2003) and Snowden & Boone (2007).

[4]Note that the four roles do *not* map onto the problem typology outlined earlier. The fact that there are four problem types and four brokering roles is purely coincidental.

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Phil. Trans. R. Soc. B (2009) **364**, 2127–2139
doi:10.1098/rstb.2008.0289

Biodegradable and compostable alternatives to conventional plastics

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Packaging waste forms a significant part of municipal solid waste and has caused increasing environmental concerns, resulting in a strengthening of various regulations aimed at reducing the amounts generated. Among other materials, a wide range of oil-based polymers is currently used in packaging applications. These are virtually all non-biodegradable, and some are difficult to recycle or reuse due to being complex composites having varying levels of contamination. Recently, significant progress has been made in the development of biodegradable plastics, largely from renewable natural resources, to produce biodegradable materials with similar functionality to that of oil-based polymers. The expansion in these bio-based materials has several potential benefits for greenhouse gas balances and other environmental impacts over whole life cycles and in the use of renewable, rather than finite resources. It is intended that use of biodegradable materials will contribute to sustainability and reduction in the environmental impact associated with disposal of oil-based polymers.

The diversity of biodegradable materials and their varying properties makes it difficult to make simple, generic assessments such as biodegradable products are all 'good' or petrochemical-based products are all 'bad'. This paper discusses the potential impacts of biodegradable packaging materials and their waste management, particularly via composting. It presents the key issues that inform judgements of the benefits these materials have in relation to conventional, petrochemical-based counterparts. Specific examples are given from new research on biodegradability in simulated 'home' composting systems. It is the view of the authors that biodegradable packaging materials are most suitable for single-use disposable applications where the post-consumer waste can be locally composted.

Keywords: biodegradable; compostable; biopolymers; packaging; environment; waste management

1. INTRODUCTION

Many different materials are used for packaging including metals, glass, wood, paper or pulp, plastics or combinations of more than one material as composites. Most of these enter municipal waste streams at the end of their service life. Over 67 million tonnes of packaging waste is generated annually in the EU, comprising about one-third of all municipal solid waste (MSW) (Klingbeil 2000). Plastics contribute 18 per cent of the 10.4 million tonnes of packaging wastes produced annually in the UK (DEFRA 2007). Discarded packaging is also a very obvious source of litter, posing a major waste management challenge (see Barnes *et al.* 2009; Gregory 2009; Oehlmann *et al.* 2009; Ryan *et al.* 2009; Teuten *et al.* 2009; Thompson *et al.* 2009*a*,*b*).

In recent years, the recycling of packaging materials has increased but the recycling rates for most plastic packaging remain low (Davis & Song 2006; Hopewell *et al.* 2009). A large number of different types of polymers, each of which may contain different processing additives such as fillers, colourants and plasticizers, are used for packaging applications (Andrady & Neal 2009; Thompson *et al.* 2009*a*). These composition complexities together with contamination during use often render recycling uneconomic compared with disposal in landfill. Although the proportion of waste being landfilled has fallen in recent years, around 60 per cent of municipal waste in England still ends up in landfill (http://www.defra.gov.uk/environment/statistics/wastats/bulletin07.htm). This presents environmental concerns, resulting in strengthening of regulations on waste (e.g. Packaging and Packaging Waste Directive (94/62/EEC) and UK Packaging Regulations (1998).

Biodegradable plastics with functionalities and processabilities (Bioplastics 07/08) comparable to traditional petrochemical-based plastic have been developed for packaging applications (e.g. www.europeanbioplastics.org). Typically, these are made from renewable raw materials such as starch or cellulose. Interest in biodegradable plastic packaging arises primarily from their use of renewable raw materials (crops instead of crude oil) and end-of-life waste management by composting or anaerobic digestion to reduce landfilling (Murphy & Bartle 2004). The disposal of packaging materials is particularly significant in view of the recent focus on waste generation

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One contribution of 15 to a Theme Issue 'Plastics, the environment and human health'.

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and management as important environmental aspects of present-day society (DEFRA 2004; Thompson *et al.* 2009*b*).

In addition to performance and price, biodegradable plastics must offer advantages for waste management systems in order to realize an overall benefit. This paper discusses the potential impact of biodegradable plastics, with particular reference to packaging, and waste management via landfill, incineration, recycling/reuse and composting. It provides an overview of the key life cycle issues that inform judgements of the benefits that such materials have relative to conventional, petrochemical-based counterparts. Specific examples are given from new research on biodegradability in simulated 'home' composting systems.

2. BIODEGRADABLE ALTERNATIVES TO CONVENTIONAL PLASTICS

Biodegradable polymers (*BDPs*) *or biodegradable plastics* refer to polymeric materials that are 'capable of undergoing decomposition into carbon dioxide, methane, water, inorganic compounds, or biomass in which the predominant mechanism is the enzymatic action of microorganisms, that can be measured by standardized tests, in a specified period of time, reflecting available disposal condition' (ASTM standard D6813). A subset of BDPs may also be *compostable* with specific reference to their biodegradation in a compost system, and these must demonstrate that they are 'capable of undergoing biological decomposition in a compost site as part of an available program, such that the plastic is not visually distinguishable and breaks down to carbon dioxide, water, inorganic compounds and biomass, at a rate consistent with known compostable materials (e.g. cellulose)' (ASTM standard D996, also see D6400). Initial steps may involve abiotic (thermal, photo) and biotic processes to degrade the polymer, under suitable conditions, to a low-molecular weight species. However, the resultant breakdown fragments must be completely used by the micro-organisms; otherwise there is the potential for environmental and health consequences (Narayan 2006*a*,*b*). The products of an industrial composting process (typically 12 weeks with an elevated temperature phase over 50°C) must meet quality criteria such as heavy metal (regulated) content, ecotoxicity and lack of obvious distinguishable polymer residues.

Depending on their origins, BDPs may be classified as being either *bio-based* or *petrochemical-based*. The former are mostly biodegradable by nature and produced from natural origins (plants, animals or micro-organisms) such as polysaccharides (e.g. starch, cellulose, lignin and chitin), proteins (e.g. gelatine, casein, wheat gluten, silk and wool) and lipids (e.g. plant oils and animal fats). Natural rubber as well as certain polyesters either produced by micro-organism/plant (e.g. polyhydroxyalkanoates and poly-3-hydroxybutyrate) or synthesized from bio-derived monomers (e.g. polylactic acid (PLA)) fall into this category. Petrochemical-based BDPs such as aliphatic polyesters (e.g. polyglycolic acid, polybutylene succinate and polycaprolactone (PCL)), aromatic copolyesters (e.g. polybutylene succinate terephthalate) and poly(vinyl alcohol) are produced by synthesis from monomers derived from petrochemical refining, which possess certain degrees of inherent biodegradability (Clarival & Halleux 2005 in Smith 2005). This classification differentiates between renewable (bio-based) and non-renewable (petrochemical-based) resources, but it should be noted that many commercial BDP formulations combine materials from both classes to reduce cost and/or enhance performance.

Biodegradable plastics, therefore, often comprise polymer blends that contain partly biogenic (renewable) carbon derived from biomass and partly petrochemical carbon. The per cent biogenic carbon present in a plastic or polymeric product can be readily calculated from the C-14 signature of the product as shown in figure 1 (Narayan 2006*a*,*b*). The carbon dioxide (CO_2) in the atmosphere is in equilibrium with radioactive $^{14}CO_2$. Radioactive carbon is formed in the upper atmosphere through the effect of cosmic ray neutrons on ^{14}N. It is rapidly oxidized to radioactive $^{14}CO_2$, and enters the Earth's plant and animal life through photosynthesis and the food chain. Plants and animals that use carbon in biological food chains take up ^{14}C during their lifetimes. They exist in equilibrium with the ^{14}C concentration of the atmosphere, that is the numbers of C-14 atoms and non-radioactive carbon atoms stay approximately the same over time. As soon as a plant or animal dies, they cease the metabolic function of carbon uptake; there is no replenishment of radioactive carbon, only decay. Since the half-life of carbon is around 5730 years, the petrochemical feedstocks formed over millions of years will have no ^{14}C signature. The quantity of bio-based content can be determined (ASTM standard D-6866) by combusting the test material in a polymer in the presence of oxygen and analysing the CO_2 gas evolved to provide a measure of its $^{14}C/^{12}C$ content relative to the modern carbon-based oxalic acid radiocarbon standard reference material (SRM) 4990c (referred to as HOxII).

After an early pilot plant phase in the 1990s, subsequent upscaling of biodegradable (bio)plastic production by both small specialized and established companies since 2000 has now reached an industrial scale, and significant proportions of established and emerging biodegradable plastics now have renewable rather than petrochemical origins (www.european-bioplastics.org; www.bioplastics24.com). Details on the chemical compositions, production, processing, structure and properties of a wide range of bioplastics used for packaging can be found elsewhere in the literature (e.g. Smith 2005) (paper-based products are traditionally regarded as a separate material group). Current production capacity for biodegradable plastics worldwide is around 350 000 tonnes (Bioplastics 07/08), representing less than 0.2 per cent of petrochemical-based plastic, at approximately 260 million tonnes (Miller 2005). However, the environmental performance benefits are insufficient on their own to enable bioplastic polymers to be more widely used as alternatives to conventional plastics. They also need to be cost-effective, fit for purpose and, ideally, provide unique benefits in use (Miller 2005). Hence, bioplastic polymers have not yet realized their full potential.

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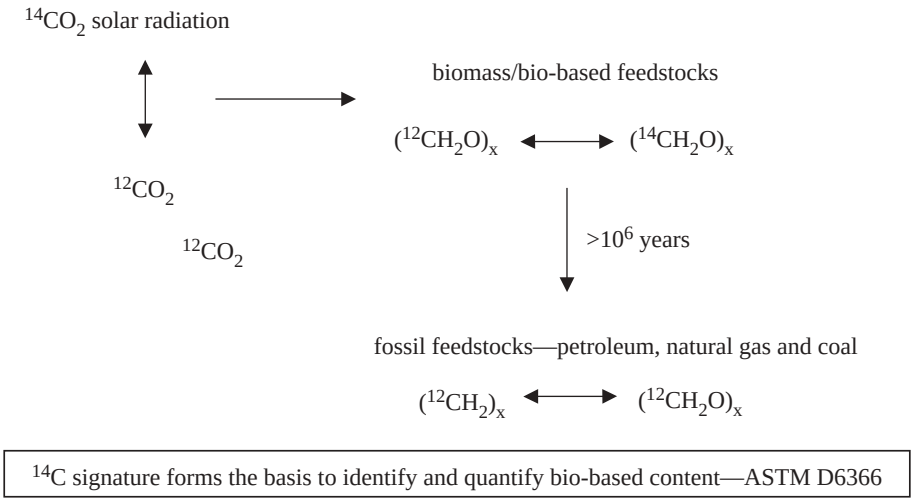

Figure 1. Carbon-14 signature of bio- and petrochemical polymers.

The costs of bioplastic polymers are generally still much higher than that of their traditional plastic counterparts (Petersen *et al.* 1999). Most fall in the range 2–5 € kg^{-1} (Bioplastics 07/08) (compared with approx. 1.2 € kg^{-1} for major petrochemical polymers) and this is a major restriction for more widespread use. However, significant growth rates have been achieved in product capacity over the last decade or so. Bioplastic polymers are expected to become priced more as commodity materials when a critical mass is achieved, driven by a combination of forces including performance and cost improvements, benefits assigned to the use of renewable (bio)resources, increasing oil prices and increasing awareness of environmental impacts and associated legislation.

Processing parameters and technical characteristics of a wide range of commercial bioplastic polymers have been reviewed recently (Bioplastics 07/08). Many bioplastics now have mechanical properties equivalent to that of their conventional counterparts (e.g. polypropylene (PP), polystyrene and polyethylene (PE)) and can be processed using technologies widely used in the polymer industry (e.g. compounding, film processing and moulding). They have found use in many short service life applications where biodegradability is a key advantageous feature (www.european-bioplastics.org) including consumer packaging (e.g. trays, pots, films and bottles in food packaging), convenience food disposables (e.g. cutlery/tableware), bags (shopping, garden or domestic waste), agriculture mulch films, personal-care disposals (e.g. nappies) and even golf tees. Bioplastic polymers have also been used in more durable applications such as in textiles, consumer goods, automotive parts and building and construction where the focus is on the use of renewable (bio)resources and any inherent biodegradability properties need to be suppressed or controlled by careful design.

Bio-based versus biodegradable: it is important to recognize that not all bio-based polymer materials are biodegradable and vice versa. Equally, it is important to recognize that attributes like biodegradability of a given polymer need to be effectively coupled with appropriate waste management in order to capture maximum environmental benefit. For durable products where biodegradability is not a required element for reasons of performance, safety and product life, alternative methods of disposal like waste to energy or recycling need to be identified. Examples of such durable bio-based polymers are bio-polyurethanes based on polyols from vegetable oils for automotive and farm vehicles (Narayan 2006*a*,*b*), biofibre composites for industrial and automotive applications and recent developments in bio-polyethylene derived from sugar cane via ethanol to ethylene.

3. WASTE MANAGEMENT OPTIONS FOR BDPs

There are many technologies available for the treatment of conventional plastic packaging waste (Tukker 2002) from household waste including: integrated collection and incineration with energy recovery, selective combustion of plastics with high calorific value (e.g. in cement kilns) and use as a reducing agent in blast furnaces or as feedstock for recycling.

Approximately 1 million tonnes of non-bottle domestic mixed plastic packaging waste arise in the UK each year, and this is estimated to increase between 2 and 5 per cent per annum (WRAP 2006, 2008). A 'Waste Hierarchy' proposed by the UK government (DEFRA 2007) as guidance for selecting the options to minimize the impact of waste recognizes reduction and reuse as the most favourable options where the aim is to minimize the material consumption or divert materials from waste streams.

The impacts of biodegradable bioplastics, when entering the waste stream and handled by current available options (recycling, incineration and landfill), are assessed briefly below. As BDPs enable a potential option for waste treatment through composting as a way to recover the materials and to produce a useful product as compost, particular attention will be given to composting biopolymers.

(a) *Recycling*

Biodegradable plastics that enter the municipal waste stream may result in some complications for existing plastic recycling systems. For example, the addition

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of starch or natural fibres to traditional polymers can complicate recycling processes (Scott 1995; Hartmann & Rolim 2002). Although it is feasible to mechanically recycle some bioplastic polymers such as PLA a few times without significant reduction in properties (Claesen 2005), the lack of continuous and reliable supply of bioplastic polymer waste in large quantity presently makes recycling less economically attractive than for conventional plastics. Finally, for certain applications such as food packaging (e.g. in modified atmosphere packaging of meat products), multilayer lamination of different biopolymers may be necessary to enhance barrier properties, just as in conventional plastics (Miller 2005), and this will compromise recyclability of the scrap during packaging manufacture and of post-consumer waste. The recycling of plastics is considered in more detail elsewhere in this volume (Hopewell *et al.* 2009).

(b) *Incineration with energy recovery*

Most commodity plastics have gross calorific values (GCV) comparable to or higher than that of coal (Davis & Song 2006). Incineration with energy recovery is thus a potentially good option after all recyclable elements have been removed. It is argued that petrochemical carbon, which has already had one high-value use, when used again as a fuel in incineration represents a more eco-efficient option than burning the oil directly (Miller 2005).

Reports by the Environment Committees of the UK Parliament (House of Commons 1993; House of Lords 1994) have supported the view that energy recovery for some types of household plastic wastes is an acceptable waste management option. Trials conducted by the British Plastics Federation demonstrated that modern waste-to-energy plants were capable of burning plastic waste, even those containing chlorinated compounds such as PVC without releasing dangerous or potentially dangerous emissions of dioxins and furans (BPF 1993). In 2005/2006, around 8 per cent (approx. 3 million tonnes) of UK municipal waste was processed through 15 incineration facilities (www.defra.gov.uk/environment/statistics/waste) and over 40 million tonnes were incinerated within the EU in around 230 incineration facilities (Musdalslien & Sandberg 2002). It is envisaged that incineration will face continued resistance in the UK unless the public is convinced about the safety of incineration and its contribution to renewable energy supplies (Miller 2005).

Energy recovery by incineration is regarded as a suitable option for all bioplastic polymers and renewable (bio)resources in bioplastic polymer products are considered to contribute *renewable energy* when incinerated (www.european-bioplastics.org). Natural cellulose fibre and starch have relatively lower GCV than coal but are similar to wood and thus still have considerable value for incineration (Davis & Song 2006). In addition, the production of fibre and starch materials consumes significantly less energy in the first place (Patel *et al.* 2003), and thus contributes positively to the overall energy balance in the life cycle. At present, the lack of scientific data on GCV of bioplastic polymers (e.g. relative importance of moisture content (MC), etc.) makes it difficult to accurately determine their value for energy recovery by incineration—further research in the area is required.

(c) *Landfill*

Landfill of waste plastics is the least favoured option in the UK waste hierarchy. It was attractive historically as it was extremely simple and cheap without necessary separation, cleaning or treatment. Western Europe sent 65 per cent of the total recoverable plastics in household waste (8.4 million tonnes annually) to landfill in 1999 (APME 2002). However, suitable sites for landfill across Europe are running out and public concerns are increasing about the impact of landfill on the environment and health from the amount of toxic materials in land-filled municipal waste and their potential leaching out of landfill sites (Miller 2005). Reducing the quantities of waste that ultimately ends up in landfill has become explicit government policy (e.g. Landfill Directive European Commission 1999/31/EC) in the UK and represents a particularly difficult task to achieve (e.g. approx. 60% municipal waste in England is still landfilled in comparison with approx. 37% in France and approx. 20% in Germany (EEA 2007)).

The landfill of biodegradable materials including bioplastic polymers, garden and kitchen waste presents a particular problem in that methane, a greenhouse gas with 25 times the effect of CO_2, may be produced under anaerobic conditions (Hudgins 1999). While such a 'landfill gas' can and is captured and used as an energy source, The Landfill Directive (99/31/EC) seeks to reduce the total amount of biodegradable municipal waste (BMW) going to landfill in three successive stages eventually to 35 per cent of the 1995 total of BMW by 2020.

(d) *Biological waste treatments: composting or anaerobic digestion*

Unlike conventional petrochemical-based polymers, biodegradable and compostable bioplastic polymers can be composted. This can be via aerobic waste management systems such as composting to generate carbon- and nutrient-rich compost for addition to soil. In the UK, there are now more than 300 composting sites that collectively compost about 2 million tonnes of waste annually (roughly 75% of which is household waste, 5% municipal non-household waste and 20% commercial waste: http://www.organics-recycling.org.uk/). The aerobic biodegradation systems are thus of primary importance for BDPs and are dealt with in detail in the following section of this paper.

Certain BDPs are also suitable for anaerobic digestors whereby biowastes can be converted to methane, which can be used to drive generators for energy production. Published reports on the anaerobic digestibility of biodegradable bioplastics are relatively scarce and these systems are not discussed further here (for further information see Ramsay *et al.* 1993; Mohee *et al.* 2008).

4. BIODEGRADABILITY AND COMPOSTABILITY

Making or calling a product biodegradable has no inherent value if the product, after use by the customer, does not end up in a waste management system

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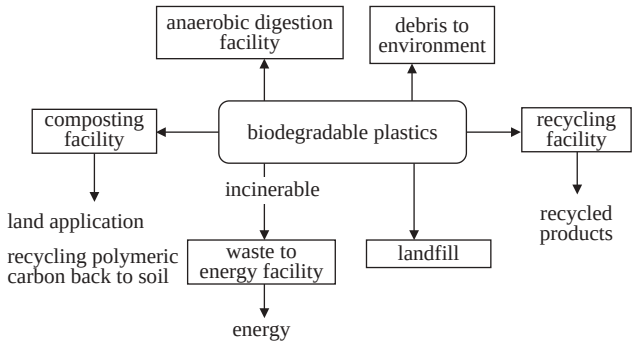

Figure 2. Integration of biodegradable plastics with disposal infrastructures.

that uses the biodegradability features (Narayan 1993, 1994). Figure 2 illustrates the integration of biodegradable plastics with disposal infrastructures that use this biodegradable function of the plastic product.

(a) *Principles and concepts of composting*

Composting has the potential to transfer biodegradable waste, including biodegradable plastics, into useful soil amendment products. Composting is the accelerated degradation of heterogeneous organic matter by a mixed microbial population in a moist, warm, aerobic environment under controlled conditions. Biodegradation of such natural materials will produce valuable compost as the major product along with water and CO_2. The CO_2 produced does not contribute to an increase in greenhouse gases because it is already part of the biological carbon cycle. Composting is also an important disposal infrastructure because it can receive other bio-based wastes in addition to biodegradable plastics—for example, more than 50 per cent of the MSW stream is typically garden and food waste and non-recyclable paper products.

(i) *Degradable versus biodegradable*

A number of polymers in the market place are designed to be degradable, i.e. they fragment into smaller pieces and may even degrade to residues invisible to the naked eye. While it is assumed that the breakdown products will eventually biodegrade, there are no data to document complete biodegradability within a reasonably short time period (e.g. a single growing season per year). Hence hydrophobic, high surface area plastic residues may migrate into water and other compartments of the ecosystem. In a recent science article, Thompson *et al.* (2004) reported that plastic debris around the globe can erode (degrade) away and end up as microscopic granular- or fibre-like fragments, and that these fragments have been steadily accumulating in the oceans. Their experiments show that marine animals consume microscopic bits of plastic, as seen in the digestive tract of an amphipod. The Algalita Marine Research Foundation (see www.algalita.org/pelagic_plastic.html) reports that degraded plastic residues can attract and hold hydrophobic elements like polychlorinated biphenyls (PCB) and dichlorodiphenyltrichloroethane (DDT) up to 1 million times background levels. The PCBs and DDTs are at background levels in soil, and diluted out, so as to not pose significant risk. However, degradable plastic residues with these high surface areas concentrate these chemicals, resulting in a toxic legacy in a form that may pose risks in the environment. Japanese researchers (Mato *et al.* 2001) have similarly reported that PCBs, DDE and nonylphenols (NP) can be detected in high concentrations in degraded PP resin pellets collected from four Japanese coasts. This work indicates that plastic residues may act as a transport medium for toxic chemicals in the marine environment (see discussion in Teuten *et al.* 2009).

Therefore, designing hydrophobic polyolefin plastics like PE to be degradable, without ensuring that the degraded fragments are completely assimilated by the microbial populations in the disposal infrastructure in a short time period, has the potential to harm the environment more than if it was not made degradable. Heat, moisture, sunlight and/or enzymes can shorten and weaken polymer chains, resulting in fragmentation of the plastic and some cross-linking, creating more intractable persistent residues. It is possible to accelerate the breakdown of the plastics in a controlled fashion to generate these fragments, some of which could be microscopic and invisible to the naked eye, and some elegant chemistry has been done to make this happen as reported in the literature (Scott & Wiles 2001). However, this degradation/fragmentation is not biodegradation *per se* and these degraded, hydrophobic polymer fragments pose potential risks in the environment unless they are completely assimilated by the microbial populations present in the disposal system in a relatively short period.

(ii) *Measurement of biodegradability*

Micro-organisms use the carbon substrates to extract chemical energy that drives their life processes by aerobic oxidation of glucose and other readily usable C-substrates (Narayan 1994):

$$\text{C-substrate} + 6O_2 \rightarrow 6CO_2 + 6H_2O,$$
$$\Delta G_0 = -686\ \text{kcal/mol} \qquad (CH_2O)_x; \quad x = 6$$

Thus, a measure of the rate and amount of CO_2 evolved in the process is a direct measure of the amount and rate of microbial use (biodegradation) of the C-polymer. This forms the basis for various international standards for measuring biodegradability or microbial use of the test polymer/plastics. The rate and extent of biodegradation or microbial use of a test plastic material can be measured by using it as the sole added carbon source in a test system containing a microbially rich matrix-like compost in the presence of air, and under optimal temperature conditions (preferably at 58°C—representing the thermophilic phase). Figure 3 shows typical data obtained when the per cent carbon released (as CO_2) from a bioplastic exposed in a composting environment is plotted as a function of time. First, a lag phase occurs during which the microbial population adapts to the available test C-substrate. Then follows the biodegradation phase during which the adapted microbial population begins to use the carbon substrate for its cellular life processes, as measured by the conversion of the carbon in the test material to CO_2. Finally, the output reaches a plateau when use of the substrate is largely complete.

Based on the above concepts, the ASTM committee D20.96 on Biobased and Environmentally Degradable

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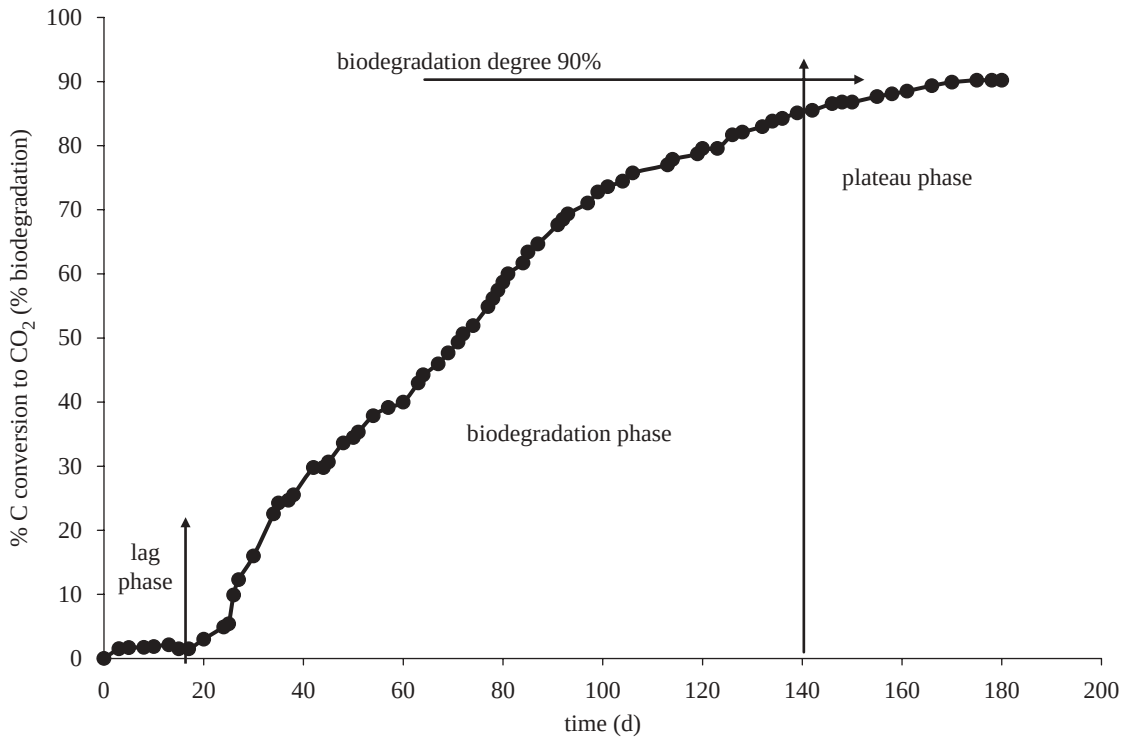

Figure 3. Example data from a biodegradation test of a biodegradable biopolymer assessed as CO_2 release over 180 days. CO_2 release curve shows typical lag phase, biodegradation phase and plateau phase.

Plastics (www.astm.org) developed a Specification Standard D6400 (see also D6868) for products claiming to be biodegradable under composting conditions or compostable plastics (ASTM, 2002). The above specification standard is in harmony with standards in Europe, Japan, Korea, China and Taiwan. EN13432 'Requirements for Packaging Recoverable through Composting and Biodegradation—Test Scheme and Evaluation Criteria for the Final Acceptance of Packaging' is the European standard (norm) and similar to D6400. The current UK standard BS EN 13432 (2000) covers the requirements for packaging recoverable through composting and biodegradation and test scheme and evaluation criteria for the final acceptance of packaging. At the international level, the International Standards Organization (ISO) has developed ISO 17088, 'Specification for Compostable Plastics' which is in harmony with these European and US norms.

The fundamental requirements of these worldwide standards for complete biodegradation under composting conditions are:

(i) conversion to CO_2, water and biomass via microbial assimilation of the test polymer material in powder, film or granule form.
(ii) Ninety per cent conversion of the carbon in the test polymer to CO_2. The 90 per cent level set for biodegradation in the test accounts for a ± 10 per cent statistical variability of the experimental measurement; in other words, there is an expectation for demonstration of a virtually complete biodegradation in the composting environment of the test.
(iii) Same rate of biodegradation as natural materials—leaves, paper, grass and food scraps.
(iv) Time—180 days or less (ASTM D6400 also has the requirement that if radiolabelled polymer is used and the radiolabelled evolved CO_2 is measured, then the time can be extended to 365 days).

Two further requirements are also of importance:

(i) Disintegration: <10 per cent of test material mass retained by a 2 mm sieve using test polymer material in the shape and thickness identical to the product's final intended use—see ISO 16929 and ISO 20200.
(ii) Safety: the resultant compost should have no impacts on plants, using OECD Guide 208, Terrestrial Plants, Growth Test or similar, such as PAS 100 (BSI 2002). Furthermore, regulated (heavy) metals content in the polymer material should be less than defined thresholds e.g. 50 per cent of EPA (USA and Canada) prescribed threshold.

(b) *Composting in practice*

The treatment of biodegradable plastics by composting is now considered in many parts of the world to be an appropriate form of material recovery. In the UK, it is a permitted recovery option specified in the Producer Responsibility (Packaging Waste) Regulations as amended in 1997.

In a large-scale study from March 2001, in Kassel, Germany, BDP packaging was introduced into the local retail trade (Klauss 2001). The purpose of this scheme was to introduce biodegradable packaging and manage its source separation by householders so

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that it could be collected with the organic waste stream to produce compost. The scheme required much planning prior to the launch, to ensure that the public had received sufficient information about the BDPs, their labelling, separation and collection. The mixed packaging and organic waste was composted at a full-scale composting site and was undertaken at a commercial level. The compost feedstock was monitored to ensure a relatively low proportion of one plastic to 99 parts organic waste on a weight basis. The compost produced showed no differences in terms of quality parameters compared with conventional compost comprising solely green waste (no BDPs) and had the same positive effects on soil and plant characteristics (Klauss & Bidlingmaier 2004).

Householder surveys indicated that 82 per cent of Kassel's population could clearly identify the logo printed on compostable polymers and 90 per cent supported the replacement of conventional plastic packaging with compostable packaging. The success of this programme has created a demand for further products that can be digested/degraded in the same way as 'conventional' organic waste. The benefits for this are twofold: (i) increased separation and collection efficiency (household or centralized) and (ii) reduced amount of waste to landfill or incineration.

Some legislation, however, imposes a number of constraints on the composting industry. In May 2003, the Animal By-Products Regulation (ABPR) started the UK implementation of an EU Regulation. The ABPR divides animal by-products into three categories and stipulates the means of collection, transport, storage, handling processing and use or disposal for each category: category 1, highest risk materials such as carcasses infected with BSE, scrapie, etc.; category 2, also high-risk materials such as animals that die on farms and animals that are unfit for human consumption; and category 3, materials that are fit (but not intended) for human consumption such as fish, milk, parts of slaughtered animals, etc. Household kitchen waste and, by association, biodegradable food packaging (because it has come into contact with food, meat or non-meat) are classified under Category 3. Categories 2 and 3 materials may be composted or treated via anaerobic digestion following strict requirements on handling, temperature and retention times.

Although the ABPR does not apply to sites accepting only green botanical garden waste, many UK Local Authorities have already started mixed organic waste (garden and kitchen) collections or are considering mixed collections in order to meet legislative targets. For mixed organic waste collections, the majority of the material collected is from botanical sources; however, due to the presence of kitchen/catering waste all the waste must be composted in-vessel in order to meet the requirements. Local Authorities could collect the organic botanical waste separately from the kitchen-derived waste, but this has extensive logistical and cost issues (separate vehicles, crew and composting facility). In-vessel composting is more costly than the open-windrow methods commonly adopted in the UK for pure 'green waste'. This results in increased composting costs per tonne, gate fees charged to Local Authorities and reduction in the competitiveness of in-vessel composting against other treatment and disposal options such as landfill.

Concerns over the potential ecotoxicity of degradation products have resulted in the formulation and adoption of suitable international standards for compostable polymer products. For example, EN 13432 requires that compostable polymer materials have to fulfill European, or where none exist, national requirements for compostability. In December 2003, the Composting Association in the UK launched a Certification Scheme for Compostable Packaging in order to assist UK Local Authorities with the selection of sacks for organic waste collections. As there is currently no European standard on compost quality (besides the ecological criteria for the award of the EU Eco-label), the UK adopted the BSI PAS 100 in November 2002 (BSI 2002). Other standards such as the ASTM D6400 and ISO 17088 also define product classification and requirements for composts.

(c) *Home (domestic) composting*

In the UK, home composting has been identified by the Strategy Unit of the Cabinet Office as one of five key measures to reduce the growth rate of household waste (Anon. 2002; Murphy & Bartle 2004). In addition to kitchen and garden waste, home composting of biodegradable packaging materials could divert waste from municipal collection systems and complement industrial composting. It must be noted that it is difficult to regulate home composting, and anaerobic composting conditions occurring in poorly managed systems will result in the generation of methane. Moreover, home composting using compost bins or heaps is more variable and less optimized than industrial composting and the temperature achieved is rarely more than a few °C above ambient temperature. Under such conditions, certain compostable materials certified for industrial composting (EN13432) may not biodegrade sufficiently. The 'OK Compost Home' standard, which repeats the EN13432 test protocol at ambient temperature, as shown in table 1, has been established by AIB-VINÇOTTE in Brussels (www.aib-vincotte.com). These temperature conditions do not reflect true composting process principles which require them, by definition, to go through a thermophilic phase (55–65°C) that can last from a few days to a couple of months depending on the composting volume. The thermophilic phase of composting is of importance to ensure the destruction of thermosensitive human and plant pathogens, fly larvae and weed seeds. Regulations by the US Environmental Protection Agency specify that to achieve a significant reduction of pathogens during composting, the compost should be maintained at minimum operating conditions of 40°C for 5 days, with temperatures exceeding 55°C for at least 4 h of this period.

Some bioplastic polymers, particularly used as bags and pots for horticulture or waste collection bag applications, have been certified by the OK Compost Home scheme while others passed only 'OK Compost' standard for industrial composting (http://www.aib-vincotte.com/data) and are not suitable for home

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Table 1. Comparison of standards for industrial and home composting.

	industrial composting (EN 13432)	home composting (vincotte certification)
biodegradation	test at 58°C in 180 days biodegradation min. 90%	test at 20–30°C in 365 days biodegradation min. 90%
disintegration	test at 58°C in 90 days sieve 2 mm mesh disintegration >90% max. 10% of dry weight allowed to be retained by 2 mm sieve	test at 20–30°C in 180 days sieve 2 mm mesh disintegration >90% max. 10% of dry weight allowed to be retained by 2 mm sieve
certification	Din Certco/OK Compost	OK Home

composting. This distinction is important and it is vital that clear guidance is communicated to the public who may otherwise assume that any products labelled as 'biodegradable', 'compostable' or 'eco-' under the numerous certification systems can simply be put into their home or garden compost bins. These are unlikely to reach the thermophilic compost temperatures required for both suitable degradation of certain materials and to achieve sanitization.

New research to characterize the extent of biodegradation when a range of biodegradable or potentially biodegradable packaging materials are disposed of in simulated home composting typical of the UK is presented briefly below. The objective was to establish whether potentially biodegradable packaging materials would show appropriate levels of biodegradation when exposed to 'typical' home compost conditions (non-thermophilic) together with green garden waste. Small specimens of 12 bio-based materials (six were from materials used commercially and six were from developmental materials that were designed to be biodegradable—see table 2) were assessed as material weight loss over a 24-week winter/spring period between November and May in the southeast of the UK. Whole food packaging units (trays/plate) made from three of the materials were also assessed under the same conditions but were mixed directly into the compost matrix.

The composting was undertaken outdoors in the home-composter, lidded 'cone' systems (volume 160 l) filled with a 'base mixture' of approximately 60 per cent green herbaceous and grass clippings and 40 per cent chopped 'woody' herbaceous material from the local site that was free of pesticides or herbicides and had previously been composted for 30 days to establish an active microflora/fauna. Twelve packaging materials (approx. 25 × 25 mm sheets) were individually secured into nylon mesh bags and replicate specimens placed into a stainless steel rack for easy retrieval. The sample racks were inserted in the middle of a composting bin between layers of base mixture (approx. 600 mm below the compost mixture surface). Three replicate composter units were established with three replicate specimens of each material removed per composter per sampling time. Additional six composter bins were set up, two of each with 6.4 wt% of one of the three main packaging materials (potato starch trays, PLA trays and paper plate) as whole units mixed in with the green waste base mixture. Two further composter bins containing only the compost base mix and no added biodegradable packaging materials were used as controls for a subsequent seed germination comparison.

The composters were sampled on a monthly basis from November to May for determination of specimen mass loss and MC (od basis), temperature and overall compost volume reduction. Replicate samples of the small test materials or whole units were removed at each sampling interval. 'Turning' of the composts was done only on these occasions.

A bioassay of the composts from the whole packaging unit test and the control compost was also conducted in accordance with the 'Specification for composted materials' (PAS 100; BSI 2002). F1 tomato seeds (variety Shirley, Sutton Seeds, UK) were placed in the prescribed mixture of a peat-based growth medium (PBGM) and test compost (1 : 2 ratio by volume of compost to PBGM base mix) in seed trays and maintained with regular watering at a temperature of 20–25°C in a natural light greenhouse in early summer 2005 over a 28-day period of the test. Seed germination, fresh plant mass, abnormalities and weed emergence were recorded in accordance with PAS 100.

The compost bin systems functioned as a low temperature composting environment between 15°C and 18°C in November at the start of the experiment. The temperature dropped to a low of approximately 8–10°C in January/February/March and then rose again to approximately 14°C in May. The composter bin temperatures were considerably lower than specified (20–30°C) in the OK Compost Home standard (table 1) but reflect the typical seasonal temperatures in the southeast of the UK. All composter bins showed an acceptable level of reduction in biomass volume (approx. 50%) during the composting period. The temperature profiles of the bins and the degradation of their contents were largely consistent across the whole study.

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Table 2. Packaging materials used for simulated home composting. (Fast degrader = mass loss approximately 80% after 90 days; medium degrader = mass loss approximately 40% after 90 days; slow degrader = mass loss <5% after 90 days.)

name	commercial (C)/ experimental (E) material	material	principal components (wt%)	small sample	whole unit	rate of degradation
potato starch	C	potato starch-based tray	potato starch (<75%)	✓	✓	fast
starch laminate	C	starch-based tray with a starch/PCL laminate	starch; starch PCL surface overlay	✓		fast
paper	C	pressed wood pulp plate	wood pulp 70%; starch size 20%; other 10%	✓	✓	medium
silvergrass	C	pressed silvergrass pulp plate	*Miscanthus* spp. pulp	✓		fast
coconut	C	moulded coconut fibre tray	*Cocos nucifera* fibre	✓		medium
recycled paper	C	moulded recycled paper pulp tray	recycled paper	✓		medium
PLA	E	PLA tray	100% PLA	✓	✓	slow
starch/PCL	E	starch/PCL—extrudate sample	100% starch/PCL	✓		slow
PP(A)	E	PP with biodegradability additive A	90% PP; 10% bio-additive A	✓		slow
PP(B)	E	PP with biodegradability additive B	90% PP; 10% bio-additive B	✓		slow
PP(B)+	E	PP with biodegradability additive B plus chalk filler	60% PP; 10% bio-additive B; 30% chalk	✓		slow
PP/starch	E	PP compounded with starch granules	88% PP; 10% starch granules; 2% other	✓		slow

The visual assessment showed that complete disintegration and incorporation of the starch trays into the compost matrix had occurred after 90 days of composting. The paper-plate material was also extensively broken down over the composting period, although it was possible to distinguish elements of the original plate material after 180 days, despite their being heavily discoloured and lacking structural integrity. The PLA polymer showed no visual evidence of microbial breakdown after 180 days, although some fragments had broken off from the trays. This was not considered to be disintegration as a result of biodegradation but was attributed to disturbance of the bins and mechanical damage when retrieving samples.

The mass loss (as an indicator of the biodegradation) data for the full range of material types as small specimens are presented in figure 4 and for the whole units in figure 5. From approximately 90 days exposure, three groups of materials could be clearly distinguished:

(i) The fast degraders (starch-based polymers and the plant fibre-base silvergrass) exhibiting mass losses of approximately 80 per cent.
(ii) The medium degraders (wood fibre-based paper and the coconut fibre) with mass losses of approximately 40 per cent.
(iii) The slow degraders (PLA, PP with additives and starch/PCL) with negligible mass loss <5 per cent.

This differentiation of the three groups was then maintained to the conclusion of the experiment at 180 days (table 2). The fast degraders lost approximately 90 wt% and became visually indistinguishable from their sealed packets; the medium degraders lost approximately 50 wt% and remained recognizable on close inspection. The slow degraders lost typically less than 5 wt% and were clearly recognizable.

The results for MC assessment showed that fast and medium degraders absorbed moisture readily during the composting process, typically ranging from 100 to 300 per cent for the starch and fibre materials over the 30- to 180-day period. The slow degrader group exhibited very low levels of moisture absorption with the starch/PCL, PP/starch and PLA typically below 10 per cent and the PP/modifiers below 1 per cent.

The results of the PAS 100 bioassay (data not shown) showed that composts derived from the composters containing whole packaging units (starch, paper and PLA) and from the controls gave equal or higher seed germination rates and equivalent or better fresh seedling weights compared with the growth medium base alone (an exception was one PLA compost bin that had a 21 per cent reduction in seedling fresh weight). All the amended composts failed the weed criterion of PAS 100, but this is expected because low-temperature composting systems do not achieve sterilization of weed seeds.

This study has shown that biodegradable packaging materials exhibited a wide range of biodegradation properties in this simulated home composting system run under non-thermophilic conditions (a regime where mesophilic micro-organisms dominate). It is clear that this mesophilic home composting condition may be less favourable for biodegradation than those specified in some standards. For instance, the home composting system used in this study operated over a temperature range of approximately 5–18°C rather than the 20–30°C range specified in the OK

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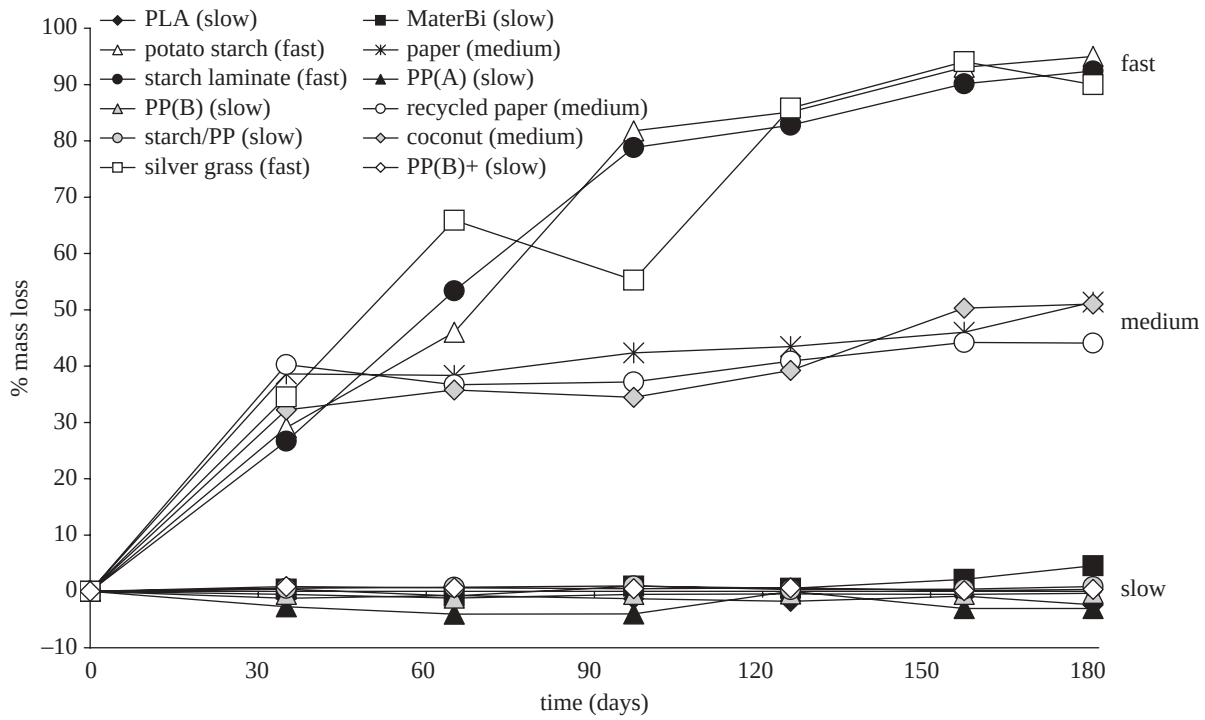

Figure 4. Mass loss (wt%) over time—test materials as small samples (Note: PLA, potato starch and paper are also represented as whole packaging units in figure 5) (error bars not shown for clarity, 95% confidence interval ranged between 1% and 20% mass loss). 'Fast', 'medium' and 'slow' refer to rates of degradation indicated in table 2.

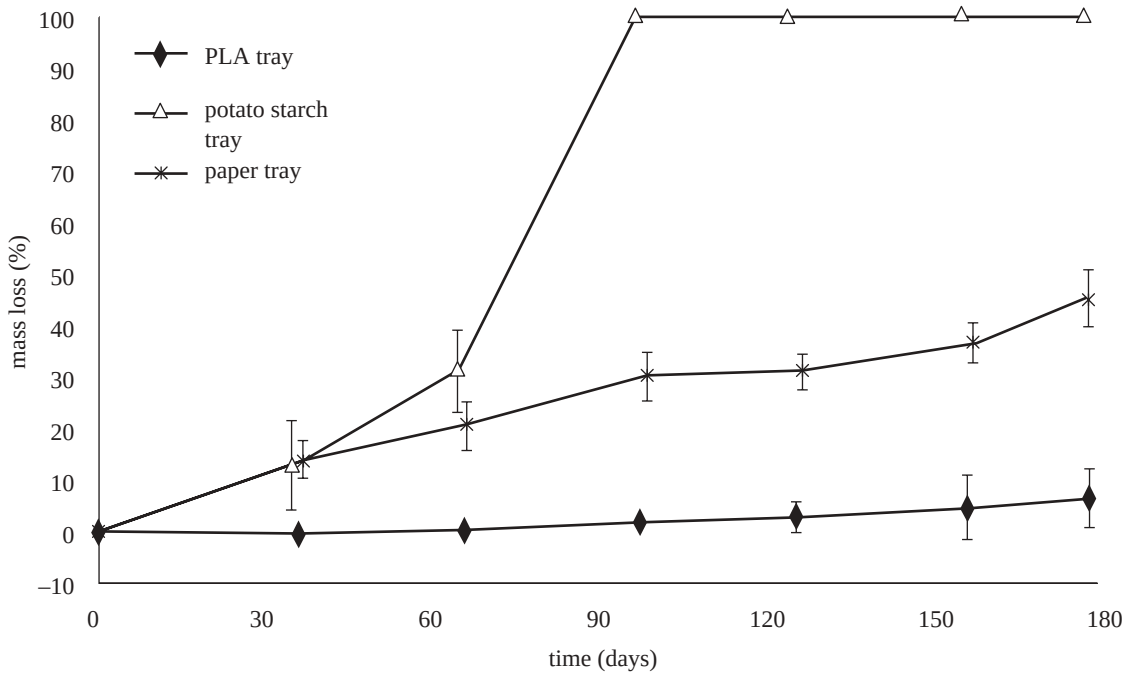

Figure 5. Mass loss (wt%) over time—whole packaging units (Note: PLA tray, potato starch tray and paper plate are also represented as small samples in figure 4) (error bars are 95% confidence intervals).

Compost Home standard. The fast degrader bioplastics, predominantly based on high levels of starch and the grass fibre/starch composite, were readily biodegraded in the home composting system. The medium degraders based on wood or coconut fibres exhibited mass losses of approximately 50 per cent over the composting period. The easily fragmentable nature of the residual material at the end of the 180-day period enabled the medium degraders to be readily incorporated into the compost matrix and we conclude that medium degraders would be acceptable in terms of disintegration. The extent of biodegradation of these materials, however, failed to satisfy the >90 per cent requirement within 180 days of BS EN 13432. How this may change should the test be extended to 360 days (as in the OK Compost Home standard) and whether this can be mitigated (as for cellulose residues in farm compost) remain to be

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studied further. The slow degraders (e.g. combined starch/biodegradable polyester formulation and PLA), including bioplastic polymers certified as compostable under EN 13432 conditions, exhibited either no or very low levels of biodegradation and fragmentation over the composting period. Although greater degradation may be achieved over longer periods (e.g. expansion to 360 days), elevated temperature around 60°C has been shown to be a crucial parameter, enabling the induction of biodegradation of polymers such as PLA (e.g. Agarwal *et al.* 1998; Scott & Wiles 2001; Tokiwa & Jarerat 2004). Such temperatures are clearly lacking in home composting systems of the type modelled. The seed germination study indicated that composts made from green waste incorporating approximately 6 per cent by mass of home composted starch or paper trays give growth media that support good seed germination and seedling development. Although similar results were also achieved with compost incorporating non-biodegraded PLA materials, it must be noted that the compost with PLA trays would fail the disintegration requirements set in the OK Compost Home as the PLA trays remained almost intact. Inhibition of seedling development, in composts with degradable PE and control composts from open-windrow systems, has been found by Davis *et al.* (2005).

It is clear from this research that several biodegradable packaging materials can be processed in home composting systems and yield compost materials suitable for plant growth. This capability will enable such materials to be disposed of in well-run home composting systems and result in waste diversion from municipal waste streams. However, we have also demonstrated that a number of packaging materials that typically biodegrade well in industrial, thermophilic high-temperature composting systems failed to biodegrade adequately in home composting environments that operate as low temperature, mesophilic environments.

At a practical level, these results suggest that it is vital to clearly distinguish biodegradable packaging materials that can be expected to biodegrade under ambient, mesophilic conditions typically found in UK home composting systems from those that biodegrade under the complete thermophilic–mesophilic (55–65°C) regime of an industrial composting systems. Labelling schemes and consumer education and information should support such a distinction.

5. CONCLUDING REMARKS

Biodegradable polymers will play a greater role in the packaging sector in the future. Post-use biodegradable plastics and other biowastes like paper, food and garden waste are generally unsuitable for landfill due to their potential to release methane under anaerobic conditions and their disposal by this method is inconsistent with policies like the EU Landfill Directive. Biodegradable bioplastics are most suitable for biological waste treatment through industrial and/or domestic composting and, subject to further demonstration, potentially in anaerobic digestion systems. They should ideally be separated at the household level from other, non-biodegradable materials and collected with organic waste, including food waste. By using these biological treatment methods, the total quantities of waste sent to landfill are reduced and the composts generated can be used as valuable soil improvers.

Implementing effective biological treatments for the developing range of biodegradable bioplastics requires the support of clear certification and labelling schemes. Biodegradable plastics that pass the relevant compostability standards will biodegrade well in industrial composting systems. However, as discussed, only some of those plastics will also biodegrade adequately under ambient, mesophilic regimes typical of UK home composters, and this distinction needs to be communicated effectively to the wider public (see Thompson *et al.* 2009*b*).

Bioplastic polymers have great potential to contribute to material recovery, reduction of landfill and use of renewable resources. Widespread public awareness of these materials and effective infrastructure for stringent control of certification, collection, separation and composting will be crucial to obtaining these benefits in full.

The authors would like to thank the Eng. D. programme of the EPSRC for support for the postgraduate studies of Gareth Davies. We are also grateful to Pactiv UK for their support as the collaborating body in the Eng. D. programme.

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PHILOSOPHICAL TRANSACTIONS OF THE ROYAL SOCIETY B BIOLOGICAL SCIENCES

Components of plastic: experimental studies in animals and relevance for human health

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Phil. Trans. R. Soc. B 2009 **364**, 2079-2096
doi: 10.1098/rstb.2008.0281

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PHILOSOPHICAL TRANSACTIONS OF THE ROYAL SOCIETY B

Phil. Trans. R. Soc. B (2009) **364**, 2079–2096
doi:10.1098/rstb.2008.0281

Review

Components of plastic: experimental studies in animals and relevance for human health

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Components used in plastics, such as phthalates, bisphenol A (BPA), polybrominated diphenyl ethers (PBDE) and tetrabromobisphenol A (TBBPA), are detected in humans. In addition to their utility in plastics, an inadvertent characteristic of these chemicals is the ability to alter the endocrine system. Phthalates function as anti-androgens while the main action attributed to BPA is oestrogen-like activity. PBDE and TBBPA have been shown to disrupt thyroid hormone homeostasis while PBDEs also exhibit anti-androgen action. Experimental investigations in animals indicate a wide variety of effects associated with exposure to these compounds, causing concern regarding potential risk to human health. For example, the spectrum of effects following perinatal exposure of male rats to phthalates has remarkable similarities to the testicular dysgenesis syndrome in humans. Concentrations of BPA in the foetal mouse within the range of unconjugated BPA levels observed in human foetal blood have produced effects in animal experiments. Finally, thyroid hormones are essential for normal neurological development and reproductive function. Human body burdens of these chemicals are detected with high prevalence, and concentrations in young children, a group particularly sensitive to exogenous insults, are typically higher, indicating the need to decrease exposure to these compounds.

Keywords: plastic; endocrine disruptor; phthalates; bisphenol A; polybrominated diphenyl ether; tetrabromobisphenol A

1. INTRODUCTION

Due to the high-volume generation of plastics and low production costs, consumers typically use many plastic items only once before discarding them. Environmental concerns associated with plastic use are not only because of the amount of waste, but also the leaching of substances out of the plastic. Components used in plastics, such as bisphenol A (BPA), polybrominated diphenyl ethers (PBDE), tetrabromobisphenol A (TBBPA) and phthalates, are released from plastic products, and are also known as endocrine-disrupting compounds (EDCs) owing to their ability to modulate the endocrine system. The detection of EDCs in the environment, biota and humans is of concern due to their potential to interfere with the physiology of living organisms (see Oehlmann *et al.* 2009).

In general, EDCs may disrupt the endocrine system by competing with endogenous steroid hormone binding to receptors and hormone transport proteins or by altering the metabolism or synthesis of endogenous hormones, eventually influencing recruitment of transcription factors and altering gene expression in cells (Wetherill *et al.* 2007). This is of particular concern for the developing organism, as it is sensitive to changes in the hormonal milieu, or drug or chemical exposure, which can result in organizational changes that are permanent (Guillette *et al.* 1995). This is in contrast to the situation in adults where changes in steroid hormones often lead to activational changes that are transient. Epigenetic changes have been seen following early developmental exposure to EDCs in animal studies, characterized by modified methylation patterns of genes leading to altered gene expression and phenotypic changes (Skinner & Anway 2007).

Human exposures to EDCs during this particularly vulnerable developmental time frame have been documented in a number of studies showing contamination of human breast milk (Norén & Meironyté 2000; Sun *et al.* 2004; Main *et al.* 2006), foetal liver (Schecter *et al.* 2007), amniotic fluid (Ikezuki *et al.* 2002) and cord blood (Schönfelder *et al.* 2002; Guvenius *et al.* 2003). Data from animal and human studies suggest

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One contribution of 15 to a Theme Issue 'Plastics, the environment and human health'.

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that EDCs may play a role in the development of cancer (Takashima *et al.* 2008), the reported decline in human sperm count (Mocarelli *et al.* 2008), temporal increases in the frequency of developmental abnormalities of the male reproductive tract (Sharpe & Skakkebaek 2008) and the trend towards precocious puberty in human females (Schoeters *et al.* 2008).

Phthalates function as plasticizers to give flexibility to high-molecular-weight polymers and are found in soft plastic products (the addition of phthalates makes brittle polyvinyl chloride (PVC) soft). In addition, phthalates are also used as chemical additives in gel capsules, cosmetics and other personal-care products. Bisphenol A is widely used in the production of epoxy resins, polycarbonate plastics and brominated flame retardants (BFRs). BPA is the monomer (as opposed to an additive) used for production of polycarbonate plastic intended for food and beverage contact and many other products; it is also used to make resins that line metal food and beverage cans. TBBPA and PBDEs make up approximately 59 and 33 per cent of the world market of BFRs (Law *et al.* 2006), respectively, and are incorporated in a wide variety of materials including plastics intended for electronics and appliances as well as fabrics.

This review will present some aspects of human exposure to these compounds and investigations describing effects in animal models. The potential for human health effects based on evidence derived from experimental studies in animals will be discussed for each of these chemicals.

2. PHTHALATES

(a) *Human exposure*

Phthalate esters have recently attracted the special attention of the scientific community, regulatory agencies and the general public as a consequence of their high production volume, widespread use as plasticizers and chemical additives and possible endocrine-related effects (Mylchreest *et al.* 1999). Phthalates can easily leach out of products to contaminate the external environment because they are not chemically bound to the plastic matrix or to other chemicals in formulations. Di-(2-ethylhexyl) phthalate (DEHP) is the most commonly used phthalate plasticizer for PVC (Matsumoto *et al.* 2002), but other compounds such as di-butyl phthalate (DBP), used as a fixative in cosmetics and other formulations, are also a cause of concern. Recent biomonitoring studies in the USA and Europe have detected relatively high levels of monoester metabolites of phthalates in the urine of the general population (Koch *et al.* 2004, 2006; Silva *et al.* 2004). In a recent study (Koch *et al.* 2006), the estimated median and 95th percentile of daily DEHP intake for a German population ($n = 85$) was 5.6 and 21 μg (kg bw^{-1}) d^{-1}, respectively. For children aged 3–14 years, these values were significantly higher—7.7 and 25 μg (kg bw^{-1}) d^{-1} ($n = 254$). Other studies have confirmed these data (reviewed by Heudorf *et al.* 2007). In addition, critically ill patients and neonates hospitalized in intensive care units may be exposed to significantly higher doses of phthalates that migrate from medical devices such as blood bags, catheters and nasogastric and intravenous tubes (Koch *et al.* 2006). Considering that developing organisms are particularly vulnerable to the effects induced by phthalates, newborns undergoing medical treatment in intensive care units may be at increased risk when compared with the general population. In a pilot study conducted by Koch *et al.* (2006), the daily DEHP intake of 45 neonates who were treated with various medical procedures was calculated. The median and 95th percentile was 0.042 and 1.780 mg DEHP (kg bw^{-1}) d^{-1}, respectively, and the maximum calculated intake was 2.3 mg DEHP (kg bw^{-1}) d^{-1}.

(b) *Experimental studies*

Although phthalates display low general toxicity, exposure to certain compounds is associated with disruption of endocrine and reproductive functions in experimental animals. The male reproductive tract seems to be particularly sensitive to phthalate exposure. Treatment of adult male rats with high doses of certain phthalates (e.g. 2000 mg DEHP kg^{-1} d^{-1}) results in rapid and severe changes in the testis (Gray & Gangolli 1986; Dostal *et al.* 1988). The observed alterations in spermatogenesis are thought to result from dysfunction in Sertoli cells, which cannot adequately provide physical and metabolic support to germ cells (Gray & Gangolli 1986). There is also experimental evidence showing that phthalates can target the Leydig cells and induce multiple hormonal disturbances (Akingbemi *et al.* 2004; Lin *et al.* 2008). However, most reproductive effects are not exerted by phthalate diesters themselves but rather by their active primary monoester metabolites formed in the liver, which are considered the proximate toxicants (Gray & Gangolli 1986). Recent evidence suggests that phthalates can also induce adverse responses in females following pre- and post-natal exposure (Grande *et al.* 2006, 2007; Gray *et al.* 2006).

Although most reproductive effects have been described in rats, phthalates can induce testicular injury in several other species including mice (Lamb *et al.* 1987), guinea pigs (Gray *et al.* 1982) and ferrets (Lake *et al.* 1976). However, some species, such as hamsters and non-human primates, seem to be less sensitive than rats (Gray *et al.* 1982; Foster *et al.* 1983; Kurata *et al.* 1998), and part of this variability may be attributed to differences in phthalate bioavailability (Foster *et al.* 1983; Ito *et al.* 2005). Accordingly, it has been argued that humans and non-human primates are less susceptible to the effects of phthalates, owing to the lower conversion of parent compounds into active monoester metabolites (Mckee *et al.* 2004). In a study by Kurata *et al.* (1998), no changes in testis weight or histopathology were observed in adult marmosets administered orally at 100, 500 or 2500 mg DEHP kg^{-1}d^{-1} for 13 weeks. More recently, the absence of testicular effects was also reported in young adult cynomolgus monkeys and juvenile marmosets exposed to high DEHP doses (500–2500 mg kg^{-1} d^{-1}) (Pugh *et al.* 2000; Tomonari *et al.* 2006). However, non-human primates have not been extensively evaluated during foetal and neonatal periods, which represent developmental windows especially susceptible to exogenous insults. A recent

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work by Hallmark *et al.* (2007) indicates that neonatal marmosets treated orally with 500 mg kg^{-1} d^{-1} DBP respond similarly to rats in relation to changes in testosterone production and Leydig cell alterations, although marmosets seem to be able to reverse the suppression of testosterone production more efficiently than rats. Due to the lack of data on *in utero* and early post-natal exposures, it is not possible to draw conclusions regarding possible developmental effects in non-human primates.

In fact, the main concern involving phthalates is related to the effects induced during pre- and early post-natal development. Recent animal toxicity studies indicate that exposure to certain phthalates results in severe disorders in the developing male reproductive system, including defects in the external genitalia, undescended testes and testicular lesions. Reproductive toxicity induced during the perinatal period was reported for DEHP (Gray *et al.* 2000; Andrade *et al.* 2006*a*), DBP (Mylchreest *et al.* 1999), butyl benzyl phthalate (Gray *et al.* 2000; Nagao *et al.* 2000) and, to a lesser extent, for diisononyl phthalate (Gray *et al.* 2000). Male rat offspring exposed *in utero* or both *in utero* and during lactation to high phthalate doses (e.g. 500 mg DBP $kg^{-1}d^{-1}$) display reproductive tract abnormalities compatible with disruption of androgen-dependent development and impaired testicular function (Mylchreest *et al.* 1999; Gray *et al.* 2000; Andrade *et al.* 2006*a*). The phenotypic alterations manifested in male offspring include cryptorchidism, hypospadias, atrophy or agenesis of sex accessory organs, testicular lesions (e.g. small fluid-filled testes), reduced daily sperm production, delayed preputial separation, permanent retention of nipples and decreased (feminized) anogenital distance.

Unlike other anti-androgens, which act by binding to the androgen receptor and thus inhibit its ability to respond to androgens, phthalates disrupt the development of androgen-dependent structures mainly by inhibiting foetal testicular testosterone biosynthesis (Parks *et al.* 2000; Wilson *et al.* 2004; Howdeshell *et al.* 2008). This effect is mediated by changes in gene expression of enzymes and proteins involved in testosterone production by foetal Leydig cells, including the steroidogenic acute regulatory (StAR) protein, which participates in the transport of cholesterol to the inner mitochondrial membrane, the step in the steroidogenic pathway that is considered to be rate-limiting (Shultz *et al.* 2001; Lehmann *et al.* 2004). Recently, the expression of another product of the foetal Leydig cell, insulin-like factor 3 (Insl3), has been shown to be reduced in phthalate-exposed animals (Wilson *et al.* 2004). Such an effect might explain the incidence of retention of testes in the abdomen (cryptorchidism) following phthalate exposure, as Insl3 is involved in the initial stages of testicular descent into the scrotum (Wilson *et al.* 2004; Foster 2006).

Testicular histopathology resulting from phthalate exposure is seen early in the foetal testis with the presence of dysgenetic areas characterized by malformed seminiferous cords containing multinucleated gonocytes and aggregates of Leydig cells (Barlow & Foster 2003; Fisher *et al.* 2003). In adult offspring, affected testes display reduced germ cell differentiation, Sertoli cell-only (SCO) tubules, Leydig cell aggregates and multinucleated giant germ cells (Gray *et al.* 2000; Fisher *et al.* 2003; Andrade *et al.* 2006*b*). Reductions in Sertoli cell number and/or proliferation have also been reported in neonatal rats treated with phthalates (Dostal *et al.* 1988; Li *et al.* 2000). However, this appears to be a transient effect, as no changes in the number of Sertoli cells are observed later in life (Dostal *et al.* 1988; Andrade *et al.* 2006*b*).

In both adult and developing males, disturbance of Leydig and Sertoli cell functions constitutes integral effects of phthalates. In addition, gene profiling data obtained by microarray analysis indicate that phthalates affect similar genetic targets in pre-pubertal and foetal rat testes (Lahousse *et al.* 2006). However, hormonal and local cell signalling perturbations during early development may irreversibly alter reproductive and endocrine functions in a manner that may not be predicted from post-natal exposure. In addition, although several target genes involved in the development and function of foetal Leydig and Sertoli cells have been identified so far (Shultz *et al.* 2001; Lehmann *et al.* 2004), the mechanisms by which phthalates alter the expression of these genes are currently unknown.

(c) *Relevance of phthalate effects for humans*

Interestingly, the spectrum of effects obtained following perinatal exposure of male rats to phthalates has remarkable similarities with the human testicular dysgenesis syndrome (TDS). According to Skakkebaek *et al.* (2001), the human TDS is characterized by low sperm counts, cryptorchidism, hypospadias and testicular cancer, and the clinical expression of these symptoms may vary with the severity of the syndrome. Accordingly, less severe manifestations would result in impaired spermatogenesis while other symptoms such as testicular cancer may be present in more severely affected individuals (Skakkebaek *et al.* 2001). Similar to the human TDS, the effects induced by phthalates in rats constitute a continuum of response with the most severe manifestations and highest incidence of reproductive tract malformations observed at high doses (Foster 2006). *In utero* exposure of rats to active phthalates such as DEHP and DBP has been suggested as a useful animal model for human TDS, as in both humans and rodents disturbance of foetal Leydig and Sertoli cell functions plays a major role in induction of TDS-like symptoms (Fisher *et al.* 2003).

However, the main question involving phthalates is whether the level of human exposure is sufficient to adversely impact male and/or female reproductive health. A study by Swan *et al.* (2005) reported an association between phthalate exposure and reduced anogenital distance in human infants, an effect that is also observed in rats. However, the range of doses typically used in animal studies is three to four orders of magnitude greater than the estimated daily exposure of humans. Only recently have researchers begun to investigate and report on possible biological changes at doses within the range of median human phthalate exposure (Lehmann *et al.* 2004; Andrade *et al.* 2006*c*; Lin *et al.* 2008).

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Some studies indicate that treatment of rat dams with active phthalates may result in non-monotonic (biphasic) dose–responses for the activity/expression of enzymes and proteins involved in the biosynthesis of steroid hormones in the offspring. Lehmann *et al.* (2004) studied alterations in genes coding for steroidogenic enzymes in the foetal testis of rats exposed to DBP. In this study, the authors reported reductions in several genes at doses that approach maximum human exposure levels. In a recent study with DEHP, Andrade *et al.* (2006*c*) have shown a striking biphasic dose–response for aromatase enzyme activity in the brain of neonatal males, with low-dose inhibition and high-dose stimulation. Since aromatase is a key enzyme in the biosynthesis of oestrogens, phthalate exposure might be associated with disturbances of the normal balance between androgens and oestrogens.

In addition to studies using low phthalate doses, recent studies have been conducted in order to evaluate the possibility of cumulative effects, since humans are exposed to multiple phthalates that are known to affect male reproductive development in rats. Howdeshell *et al.* (2007, 2008) demonstrated that different phthalates can act in a cumulative, dose-additive manner to reduce the testicular testosterone production by the rat foetal testis. Rider *et al.* (2008) reported dose-additive responses on reproductive malformations and androgen-dependent organ weights with developmental exposure to a seven-chemical anti-androgenic mixture that included some active phthalates. Coadministration of no-effect doses of DEHP and DBP reduces testicular testosterone levels and results in misshapen seminiferous cords and gonocyte multinucleation in the rat foetal testis (Martino-Andrade *et al.* 2008).

(d) *Conclusions*

Overall, the resemblance of phthalate effects in rats to human reproductive disorders has raised concerns over a possible link between phthalate exposure and human disease, even though most reproductive tract abnormalities in animal studies have been reported at dosages that probably result in internal dose concentrations that are well above those observed in humans. However, recent evidence for effects at lower doses as well as the results of combined toxicity studies and some epidemiological data on possible associations between phthalate exposure and reproductive effects in humans have provided further evidence for concern. The investigation of phthalate effects in non-human primates during pre- and early post-natal periods, the possibility of cumulative effects at low doses and a better understanding of the mode of action of phthalates and its relevance to humans are critical issues that should be addressed in future experimental studies.

3. BISPHENOL A

(a) *Sources and amounts of human BPA exposure*

Bisphenol A was reported to be a synthetic oestrogen (relative potency was not assessed) in 1936 (Dodds & Lawson 1936), 2 years before the oestrogenic activity of the structurally and functionally related drug diethylstilbestrol (DES) was described (Dodds *et al.* 1938). In the 1950s, polymer chemists discovered that BPA molecules could be polymerized to make polycarbonate plastic. Bisphenol A is also the base compound used in the manufacture of the resin lining of food and beverage cans in the USA and many other countries, although the Japanese can industry changed the formulation of the plastic lining of cans in the late-1990s. This voluntary action was reported to be associated with a loss of the previous correlation between use of canned drinks and urine levels of BPA in Japanese students as well as over a 50 per cent decrease in BPA levels (Matsumoto *et al.* 2003). These latter findings suggest that use of BPA-based resins to line cans contributes significantly to the human body burden of BPA. Food contact items (can lining, food packaging and food and beverage containers) are thought to be the major contributors to the median and mean values of approximately 2–4 ng ml^{-1} of unconjugated BPA detected in adult and foetal serum (Vandenberg *et al.* 2007). Pure water was poured into food cans that had contained different products and a polycarbonate food storage container. The products were heated to 100°C for 24 h, and BPA was analysed by HPLC with CoulArraay detection. The data presented in figure 1 reveal that all products leached detectable levels of BPA, although there were differences in the amount of leaching from different manufacturers' products. As expected, the cans that had contained the acidic tomato sauce resulted in the highest BPA values, since acid accelerates hydrolysis of the ester bond linking BPA molecules in polycarbonate and resins.

Another common use for BPA is as the monomer in dental sealants and composites used for fillings, from which BPA leaches in variable amounts and for different lengths of time depending on the product (Joskow *et al.* 2006). BPA is also an additive (referred to as a plasticizer) in PVC plastic products. For example, BPA has been reported to be present in PVC products such as stretch film (Lopez-Cervantes & Paseiro-Losada 2003). It is also used in printer ink and to coat paper used for receipts (referred to as 'carbonless paper'). It is thus a major contaminant in recycled paper products (Vandenberg *et al.* 2007). Polycarbonate food storage and beverage containers (the hard, clear containers, which may be tinted in the case of sport water bottles or baby bottles) cause concern regarding their potential to leach BPA because they are re-usable (Nerin *et al.* 2003), and repeated use leads to an increase in leaching (Brede *et al.* 2003). Many of these containers are marketed for use in the microwave, despite the fact that heating is known to increase BPA leaching levels. It is a basic characteristic of the ester bond linking BPA molecules together in polycarbonate plastic and resins that the rate of breaking of the bond by hydrolysis increases with heat, releasing free BPA (Bae *et al.* 2002), and an increase in leaching also occurs as a result of either an increase or decrease in pH (Welshons *et al.* 2006; Vandenberg *et al.* 2007). When food or liquids (such as beer) are placed into a can, they are heated to a high temperature for sterilization. The consequence is that food and beverages in cans have variable levels of BPA based on whether the contents are lipophilic

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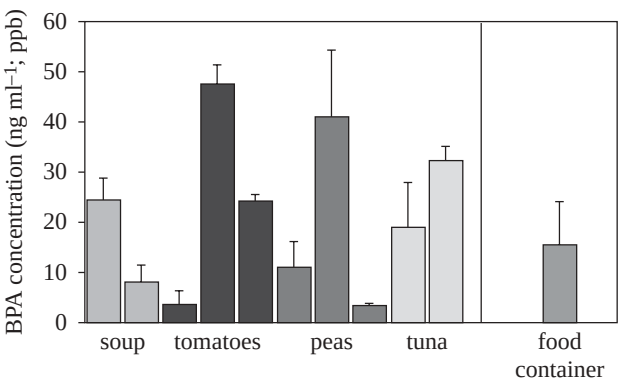

Figure 1. Bisphenol A (BPA) concentration in water placed into food cans that had contained different products. Two or three different brands of each product were examined. The cans were emptied, cleaned and rinsed with water that did not contain detectable levels of BPA. The 10 cans and a single new reusable 'microwave safe' polycarbonate food container were filled with HPLC-grade water and heated to 100°C for 24 h. Bisphenol A was extracted and analysed by HPLC with CoulArray detection (limit of detection was 0.01 ppb). Negative controls in glass bottles gave undetectable levels of BPA. Spiked samples in glass containers gave >95 per cent recovery. Significant leaching of BPA occurred from each type of product tested.

and/or acidic or alkaline (figure 1), all of which increase leaching (see the Environmental Working Group website for additional data on leaching of BPA from cans, at www.EWG.org).

A critical issue regarding routes of exposure to BPA was revealed by the Canadian Ministry of the Environment, which determined that BPA poses a threat to aquatic wildlife at current levels at which it is found in aquatic ecosystems (Canada 2008). This government report and other reviews (Vandenberg *et al.* 2007) identify that a number of studies show that BPA is leaching out of products being thrown into landfills and getting into ground water, contaminating rivers, streams and drinking water. While in an aerobic environment, BPA has a half-life of days, has a greater density than water and thus ends up in the sediment. In an anaerobic environment, BPA does not degrade. The Canadian Government is thus taking a regulatory approach to this problem that involves not only trying to limit the use of BPA in food and beverage containers, but also involves trying to deal with the problem of disposal of BPA-containing plastic that cannot be recycled to produce new BPA-based products.

The US Centers for Disease Control and Prevention (CDC) has measured BPA in the urine of people in the USA as part of the last national health survey (Calafat *et al.* 2008). The CDC reported that 93 per cent of people had detectable levels of BPA in their urine. Interestingly, the median and mean levels of unconjugated (parent) BPA reported in blood, as well as the lower and upper range reported for women and their foetuses at the time of parturition in Germany (Schönfelder *et al.* 2002), were virtually identical to values reported for total BPA in urine by the CDC. The findings by Schönfelder *et al.* are consistent with reports of blood levels of unconjugated BPA in people from other countries such as Japan (Vandenberg *et al.* 2007). In this review of the published literature, the authors determined that blood levels in humans identified by numerous analytical methods consistently showed approximately 10-fold higher median or mean levels of unconjugated BPA than average blood levels found throughout a 24 h period after laboratory rats were administered a BPA dose equal to the tolerable daily intake (TDI). The TDI corresponds to the amount of a chemical a person can be exposed to on a daily basis over an extended period of time (usually a lifetime) without suffering deleterious effects; this is also referred to by US regulatory agencies as the 'reference dose' and 'acceptable daily intake dose'.

Currently, a dose of 50 $\mu g\,kg^{-1}\,d^{-1}$ is considered 'safe' for daily human consumption by the US Environmental Protection Agency (EPA) (IRIS 1988) and Food and Drug Administration (FDA). Findings reported by Vandenberg *et al.* (2007) led to a consensus conclusion by 38 scientists who attended a US National Institutes of Health (NIH) sponsored conference on BPA (vom Saal *et al.* 2007) that current levels of human exposure to BPA already exceed the presumed safe daily exposure dose. The data presented in the review indicate that: (i) BPA is detected at the nanogram per millilitre (ppb) levels; (ii) unconjugated (bioactive) BPA is found in blood and (iii) the same levels are found in its conjugated (glucuronidated or sulphated) form in urine (Vandenberg *et al.* 2007). This is in sharp contrast to a review that concluded that BPA is rapidly and completely cleared from the blood of adults following oral absorption (Willhite *et al.* 2008). The conclusion that all ingested BPA is immediately metabolized (conjugated) to biologically inactive metabolites is thus not consistent with the published literature. The 'back calculation' approach of estimating human exposure to BPA based on the assumption that all orally absorbed BPA is immediately cleared from blood into urine is an invalid approach if the chemical is not immediately and completely metabolized (Barr *et al.* 2005).

BPA is one of the highest production volume chemicals in commerce, with over 6 billion pounds produced in 2003 (Burridge 2003), and a significant increase in production volume was expected at that time. In order to accurately estimate human exposure to BPA, we would need to know what products it is used in, since without this knowledge we have no way to determine the potential for different routes of exposure. This is currently not possible as identification of the chemicals used in products, such as baby toys, food and beverage containers or paper products, is not required. The latter is interesting in that printer ink contains BPA, and the 'carbonless paper' used to provide receipts for purchases is coated with BPA (the BPA reacts with dye in the paper in response to pressure or heat). Newspapers and carbonless paper could be significant sources of transdermal BPA exposure, and all recycled paper has BPA in it from these sources (Vandenberg *et al.* 2007). Dermal absorption of BPA from these potential sources of exposure has not been investigated.

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(b) *Adverse health effects of BPA in laboratory animals*

There were three reports released in 2007 and 2008 in North America: two from the USA and one from Canada (discussed subsequently). The first was released as a consensus statement from an NIH-sponsored conference on BPA (vom Saal *et al.* 2007). This document was co-authored by 38 scientists who are experts in the field of endocrine disruption, and the majority of scientists at this conference had conducted research on BPA. The consensus document was a summary of the findings in five accompanying review articles that covered the entire published literature relating to the health effects of BPA as of the end of 2006 (Crain *et al.* 2007; Keri *et al.* 2007; Richter *et al.* 2007*a*,*b*; Vandenberg *et al.* 2007; Wetherill *et al.* 2007).

(i) *Adult exposure, experimental findings and relevance for human health*

There is extensive published literature showing the effects of acute exposure to very low doses of BPA in adult experimental animals: rats, mice and various aquatic species (Richter *et al.* 2007*a*,*b*). However, an issue that has generated much confusion is the rate at which BPA is metabolized. As described earlier, if one makes the assumption that virtually all BPA exposure is via an oral route and that virtually all BPA is immediately conjugated in the liver after absorption from the gut (transported from the gut to the liver via the direct hepatic portal vessels), then of course there would be no concern about BPA posing a threat to adults. However, this assumption is not consistent with the published biomonitoring literature regarding levels of unconjugated BPA in human blood (Vandenberg *et al.* 2007). Importantly, no rodent or human experiment has been conducted that involves chronic exposure, although the human biomonitoring data suggest virtual continuous exposure (Calafat *et al.* 2008). This is a major gap that needs to be filled.

The issue of route of exposure to BPA has generated significant controversy, since the default assumption is that oral exposure accounts for most (or virtually all) BPA exposure in humans. It is well known that exposure to BPA via injection, which does not result in the first-pass liver metabolism (glucuronidation/sulphation) that would occur following oral exposure, results in about a 10–20-fold higher amount of BPA in the blood relative to levels following oral administration (Vandenberg *et al.* 2007). However, if BPA is injected into rodents at doses of BPA that are thousands of times lower than the lowest adverse effect level (LOAEL) of 50 mg kg^{-1} d^{-1} that was used to establish the safe dose of 50 μg kg^{-1} d^{-1}, it is logical to assume that findings from studies that used injection as the route of administration are also relevant to assessing the potential health hazards posed by BPA, while recognizing that a correction for differences in pharmacokinetics based on route of administration is required. Just a few examples of effects of low doses of BPA in adult rodents are a significant stimulation of insulin secretion followed by insulin resistance in mice (Ropero *et al.* 2008), a significant decrease in daily sperm production in rats (Sakaue *et al.* 2001), a decrease in maternal behaviour in mice (Palanza *et al.* 2002) and disruption of hippocampal synapses, leading to the appearance of a brain typical of that seen in senility in both rats and monkeys (MacLusky *et al.* 2005; Leranth *et al.* 2007, 2008).

Related to the fact that type 2 diabetes is increasing in many regions of the world is the finding that exposure of adult mice to a low oral dose of BPA (10 μg kg^{-1} d^{-1}) resulted in stimulation of insulin secretion that was mediated by oestrogen receptor (ER) alpha, occurring as a rapid response mediated by the extracellular signal-regulated kinases 1 and 2 (ERK1/2) pathway; and that the prolonged hypersecretion of insulin was followed by insulin resistance and postprandial hyperinsulinaemia (Alonso-Magdalena *et al.* 2005, 2008; Ropero *et al.* 2008). The fact that these very-low-dose studies of BPA are confirmed in cell culture studies that reveal molecular pathways that mediate effects of BPA in the low parts per trillion range has been reviewed (Wetherill *et al.* 2007). A consensus conclusion from the NIH-sponsored meeting on BPA was that the more recently discovered rapid-response pathway (as opposed to the nuclear ERs acting as ligand-activated transcription factors) may mediate many, but not all, of the low-dose effects of BPA (vom Saal *et al.* 2007).

The prediction from mechanistic studies in mice concerning the molecular pathways by which very low doses of BPA stimulate insulin production and secretion, which is then followed by insulin resistance (Ropero *et al.* 2008), leads to the fact that BPA may be related to elevated blood insulin and glucose, as well as insulin resistance. In addition, Hugo *et al.* (2008) reported that human adipocytes removed from different regions showed a marked suppression of the critical regulatory cytokine adiponectin, with the maximum response occurring at 1 nM (0.23 ppb), directly in the range of human exposure to BPA. A decrease in adiponectin is related to an increased risk for type 2 diabetes and cardiovascular disease and heart attack (Beltowski *et al.* 2008). It is thus of considerable interest that Lang *et al.* (2008) reported that in an analysis of data from 1455 people examined for BPA levels in urine as part of the US National Health and Nutrition Examination Survey (NHANES) conducted in 2003–2004, they found a significant relationship between urine levels of BPA and cardiovascular disease, type 2 diabetes and abnormalities in liver enzymes. This report, suggesting links between BPA and some of the most significant and economically burdensome human diseases, is based on a cross-sectional study and therefore cannot establish causality, but the fact that these findings are related to other studies that identify plausible mechanisms by which BPA at current levels of human exposure could result in these diseases greatly strengthens the importance of the findings (vom Saal & Myers 2008).

(ii) *Developmental exposure, experimental findings and relevance for human health*

The greatest concern regarding exposure to BPA is during development: foetuses, neonates, infants,

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children and adolescents. The laboratory animal research on BPA is unique in that there are now hundreds of studies that have examined doses of BPA within the range of human exposure rather than the more typical approach in regulatory toxicology of only testing a few doses that are typically thousands of times higher than human exposure levels. The surprise associated with the first 'low dose' publication on the effects of BPA in laboratory mice in 1997 (Nagel *et al.* 1997) was that effects on the reproductive system in male offspring were found at a daily oral dose to pregnant mice that was 25 000 times lower than had ever been examined, and 25 times below the still current safe daily exposure dose according to the US FDA and US EPA, as well as the European Food Safety Authority (EFSA 2006). The lowest dose of BPA that had previously been examined was 50 mg kg^{-1} d^{-1} in some traditional very-high-dose toxicological studies conducted in the1980s. The 50 mg kg^{-1} d^{-1} was thus the LOAEL, and this dose was divided by 10 to 'estimate' the no observable adverse effect level (NOAEL) and divided by 1000 for the EPA to estimate that 50 μg $kg^{-1}d^{-1}$ was safe for daily human exposure (IRIS 1988). Numerous reviews have been published discussing the assumptions used in estimating safe exposure levels for endocrine-disrupting chemicals such as BPA (vom Saal & Sheehan 1998; Markey *et al.* 2003; Welshons *et al.* 2003, 2006; vom Saal & Hughes 2005; Myers & vom Saal 2008).

One of the main concerns with the adverse effects reported in response to developmental exposure to very low doses of BPA (that produce blood levels in animals below those in humans) is that they all relate to disease trends in humans (table 1). For example, there is an obesity epidemic in many regions of the world, and developmental exposure to BPA increases body weight later in life (Howdeshell *et al.* 1999; Takai *et al.* 2000; Rubin *et al.* 2001). In addition, neonatal exposure to a low dose of DES (1 μg kg^{-1} d^{-1}) stimulated a subsequent increase in body weight and fat in CD-1 mice, while a dose 1000 times higher resulted in a significant decrease in body weight (Newbold *et al.* 2004).

Relevance for human health. The largest literature on the adverse effects of BPA exposure during development concerns adverse effects on brain structure, chemistry and behaviour (Richter *et al.* 2007*a*,*b*). One of the most interesting aspects of this literature is that there is a consistent finding of a loss of sex differences in brain structure, chemistry and behaviour owing to foetal/neonatal exposure to low doses of BPA. BPA thus appears to interfere with the normal processes that govern sexual differentiation, with brain changes reported in both males and females, depending on the outcome measured (Fujimoto *et al.* 2006; Rubin *et al.* 2006; Palanza *et al.* 2008). The implications at the population level for disruption of normal socio-sexual behaviours have not been extensively studied, although there are reports of changes in play behaviour (Dessi-Fulgheri *et al.* 2002) as well as other socio-sexual behaviours (Farabollini *et al.* 2002) that could impact population dynamics.

There are also numerous studies of the effects of low doses of BPA on the development of the female (Soto *et al.* 2008) and male reproductive organs in female rats and mice. Findings include chromosomal abnormalities in oocytes in females (Susiarjo *et al.* 2007), and long-term effects on accessory reproductive organs that are not observed until mid-life, such as uterine fibroids and para-ovarian cysts (Newbold *et al.* 2007). Other studies have shown that very low doses of BPA during pre-natal or neonatal development can result in permanent effects in male rats and mice. Low doses of BPA cause a decrease in daily sperm production and an increase in prostate size (vom Saal *et al.* 1998), an increase in prostatic androgen receptors (Gupta 2000; Richter *et al.* 2007*a*,*b*) and a progression from hyperplasia of prostate basal (stem) cells in the primary prostatic ducts during foetal life (Timms *et al.* 2005) to basal cell squamous metaplasia in adulthood (Ogura *et al.* 2007) and eventually to early stage prostate cancer (prostatic interepithelial neoplasia or PIN) in response to adult administration of testosterone and oestradiol (Ho *et al.* 2006).

Table 1. Prenatal–neonatal exposure of mice and rats to BPA at human exposure levels in relation to human health trends.

effects in mice and rats	human health trends
prostate hyperplasia and cancer	prostate cancer increase
mammary hyperplasia and cancer	breast cancer increase
abnormal urethra/ obstruction	hypospadias
sperm count decrease	sperm count decrease
early puberty in females	early sexual maturation
ovarian cysts/uterine fibroids	polycystic ovary syndrome/ uterine fibroids
abnormal oocyte chromosomes	miscarriage
body weight increase	obesity increase
insulin resistance	type 2 diabetes
hyperactivity/impaired learning	attention deficit hyperactivity disorder

(c) *Conclusions*

The required testing of chemicals for regulatory purposes is not aimed at understanding molecular mechanisms and focuses on effects occurring at high doses that are typically not relevant for human exposure scenarios. Over the last 10 years, there has been a dramatic shift in the approach of scientists with regard to investigating doses that are relevant for human exposure to study the effect of BPA in laboratory animals. This has led to a totally unique toxicological literature revealing extensive evidence that effects in laboratory animals are occurring at blood levels that are lower than those found in the average person in a developed country. This is clearly of great concern for possible impact on human health.

The assumptions used in chemical risk assessments, such as all dose–response curves are monotonic, and there is a threshold dose below which no effect

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occurs at which the system is 'off', clearly do not apply to endogenous hormones, hormonally active drugs or hormonally active chemicals. How can there be a threshold at which no oestrogen responses occur for a chemical that is adding oestrogenic activity to a system that is already 'on' at zero dose of the exogenous chemical (Sheehan *et al.* 1999; Sheehan 2005)? Also, all hormones show responses that are non-monotonic, and non-monotonic dose–response curves have been reported in many BPA studies (Richter *et al.* 2007*a*,*b*; Alonso-Magdalena *et al.* 2008). The need to integrate concepts of endocrinology in the assumptions underlying chemical risk assessments will some day result in the development of a new system, and the data from findings with low doses of BPA will play an important role in this paradigm shift (Myers & vom Saal 2008).

4. BROMINATED FLAME RETARDANTS

(a) *Uses of TBBPA*

Brominated flame retardants function by increasing the time between ignition of a fire and flash over, which is the point when enough heat is generated to cause combustion of flammable materials. TBBPA is the classical halogenated flame retardant chemically bonded to epoxy and polycarbonate resins. It is present in printed circuit boards and casings used in personal computers, printers, fax machines and copiers. Dimethyl TBBPA is added to acrylonitrile–butadiene–styrene (ABS) resin and high impact polystyrene (HIP). This derivative is blended with the polymers, meaning that it exists free in the chemical matrix. ABS resins have a wide variety of applications including automotive parts, pipes and fittings, and domestic and office appliances. Polystyrene is used in packaging, electrical and electronic equipment enclosures for televisions, furniture and construction materials. Another derivative, bis (2-hydroxyethyl ether) TBBPA is used as a flame retardant for paper and textile adhesives and coatings (Alaee *et al.* 2003).

(b) *Exposure to TBBPA*

Products with both additive and chemically bonded forms of TBBPA have been shown to release TBBPA into the environment (Birnbaum & Staskal 2004), resulting in detection in sewage sludge (Oberg *et al.* 2002), soil, sediments, birds, fish (Morris *et al.* 2004) and air from different occupational settings (Sjödin *et al.* 2001). TBBPA serves as a source of environmental BPA as it has been shown to break down to BPA in marine sediments (European Union Risk Assessment Report 2005).

Although studies indicate high first-pass metabolism of TBBPA in rats and humans owing to rapid conjugation with glucuronic acid and elimination in the bile (Kuester *et al.* 2007), TBBPA has been detected in cow and human milk (Thomsen *et al.* 2002*a*; Antignac *et al.* 2008), human serum (e.g. Hayama *et al.* 2004), human adipose tissue (Johnson-Restrepo *et al.* 2008) and umbilical cord serum (Antignac *et al.* 2008). Evaluation of BFRs in archived human serum samples in Norway showed a temporal increase in concentrations of six PBDE congeners and TBBPA over the period 1977–1999. The concentrations for different age groups were relatively similar, except for the 0–4 year olds who had 1.6–3.5 times higher serum concentrations (Thomsen *et al.* 2002*b*).

(c) *Experimental studies on TBBPA*

There are only a few published studies regarding the toxicology of TBBPA. Information regarding the endocrine-disrupting potential of TBBPA mainly comes from *in vitro* studies and *in vivo* studies performed in quail, fish and tadpoles; however, two papers describing a study in the rodent model have been recently published.

(i) *Thyroid hormone effects*

An increase in thyronine (T3) in female rat offspring and a reduction in circulating total thyroxine (T4) were observed in both sexes in a reproductive study with exposure to TBBPA in food, beginning 70 or 14 days prior to mating of F0 males and females, respectively, and which continued throughout gestation and lactation up to 14 weeks of age at necropsy (Van der Ven *et al.* 2008). Thyroid hormone concentrations were similarly altered at 12 weeks of age in a subacute toxicity study (28 days of repeated dosing of TBBPA) with a significant decrease in T4 and increase in T3 in males. Parallel changes were observed in females; however, they were not statistically significant (Van der Ven *et al.* 2008). In the same reproductive experiment, appraisal of brainstem auditory evoked potentials (BAEP), a means to detect effects on hearing, indicated changes in hearing latency and hearing threshold similar to previous reports on developmental exposure to polychlorinated biphenyls (PCBs) (Lilienthal *et al.* 2008). It is plausible that the changes in thyroid hormone status mediated these effects as thyroid hormones play a crucial role in the development of auditory function, and the benchmark dose levels for changes in thyroid hormones (2.3–30.8 mg kg^{-1} d^{-1}) and BAEP (1–40 mg kg^{-1} d^{-1}) were in the same range (Van der Ven *et al.* 2008).

(ii) *Binding to transthyretin*

TBBPA has an even closer structural relationship to T4 than PCBs. *In vitro* competitive binding assays demonstrated that TBBPA binding to human transthyretin (TTR) is more potent than the natural ligand (Meerts *et al.* 2000; Hamers *et al.* 2006). It is theorized that these compounds may decrease serum T4 concentrations by displacing it from the carrier proteins, leading to increased clearance. This displacement could also make more T4 available for deiodination, leading to increased formation of T3 or reverse T3.

(iii) *Binding to thyroid hormone receptor and thyroid hormone activity*

During development, most tadpole tissues are influenced by thyroid hormones, making them suitable to evaluate potential disruption of thyroid homeostasis. An *in vivo* study simultaneously exposing *Rana rugosa* tadpoles to T3 and TBBPA found TBBPA

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suppression of T3 induced tail shortening, indicating a thyroid hormone antagonist effect (Kitamura *et al.* 2005*a*). While only mild effects on larval development were observed, a study in *Xenopus laevis* tadpoles also suggested thyroid hormone antagonism by TBBPA (Jagnytsch *et al.* 2006). *In vitro*, TBBPA competes with T3 binding to a nuclear suspension from rat pituitary Mt/T/e-2 cells (Kitamura *et al.* 2002) and was shown to inhibit the activity of T3 in Chinese hamster ovary cells transfected with human TRα or TRß (Kitamura *et al.* 2005*a*).

(iv) *Sex steroid effects*

An increase in pituitary weight in male offspring was noted in the reproductive study (Van der Ven *et al.* 2008), which correlates with previously published *in vitro* findings from Kitamura *et al.* (2002, 2005*b*), showing that TBBPA increased proliferation of GH3 cells (rat pituitary cell line) and growth hormone production. Ghisari & Bonefeld-Jorgensen (2005) also observed that TBBPA stimulates GH3 cell growth, which was inhibited by the anti-oestrogen ICI 182 780, suggesting that this effect is ER-mediated.

In vitro studies indicate that the lower brominated forms of TBBPA have higher affinity with the ER than the higher brominated analogues (Samuelsen *et al.* 2001). In MCF-7 human breast cancer cells, TBBPA functioned as a partial ER agonist (Olsen *et al.* 2003) and demonstrated mixed agonist/antagonist activities in estrogen responsive element (ERE)-luciferase reporter gene assays. The oestrogenic activity was confirmed *in vivo* using the uterotrophic assay (Kitamura *et al.* 2005*b*).

Inhibition of oestradiol sulphation by TBBPA has been reported *in vitro* (Kester *et al.* 2002; Hamers *et al.* 2006), which suggests the potential for impaired elimination *in vivo* leading to increased biologically active oestradiol concentrations. Environmentally relevant TBBPA concentrations have been shown to decrease reproductive success in zebra fish (Kuiper *et al.* 2007) and inhibition of oestradiol metabolism in lake trout (Jurgella *et al.* 2006). In male Japanese quail, however, no oestrogenic effects after embryonic exposure to TBBPA were observed for reproductive behaviour, plasma testosterone and testicular morphology (Berg *et al.* 2001; Halldin *et al.* 2005).

(d) *Products with PBDEs*

Two (penta and octa formulations) of the three main commercial mixtures of PBDEs added to polymers for the manufacture of goods are no longer in production. The penta mixture was applied to polyurethane foams used in furniture, mattresses, carpet pads and automobile seats and styrene plastics used for electrical appliances and flame-retardant textiles. The octa-technical mixture was blended with ABS resins used in automobile electronics, home and office appliances and high-impact sport equipment and toys. The deca product is added to a variety of polymers, and examples of end products include fabric backings and housings for electronics (Frederiksen *et al.* 2008).

(e) *Exposure to PBDE*

The presence of PBDEs in breast milk, adipose tissue and serum has been confirmed in several studies (as reviewed in Frederiksen *et al.* 2009). An examination of a cohort of 4-year-old children revealed that breast-fed children have 6.5 times higher average body burden of total PBDEs than formula-fed children (Carrizo *et al.* 2007). Detection of PBDEs in liver tissue of human foetuses (Schecter *et al.* 2007) and in cord blood (e.g. Antignac *et al.* 2008) demonstrates that *in utero* exposure is taking place. On a pro-kilogram basis, studies have described higher levels of exposure to PBDEs for children than adults (as reviewed in Frederiksen *et al.* 2009).

(f) *Animal studies investigating developmental exposure to PBDEs*

There are more published studies regarding effects following *in vivo* exposure to PBDEs than for TBBPA. Animal studies have revealed the potential for endocrine disruption and effects on neurobehaviour and the reproductive system.

(i) *Thyroid hormone*

In rodent studies, perinatal or peripubertal exposure to PBDEs causes a reduction in T4 while effects on T3 and TSH are less consistent. Perinatal exposure to 1, 10 or 30 mg kg^{-1} d^{-1} DE-71 (commercial penta-mixture) (Zhou *et al.* 2002) or 18 mg $kg^{-1}d^{-1}$ (Ellis-Hutchings *et al.* 2006) did not alter T3 concentrations. However, a concomitant decrease in T3 and an increase in thyroid stimulating hormone (TSH) in males only were observed following peripubertal exposure to 30 or 60 mg kg^{-1} d^{-1} DE-71 (Stoker *et al.* 2004), and another study reported a decrease in T3 in female rats following exposure to 100 or 300 mg kg^{-1} d^{-1} DE-71 or 60 or 100 mg kg^{-1} d^{-1} DE-79 (octa-commercial mixture) on post-natal day (PND) 28–32 (Zhou *et al.* 2001). Administration of 300 μg kg^{-1} d^{-1} BDE-99 (2,2′,4,4′,5-pentabromodiphenyl ether) on gestation day (gd) 6 resulted in a transient decrease in T3 in male offspring on PND 1 with reductions in T4 occurring in both male and female offspring on PND 22 (Kuriyama *et al.* 2007).

(ii) *Effects on sex steroids and the reproductive system*

Following gestational exposure, *in vivo* oestrogen activity for BDE-99 has been indicated by increased expression of mRNA of oestrogen responsive genes in the uterus (Ceccatelli *et al.* 2006). Alterations in the sex hormone profile have been reported, including reductions in circulating oestradiol and testosterone in adult male offspring (Lilienthal *et al.* 2006) and lower circulating oestradiol concentrations without changes in whole ovarian aromatase activity in young female rat offspring exposed to 700 μg kg^{-1} of BDE-47 on gd 6 (Talsness *et al.* 2008). Possible impairment of androgen function following high-dose peripubertal exposure to DE-71 was indicated by postponed puberty and delayed growth of androgen-dependent tissues in male rats without an effect on circulating testosterone concentrations (Stoker *et al.* 2004). Additional *in vitro* studies by the same group suggest

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that the observed effects may be because of binding of some of the congeners in the mixture acting as antagonists to the androgen receptor (Stoker *et al.* 2005).

Functional changes to the gonads have been observed following *in utero* exposure to PBDEs. Administration of an agent used to treat hyperthyroidism, 6-*n*-propyl-2-thiouracil (5 mg l^{-1} drinking water on gd 7–21), or 60 or 300 μg BDE-99 kg^{-1} on gd 6 (Kuriyama *et al.* 2007) resulted in a reduction in spermatid and sperm counts without an influence on circulating testosterone concentrations in adult F1 males (Kuriyama *et al.* 2005). Whether the impaired spermatogenesis is accompanied by changes in the testicular cell population based on DNA ploidy (flow cytometry analysis) was examined in isolated testicular cells from the other testis (12 per group) of the same animals. DNA staining was performed by suspending the cells in phosphate buffered saline with 100 μg ml^{-1} ribonuclease A and 50 μg ml^{-1} propidium iodide. Propidium iodide fluorescence was measured at 620 nm to differentiate between mature haploid cells (elongated spermatid), immature haploid cells (round/elongating spermatid) and diploid cells (spermatogonia, spermatocyte, Sertoli cells, Leydig cells). The number of Sertoli cells in the seminiferous epithelium is related to sperm production since each cell supports a finite number of germ cells (Russel *et al.* 1990). This cell number was also determined in histological sections from other F1 males (six per group) on PND 110. Sertoli cell nucleoli were counted in 25 round or nearly round seminiferous tubule cross-sections chosen at random, and 25 Sertoli cell nucleoli diameters were measured for each animal. These counts were corrected for section thickness and the smallest recognizable nucleolar profile (cap section) as described previously (Russel *et al.* 1990).

A statistically significant decrease in the ratio of haploid/diploid cells was observed in the PBDE-99 group (figure 2), which corroborates the reduced number of spermatid/sperm counts observed. We observed slight differences in the ratio of mature haploid (mostly elongated spermatid)/diploid cells in treated animals, and this effect was more pronounced when the ratio of immature haploid (round/elongating spermatid)/diploid cells was compared with controls. No change in the number of Sertoli cells/seminiferous tubule cross-section was observed among the groups (figure 2), indicating that the reduction in haploid cells is because of another cause and supporting the finding that the changes in these ratios are owing to reductions in the haploid populations. The observed impaired spermatogenesis in male offspring exposed to low-dose BDE-99 during development (Kuriyama *et al.* 2005) has been confirmed by flow cytometry analysis and is not mediated by reduced numbers of Sertoli cells. Alternatively, a hormonal-dependent mechanism can be proposed as sperm production is dependent on permissive actions of follicle stimulating hormone (FSH) and testosterone and, therefore, luteinizing hormone (LH). Although significant differences in testosterone and LH concentrations in serum from adult animals were not found, the possibility of hormonal changes during early development cannot be ruled out. Functional changes in the seminiferous epithelium leading to impairment of normal spermatogenesis in adult rat offspring can also be speculated.

In adult female offspring, ultrastructural changes in the ovary following low-dose gestational exposure to either PBDE-99 (Talsness *et al.* 2005) or BDE-47 (Talsness *et al.* 2008) and effects on folliculogenesis after administration of 1 or 10 mg kg^{-1} d^{-1} of BDE-99 during gestation (Lilienthal *et al.* 2006) were observed. Altered folliculogenesis evaluated around the time of puberty after low-dose exposure to BDE-47 during gestation has also been reported (Talsness *et al.* 2008). The changes in the latter study were characterized by a reduction in antral follicle numbers and circulating estradiol concentrations. Whether these changes are because of thyroid hormone disruption or alterations in gonadotropins and/or sex hormones is unclear.

The animal data thus far suggest that PBDEs may affect the reproductive system via alterations in thyroid hormone profile and/or sex steroid action. Aberrations in thyroid hormone status are known to adversely affect reproduction. In animal studies, thyroid hormone has been shown to play an integral role in testicular development (Cooke *et al.* 1992) and affect ovarian follicular maturation (Baldridge *et al.* 2004).

(iii) *Neurobehaviour*

The period of rapid brain growth represents a particularly vulnerable developmental window to insults and is characterized by a dramatic increase in the number of cells and myelination, cell migration, dendritic and axonal growth and the formation of neural connections. Exposure to PBDEs during the brain growth spurt of mice modified spontaneous motor behaviour and habituation to new surroundings (e.g. Viberg *et al.* 2006) and altered levels of proteins involved in brain maturation (Viberg *et al.* 2008). Other studies have found changes in spontaneous motor activity following early post-natal exposure to mice, as well as effects on one measure of the ontogeny of sensorimotor integration (Rice *et al.* 2007) and subtle differences in neuromotor development (Branchi *et al.* 2002; Gee & Moser 2008). In addition, examination of cognitive function revealed deficits as mice exhibited impaired learning and memory on the Morris water maze test following early post-natal PBDE exposure (Viberg *et al.* 2003).

Administration of BDE-99 to rats during gestation at a dose (300 ug kg^{-1} on gd 6) resulting in dam adipose tissue concentrations (Kuriyama *et al.* 2007) only slightly higher than those reported for this congener in humans (Johnson-Restrepo *et al.* 2005) led to hyperactivity in the rat offspring (Kuriyama *et al.* 2005).

(g) *Relevance of exposure to flame retardants for human health*

(i) *Reproduction*

It is known that diseases of the thyroid gland affect the reproductive capacity of women. In particular, hypothyroid women may exhibit anovulation, and hypothyroidism is associated with hyperprolactinaemia,

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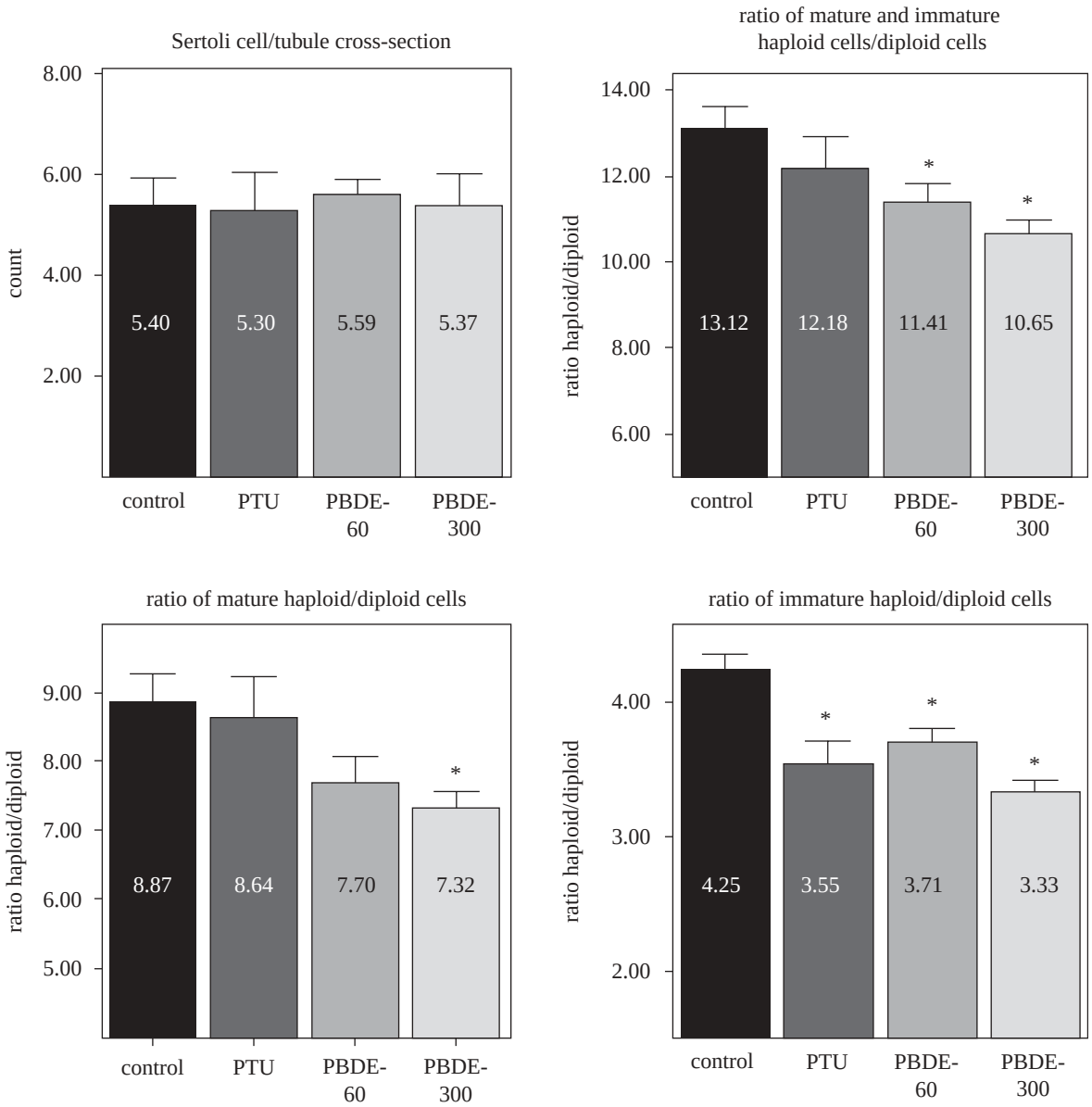

Figure 2. Number of Sertoli cells per tubule cross-section in adult male offspring (F1) on PND 110 and flow cytometry analysis of testicular cell population on the basis of their DNA ploidy in adult rats on PND 140. Gravid female rats were administered per gavage either vehicle (control) or 60 or 300 μg BDE-99 (kg bw)$^{-1}$ (PBDE-60 or PBDE-300) on gd 6. A fourth group received 5 mg 6-*n*-propyl-2-thiouracil (PTU) l^{-1} in drinking water from gd 7–21. ANOVA followed by Dunnett *t*-test, $*p < 0.05$.

which inhibits the release of pituitary gonadotropin and gonadal steroids.

Although less is known about the role of thyroid hormones in the development of the reproductive system, it is believed that they are important for gonad development in both sexes (Cooke *et al.* 2004). Animal studies and *in vitro* studies demonstrate that TBBPA and PBDEs interfere with thyroid hormone action, and the function of both the ovary and the testis has been altered following *in utero* exposure to PBDEs. Thyroid hormones indirectly affect sperm production by regulating the number of Sertoli cells, which act as nurse cells for developing sperm. It should be noted, however, that the new data presented here indicate that the observed reduction in sperm count following gestational BDE-99 exposure was not associated with changes in Sertoli cell number. In the ovary, thyroid hormone receptors are present in granulosa cells, which produce steroids and provide growth factors interacting with the oocyte.

In addition, alterations in sex steroid concentrations or function and LH levels have been reported in animal studies for PBDEs, indicating the potential to adversely affect human reproduction. Two epidemiological studies indicate that exposure may be associated with adverse affects on the testis. Preliminary data from a pilot study performed in Japan suggest an inverse relationship between BDE-153 (2,2′4,4′,5,5′-hexabromodiphenyl ether) serum concentrations and sperm concentration in young Japanese males (Akutsu *et al.* 2008). A larger study relating information regarding *in utero* and early developmental exposure to PBDEs with cryptorchidism involved 86 Danish and Finnish newborn boy–mother pairs. Associations between PBDE contamination of human breast milk and cryptorchidism and

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increased gonadotropin release to support normal testosterone production were observed (Main *et al.* 2007). Although there was no significant difference in the placental concentrations, the sum of PBDEs in breast milk was higher in cryptorchid boys than in controls and was positively correlated with infant serum LH concentration. The authors suggest that the lack of correlation between the PBDE concentrations in the placental and breast milk samples may be that the placenta represents the situation at delivery as one would find in a single blood sample and not the long-term exposure.

Although the exposure time frames are not comparable, an increase in LH was also reported for adult male rats exposed to 60 mg kg^{-1} d^{-1} of DE-71 for 3 days (Stoker *et al.* 2005). As indicated previously, Insl3 and androgen play roles in testicular descent. LH regulates testosterone production and plays a role in the differentiation of Leydig cells, which must reach a certain stage before production of Insl3 occurs (Sadeghian *et al.* 2005). The rate of this maturation process in rats has been reported to be influenced by T3 (Mendis-Handagama *et al.* 2007). Alterations in thyroid hormone profile or changes in the status of the hypothalamo–pituitary–testicular axis could influence testicular descent by impacting Insl3 formation.

(ii) *Neurodevelopment*

Thyroid hormone is crucial for growth and development and in particular, for neurodevelopment. Decreases in foetal and maternal thyroid hormone are known to impact neuropsychological development in humans, and impaired achievement on neuropsychological tests can occur even when maternal hypothyroidism is subclinical (Haddow *et al.* 1999). An evaluation of human thyroid hormone status and PBDE exposure was performed in China. TSH and serum concentrations of PBDEs were found to be significantly increased in subjects living close to an electronic waste site compared with those living 50 km away, suggesting hormone disruption (Yuan *et al.* 2008).

Aberrations in thyroid hormone homeostasis caused by PBDEs and TBBPA raise concerns regarding their potential to influence child neurodevelopment. Animal studies indicate changes in behaviour following exposure to PBDEs during a critical developmental window, which may be mediated through changes in thyroid hormones or direct neurotoxicity.

Data regarding neurotoxicants indicate that there is a continuum of toxic outcomes at low doses, i.e. chronic daily exposures, which do not induce overt clinical symptoms (Grandjean & Landrigan 2006). The authors suggest that neurodevelopmental disorders associated with exposure to known human neurotoxicants and untested chemicals have resulted in a veiled pandemic, incurring significant costs to society because of lowered productivity and reductions in intelligence (Grandjean & Landrigan 2006).

5. GENERAL CONCLUSIONS

Exposure of humans to pharmaceuticals is deliberate, with the intention of achieving a desired effect. Development and testing of medications involves a series of evaluations culminating in human clinical trials before marketing is approved. This is quite different from the situation with chemicals, whose presence in biota and humans is inadvertent. In the field of toxicology, information regarding potential human health effects is mainly derived from experimental studies and, when available, from epidemiological studies. Difficulties are not only encountered with extrapolation from animal models to humans, but epidemiological studies are also thwarted by drawbacks such as controlling for confounding factors. In particular, subjects are exposed to an assortment of chemicals on a daily basis and, often, lack of data regarding the extent of exposure at what may have been the critical time frame. One of the goals of toxicology is to identify effects in animal models with the aim to lower the risks of negatively impacting human health. Implicit in this task is that toxicological data, derived from animal studies indicating a potential for adverse effects, serve as a basis to limit exposure before effects appear or are confirmed in humans. The evidence from animal studies on single exposures to the chemicals discussed here suggests the potential for risk to human health. Moreover, data derived from co-exposure studies support the contention that the assortment of chemicals to which we are exposed on a daily basis increases the likelihood of health effects. The high prevalence of body burdens of these chemicals and simultaneous exposure to a number of substances, in conjunction with the fact that the highest concentrations have been demonstrated in the developing young, a sensitive subpopulation of society, indicate the need to decrease the exposure to these compounds.

We would like to thank Brigitte Woelffel, Zeynep Akkoc and Lars Niemann for their excellent technical assistance. Support for the previously unpublished data on BDE-99 was through a grant from the Umweltbundesamt (Federal Environment Agency, grant No. 29965221/04).

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doi: 10.1098/rstb.2008.0284

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PHILOSOPHICAL TRANSACTIONS OF THE ROYAL SOCIETY B

Phil. Trans. R. Soc. B (2009) **364**, 2027–2045
doi:10.1098/rstb.2008.0284

Transport and release of chemicals from plastics to the environment and to wildlife

Emma L. Teuten[1,2], Jovita M. Saquing[3], Detlef R. U. Knappe[3], Morton A. Barlaz[3], Susanne Jonsson[4], Annika Björn[4], Steven J. Rowland[5], Richard C. Thompson[1], Tamara S. Galloway[6], Rei Yamashita[7], Daisuke Ochi[7], Yutaka Watanuki[7], Charles Moore[8], Pham Hung Viet[9], Touch Seang Tana[10], Maricar Prudente[11], Ruchaya Boonyatumanond[12], Mohamad P. Zakaria[13], Kongsap Akkhavong[14], Yuko Ogata[15], Hisashi Hirai[15], Satoru Iwasa[15], Kaoruko Mizukawa[15], Yuki Hagino[15], Ayako Imamura[15], Mahua Saha[15] and Hideshige Takada[15,*]

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Plastics debris in the marine environment, including resin pellets, fragments and microscopic plastic fragments, contain organic contaminants, including polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons, petroleum hydrocarbons, organochlorine pesticides (2,2′-bis(*p*-chlorophenyl)-1,1,1-trichloroethane, hexachlorinated hexanes), polybrominated diphenylethers, alkylphenols and bisphenol A, at concentrations from sub ng g^{-1} to μg g^{-1}. Some of these compounds are added during plastics manufacture, while others adsorb from the surrounding seawater. Concentrations of hydrophobic contaminants adsorbed on plastics showed distinct spatial variations reflecting global pollution patterns. Model calculations and experimental observations consistently show that polyethylene accumulates more organic contaminants than other plastics such as polypropylene and polyvinyl chloride. Both a mathematical model using equilibrium partitioning and experimental data have demonstrated the transfer of

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Electronic supplementary material is available at http://dx.doi.org/rstb20080284 or via http://rstb.royalsocietypublishing.org.

One contribution of 15 to a Theme Issue 'Plastics, the environment and human health'.

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contaminants from plastic to organisms. A feeding experiment indicated that PCBs could transfer from contaminated plastics to streaked shearwater chicks. Plasticizers, other plastics additives and constitutional monomers also present potential threats in terrestrial environments because they can leach from waste disposal sites into groundwater and/or surface waters. Leaching and degradation of plasticizers and polymers are complex phenomena dependent on environmental conditions in the landfill and the chemical properties of each additive. Bisphenol A concentrations in leachates from municipal waste disposal sites in tropical Asia ranged from sub $\mu g\,l^{-1}$ to $mg\,l^{-1}$ and were correlated with the level of economic development.

Keywords: marine plastic debris; plastic resin pellet; microplastics; landfill leachate; endocrine-disrupting chemicals; persistent organic pollutants

1. INTRODUCTION

Plastics are considered to be biochemically inert materials that do not interact with the endocrine system because of their large molecular size, which prohibits their penetration through the cell membrane. However, plastic debris present in the marine environment (marine plastics) carry chemicals of smaller molecular size (MW < 1000). These chemicals can penetrate into cells, chemically interact with biologically important molecules and may disrupt the endocrine system. Such chemicals are categorized into two groups: (i) hydrophobic chemicals that are adsorbed from surrounding seawater owing to affinity of the chemicals for the hydrophobic surface of the plastics and (ii) additives, monomers and oligomers of the component molecules of the plastics. Many of the contaminants addressed herein have known biological consequences. For example, the plastic constitutional monomer bisphenol A (BPA) and alkylphenol additives exert oestrogenic effects (e.g. Sonnenschein & Soto 1998), while some phthalate plasticizers have been associated with reduced testosterone production (e.g. Foster 2006). A wide range of biological effects have been reported for polychlorinated biphenyls (PCBs; Neal 1985). Reviews of human and wildlife exposure to plastics additives are also available in this volume (Koch & Calafat 2009; Meeker *et al.* 2009; Oehlmann *et al.* 2009).

The objective of this paper is to review the phenomena by which plastics released to the environment serve as carriers of organic contaminants to wildlife. The first two sections describe leaching of contaminants from plastics in landfills. Section 2 reviews the migration and degradation of plasticizers (phthalates), additives (organotin compounds and nonylphenols (NP)) and monomers (BPA), while §3 focuses on landfill leachate as a source of plastics-derived endocrine-disrupting compounds. The following sections address the uptake of contaminants from the environment onto plastics. In §4, sorption is described mathematically and the model validated by experimental observations. Section 5 summarizes the types and quantities of contaminants found sorbed to plastics collected from the marine environment. The remaining sections emphasize plastics as a vector in the transport of contaminants to animals. Section 6 presents an overview of the transfer of plastic-derived contaminants to organisms. This is expanded in §7, which describes literature concerning the transport of contaminants to sediment-dwelling invertebrates. Finally, §8 reports initial experiments demonstrating transfer of contaminants from plastics to higher-trophic-level organisms (acronyms in this paper are listed in table 1).

2. RELEASE AND DEGRADATION OF ADDITIVES AND CONSTITUTIONAL MONOMERS FROM POLYMERS

Organic compounds are used as additives in polymers to improve the properties of the resulting products. Release of the additives to the surrounding environment is an unwanted process for both the manufacturer and the environment, since loss of additives shortens polymer lifetime, e.g. loss of plasticizers lowers the tensile strength of polyvinyl chloride (PVC; Boyer 1951), and living organisms are exposed to the released additives. Phthalates, organotins and BPA, mentioned subsequently, have been shown to target nuclear hormone receptor signalling pathways (Grun & Blumberg 2007). The release may take place during the service life of the plastics or after their disposal, for example in landfills. Both the landfill compartment and other potential receptors such as sediments represent complex environments with multiple chemical and biological processes occurring concurrently.

The migration potential of an additive in a polymer depends on several parameters. The polymer itself has a three-dimensional porous structure in which the additives are dispersed. The pore diameter and the size of the additive are correlated such that smaller (lower molecular weight) additives move more easily through a polymer with bigger pore size. Additives that fit more exactly in the pores have a small but not insignificant capacity to migrate. Therefore, the pore size in the polymer and the size of the additive molecule are important parameters. Co-migration and temperature are positive migration factors as are certain physical–chemical properties of the additive and the surrounding environment. Release of a reactively bonded compound from a polymer requires cleavage of the covalent bond(s) before migration can take place. Therefore, loss of reactively bonded chemicals from the polymer resins is most probably because of release of unreacted constituents (see BPA below).

In landfills, plastics are exposed to an extraction solvent in the form of acidic (pH 5–6) leachates with high ionic strength and neutral or alkaline leachates containing high-molecular-weight organic compounds. The different leachates have not only different potentials to extract and transport, but also different biological populations with the potential to degrade or transform the released additives.

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Table 1. List of acronyms.

acronym	meaning
BD	brominated diphenylether congener
BDEs	brominated diphenylethers
BPA	bisphenol A
CB	chlorinated biphenyl congener
DDD	2,2′-bis(*p*-chlorophenyl)-1,1-dischloroethane
DDE	2,2′-bis(*p*-chlorophenyl)-1,1-dischloroethylene
DDT	2,2′-bis(*p*-chlorophenyl)-1,1-trichloroethane
DDTs	DDT and its metabolites (i.e. DDD and DDE)
DEHP	diethylhexyl phthalate
DMP	dimethyl phthalate
DOC	dissolved organic carbon
E1	estrone
E2	oestradiol
E3	estriol
EDCs	endocrine-disrupting chemicals
EEQ	oestradiol-equivalent concentration
FTIR	Fourier transform infrared spectroscopy
GC-ECD	gas chromatograph equipped with an electron capture detector
GDP	gross domestic products
HCHs	hexachlorocyclohexanes
HDPE	high-density polyethylene
HOCs	hydrophobic organic contaminants
MW	molecular weight
NOEC	no-effect concentration
NP	nonylphenol
OP	octylphenol
PAHs	polycyclic aromatic hydrocarbons
PBDEs	polybrominated diphenylethers
PCBs	polychlorinated biphenyls
PCE	tetrachloroethylene
PE	polyethylene
PVC	polyvinyl chloride
SML	sea-surface microlayer
SOM	sorbent organic matter
T_g	glass transition temperature
TNP	trisnonylphenolphosphites
UV	ultraviolet

Plasticizers, which are the largest group of additives in polymers, range from molecular weights of approximately 200 to almost 700 g mol^{-1} and cover water solubility from g l^{-1} to sub-μg l^{-1}. Phthalates (or more chemically correct, alkyl/aryl esters of 1,2-benzenedicarboxylic acid) are the most common plasticizers and may account for more than 60 per cent of polymers of PVC (Giam *et al.* 1984). Dimethyl phthalate (DMP) is fairly easily released from its resin, as soon as the DMP-containing product is landfilled, owing to its relatively high water solubility, i.e. there is a continuous depletion of DMP from the resin surface, and the negative concentration gradient from the inside to the surface causes the migration. In contrast, the higher-molecular-weight phthalates, such as diethylhexyl phthalate (DEHP), are more resistant to migration owing to their hydrophobicity, which causes less release from the polymer surface to leachate compared with DMP.

The importance of the surrounding medium for the extraction potential can be exemplified by the different degradation phases in a landfill. Acidic pH and high ionic strength of the leachate that surrounds waste materials lower the release potential of organic compounds, which make the initial acidogenic phase in a landfill's development a very poor extraction solvent for water-resistant plasticizers (Bauer *et al.* 1998; figure 1). In contrast, a neutral leachate, as found in landfills in the stable methanogenic phase, containing colloidal humic material, facilitates leaching and transport of non-soluble plasticizers owing to sorption to the dissolved organic carbon (DOC) fraction. Therefore, concentrations of phthalate esters in landfill leachates are highly correlated to the DOC content (Bauer & Herrmann 1998). As a consequence of the depletion of plasticizer from the polymer surface, migration from the inner part of the polymer product is enhanced. However, migration from the inner part to the outer surface seems to slow down and even stop as the polymer reaches its glass transition state (Ejlertsson *et al.* 2003). Then, new release of plasticizers only occurs if the brittle polymer structure fractures to expose new surfaces.

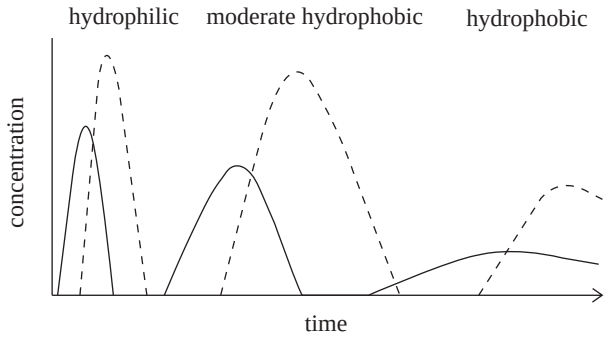

Figure 1. Schematic appearance and concentration of a hydrophilic (left), moderate hydrophobic (middle) and hydrophobic (right) phthalic acid diester (solid lines) and respective monoester (dashed lines) in landfill leachate (modified from Jonsson 2003). The appearance of the diester is correlated to its depletion in the phthalate-containing product.

Degradation of phthalates is initiated by hydrolysis of the ester moiety to phthalic acid and the corresponding alcohols via the monoesters. In landfills, biotic hydrolysis is far more important than abiotic hydrolysis (Furtmann 1996; Staples *et al.* 1997) and takes place (i) at the surface of the original products, (ii) after they have been released from the products and dissolved in the leachate or (iii) following release from another surface to which they adsorbed after leaving the original resin. The most important hydrolysis scenario depends on the water solubility of the phthalate, i.e. the soluble phthalates are probably hydrolysed in the water phase and the hydrophobic phthalates are hydrolysed on to solid surfaces. Hydrolysis is strongly correlated to the methanogenic flora (Jonsson *et al.* 2003*a*, 2006; figure 2). Accumulation of the monoester occurs if the hydrolysis rate of the diester to the monoester is faster than that of the monoester to phthalic acid (Vavilin *et al.* 2005). In fact, phthalate monoester concentrations have been observed at higher concentrations than the corresponding diesters in landfill leachates (Jonsson *et al.* 2003*b*). If the phthalate diester is slowly released during a longer period including the methanogenic stage, the time period when the

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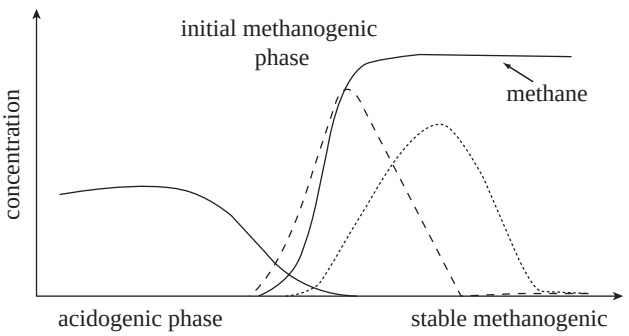

Figure 2. Degradation of a phthalic acid diester (solid line) to its corresponding monoester (dashed line) and phthalic acid (dotted line) in a landfill developing from acidogenic to stable methanogenic phase. Also, the methane production is included (reproduced with permission from Jonsson 2003).

monoester is observed in the leachate is prolonged and the concentrations of the diester are consequently lower.

Organotin compounds are used as stabilizing additives in polymers, such as PVC, and they deserve special attention because of their toxicity such as deterioration of human immune function and endocrine disruption (Batt 2006). The stabilizers are added as high molecular mono- and dialkyltin carboxylates, mercaptides up to 0.54 per cent or, more common, mercaptans or sulphides up to 0.18 per cent, calculated as tin, in the polymer (Murphy *et al.* 2000; Batt 2006). The carboxylates and mercaptides are rapidly hydrolysed to their mono- and dialkyltin species, respectively, when in contact with water (Björn 2007). The alkyltins are also hydrolysed when they act as stabilizers within the polymer and are consequently released from the polymer surface as alkyltin chlorides. As for the phthalates, it seems likely that the main release of organotin compounds from plastic material occurs when a landfill turns methanogenic (Björn *et al.* 2007). It has been shown that the tin stabilizers are co-extracted from the polymer together with the phthalates. Therefore, organotins in flexible polymers are more readily released than from rigid ones. It should be noted that 90 per cent of the tin stabilizers are used in rigid PVC (ESPA 2002). However, at temperatures above the glass transition of the polymer, more organotin compounds are released than at temperatures below this point (Björn *et al.* 2007).

The alkyltin compounds may dealkylate to inorganic tin, methylate or demethylate in the landfill environment. It is likely that the microbial methylation capacity is greater at higher concentrations (more than 500 $\mu g\,Sn\,l^{-1}$), while demethylation occurs at lower tin concentrations (below 100 $\mu g\,Sn\,l^{-1}$; Björn 2007). Formation of tetramethyl tin changes the properties of the tin species radically, since this compound is very volatile.

Alkylphenols can be used as plasticizing additives or as stabilizers when added as derivatives of phosphites (e.g. trisnonylphenolphosphites: TNP). Upon oxidation and hydrolysis, alkylphenol phosphites are hydrolysed to the corresponding alkylphenol and phosphate, for example, TNP is readily oxidized and hydrolysed to NP under ambient conditions (Murata 1999). Since the alkylphenols and the phosphites are additives, the same reasoning can be applied for these compounds as for the phthalates. More precisely, compounds with shorter alkyl chains have higher leaching potential than longer alkyl chain analogues, and methanogenic leachates are more extractive than acidogenic leachates. However, unlike the phthalates, alkylphenol phosphites are only used in concentrations up to 3 per cent, compared with 60 per cent for the phthalates. Degradation studies of the pure alkylphenols performed under landfill conditions are scarce. However, alkylphenols seem to be the ultimate degradation product when alkylphenol ethoxylates are transformed under methanogenic conditions (e.g. Ejlertsson *et al.* 1999), suggesting that no further degradation occurs under anaerobic conditions (Maguire 1999).

Bisphenol A is, in contrast to the aforementioned compounds, mainly used as a building block of polycarbonate plastics, where the alkylphenol *p*-tert-butylphenol is added as a polymerization adjustor, or as a key constituent together with epichlorohydrin of epoxy resins. Also, BPA is used as additive in PVC, printer ink and some other products. Release of BPA under landfill conditions has not been reported as far as we know, but results from an analysis of landfill leachates suggest that the additive or unreacted BPA, owing to its more hydrophilic character, is readily released from its polymer during the early age of a landfill (Asakura *et al.* 2004, i.e. under acidogenic conditions as for the phthalate DMP). This is supported by leaching studies with water-containing acetic acid and ethanol (Kawamura *et al.* 1998), which is expected to mimic acidogenic leachates. Concerning degradation, complete mineralization has only been reported under aerobic conditions (e.g. Zhang *et al.* 2007). Bisphenol A is reported to be preserved in anaerobic sediments (e.g. Ying & Kookana 2003).

3. PHENOLIC ENDOCRINE-DISRUPTING CHEMICALS IN LEACHATES FROM MUNICIPAL WASTE

Considerable amounts of plastics are disposed of in municipal landfills. As indicated earlier, certain additives and monomers can be released from plastic and will consequently be present in landfill leachate. Detection of BPA, phthalates and the alkylphenols NP and octylphenol (OP) in landfill leachate has been reported (Yasuhara *et al.* 1997; Yamamoto *et al.* 2001; Fromme *et al.* 2002; Coors *et al.* 2003; Jonsson *et al.* 2003*c*; Asakura *et al.* 2004; Deng *et al.* 2006). Although BPA concentrations varied depending on waste composition and landfill operation, concentrations of BPA in leachates ranged from ten to ten thousand $\mu g\,l^{-1}$ from sites in the USA (Coors *et al.* 2003), Germany (Fromme *et al.* 2002) and Japan (Asakura *et al.* 2004). These concentrations were much higher than those detected in municipal sewage effluents (approx. 0.01–0.1 $\mu g\,l^{-1}$, Fromme *et al.* 2002; Nakada *et al.* 2004), implying that untreated leachates from the landfills are potentially significant sources of BPA for the aquatic environment. Furthermore, the BPA concentrations in the leachates

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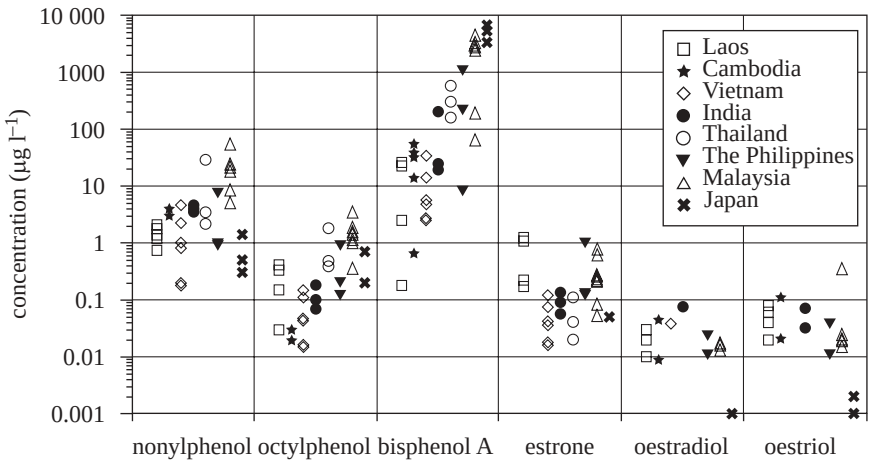

Figure 3. Concentrations of endocrine-disrupting chemicals (EDCs) in leachates from waste disposal sites in Asia.

were up to seven orders of magnitude higher than the no-effect concentration (NOEC) of BPA for endocrine disruption in freshwater organisms (i.e. at 8 ng l^{-1} to induce malformations in female organs of a freshwater snail, *Marisa cornuarietis*; Schulte-Oehlmann *et al.* 2001). Significant concentrations of NP were also detected in landfill leachate sites (Asakura *et al.* 2004). However, the reported concentration ranges of NP (Asakura *et al.* 2004) were similar to those in municipal sewage effluents (Nakada *et al.* 2004).

Economic growth and industrialization bring larger amounts of plastics into society and may increase the amount of plastic waste. To investigate the effect of industrialization on the presence of endocrine-disrupting chemicals (EDCs) in landfill leachates, we measured plastic-derived chemicals in leachates from tropical Asian countries at different stages of economic growth. Leachate samples were collected from open dumps in Malaysia (Kuala Lumpur), Thailand (Bangkok), The Philippines (Manila), Vietnam (Hanoi, Can Tho), Cambodia (Phnom Penh, Angkor), Laos (Vientiane) and India (Kolkata) between 2002 and 2006. At all the sites, municipal wastes, including plastics, are buried. As a reference, leachate samples collected from a landfill site in Japan were also collected and analysed for the EDCs. Details of the analytical procedure were described by Nakada *et al.* (2004, 2006).

Concentrations of the EDCs in the leachates from sites in tropical Asia and Japan are shown in figure 3. Among the EDCs measured, BPA showed highest concentrations in the tropical Asian leachates, ranging from 0.18 to 4300 μg l^{-1}. The highest concentrations were observed in leachates from Malaysia and were comparable to those from Japan. The concentration range of NP in the leachates (0.18–98 μg l^{-1}) was lower than BPA but higher than OP (0.03–3.4 μg l^{-1}). Bisphenol A concentrations were one to five orders of magnitude higher than those in sewage effluents (Nakada *et al.* 2004), whereas NP concentrations in the leachates were one to two orders of magnitude higher than those in effluents. This highlights the importance of the leachate as a source of BPA in aquatic environments. Bisphenol A in leachate could be derived from unreacted monomers in disposed polymers (polycarbonates and epoxy resins), degradation of the polymers and additives. In many landfill sites in industrialized countries, treatment facilities are installed and the environmental burden of these EDCs is reduced. High removal efficiency of BPA has been reported with aerobic treatment (99.3–99.7%, Kawagoshi *et al.* 2003; Asakura *et al.* 2004) and with membrane bioreactors (95.3%, Wintgens *et al.* 2003). However, because of high concentrations of BPA in raw leachates, even treated leachates showed higher BPA concentrations (0.11–30 μg l^{-1}, Wintgens *et al.* 2003; Asakura *et al.* 2004) than the NOEC to freshwater organisms (0.008 μg l^{-1}). Bisphenol A is more problematic in tropical Asian landfill sites with either no, or poorly functioning, leachate treatment facilities. Consequently, high concentrations of BPA were discharged to the surrounding environment (e.g. rivers, groundwater). Notably, BPA concentrations in water samples from a Malaysian pond, into which the leachate from the dump flowed, were an order of magnitude higher (i.e. approx. 11 μg l^{-1}) than in the upstream inflowing river (0.45 μg l^{-1}). This clearly demonstrates that waste-plastic-derived chemicals significantly increase the concentrations of EDCs in the environment.

High concentrations of natural oestrogens (estrone, E1: 0.127–1.00 μg l^{-1}; oestradiol, E2: 0.002–0.0243 μg l^{-1}) were also detected in the leachate from the tropical Asian countries. This is in contrast to leachate from a Japanese landfill site where relatively low concentrations of natural oestrogens (E1: <0.05 μg l^{-1}; E2: <0.008 μg l^{-1}) were detected. The natural oestrogens in the landfill leachates from the Southeast Asian countries could be derived from the disposal of human wastes and/or input from the faeces of scavengers living at the dumping sites. Based on the concentrations of individual EDCs and relative potency of endocrine disruption for individual compounds, oestrogenic activities of the individual compounds have been calculated and compared. The following relative potencies reported by Sumpter & Johnson (2005) for fish were used: NP (0.0025), OP (0.002), BPA (0.0004), E1 (0.3) and E2 (1.00). Oestradiol-equivalent concentrations (EEQ) were calculated by multiplying the concentrations of the individual compounds by their relative potency. The total

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EEQ ranged from 3.4 to 1355 ng-E2 l^{-1}. The highest EEQs were observed in Malaysian, The Philippines and Thai leachates, where much higher concentrations of BPA were observed. In those leachates, BPA accounted for over 50 per cent of the total EEQ. This highlights the importance of BPA in terms of endocrine disruption caused by leachate. The abundance of BPA over natural oestrogens in the leachate contrasts with municipal wastewater effluents where natural oestrogens usually dominate over synthetic chemicals (e.g. Desbrow *et al.* 1998; Nakada *et al.* 2004).

Among the countries investigated, the more industrialized countries (e.g. Malaysia and Thailand) had higher BPA concentrations in landfill leachate than less industrialized countries (e.g. Laos and Cambodia). To quantitatively express this trend, BPA concentrations were plotted against per capita gross domestic products (GDPs; Earth Trends 2007) in figure 4. Bisphenol A concentrations in the leachate show a significant positive correlation with per capita GDP of the tropical Asian countries ($r^2 = 0.66$, $n = 26$, $p < 0.0001$). The most probable reason is that more industrialized countries use larger quantities of plastics resulting in the generation of more plastic waste. This suggests that economic growth in developing countries may increase the environmental prevalence of EDCs unless the leachate is collected and properly treated. To reduce the input of EDCs to the environment, the amount of waste plastics discarded should be decreased through reduction, recycling or other methods of disposal of plastic.

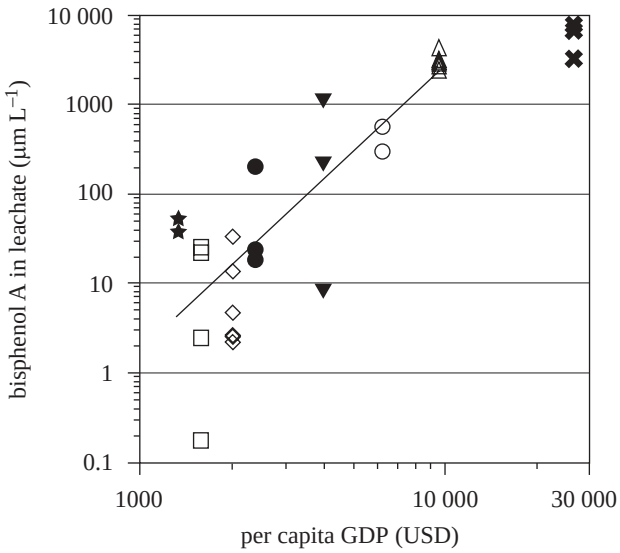

Figure 4. Relationship between BPA concentrations in leachates from waste disposal sites and per capita GDP of Asian countries ($r^2 = 0.66$, $n = 26$). Leachates from municipal waste *disposal* sites in capital and other major cities are plotted. See figure 3 for the symbols of the countries. Data are for countries except Japan.

4. SORPTION AND DESORPTION OF ANTHROPOGENIC CONTAMINANTS FROM PLASTICS

Sorption and desorption are essential fate processes governing the distribution, persistence and ecological impact of hydrophobic anthropogenic contaminants in terrestrial and aquatic systems. Anthropogenic contaminants such as alkylbenzenes, chlorinated hydrocarbons, polycyclic aromatic hydrocarbons (PAHs) and PCBs are examples of compounds that will probably associate with sorbent organic matter (SOM) in the environment. The association of hydrophobic organic contaminants (HOCs) with SOM retards their transport and reduces their availability for biological and chemical transformation. Traditionally, the organic fraction of soils and sediments was considered to be the most important form of SOM in the environment, but recent studies documented the importance of plastics in sediments and debris collected from the marine environment (Colton *et al.* 1974; Mato *et al.* 2001; Ng & Obbard 2006; Rios *et al.* 2007). Hydrophobic organic contaminants were shown to have greater affinity for a range of plastics (polyethylene, polypropylene, PVC) compared with natural sediments (Teuten *et al.* 2007) and were detected on plastic pellets collected from the marine environment (Mato *et al.* 2001; Rios *et al.* 2007) as described in §5.

The extent and rate of HOC (de)sorption are influenced by factors including sorbent (i.e. SOM) properties, sorbate (i.e. HOC) properties, dissolved organic compounds in the aqueous phase, pH and temperature. The following discussion will focus on the effects of sorbent properties on sorption equilibrium and (de)sorption kinetics.

Sorbent organic matter in the environment is composed of organic polymers that contain crystalline and amorphous regions. The crystalline region is characterized by molecules or segments of molecules that are regularly arranged in a crystal lattice. In contrast, the amorphous region has randomly arranged molecules, thus exhibiting a structure that is loose and flexible, and more similar to liquids. Sorption of HOCs generally occurs in the amorphous region, which is characterized on the basis of its internal structure as either glassy or rubbery. Hence, SOM can be envisioned as a mixture of glassy and rubbery polymers. The polymer segments of the glassy phases have higher cohesive forces and are more condensed, whereas those of the rubbery phases exhibit greater mobility and flexibility and can be visualized as a dynamic viscous liquid (Tobolsky & Mark 1980). A particular polymer can transition from the rubbery to the glassy state when it is cooled below its glass transition temperature (T_g). Based on T_g, a polymer is classified as rubbery or glassy at a given environmental temperature. At room temperature, polymers that have a low T_g (e.g. polyethylene has a T_g of $-68°C$) are rubbery, while those that have a high T_g (e.g. PVC has a T_g of $80°C$) are glassy (Brandup *et al.* 1989). The polymer characteristics of the crystalline region lie between those of rubbery and glassy polymers owing to their unique structure. Crystalline polymers are moderately hard, yet flexible and strong (Treloar 1974).

Glassy polymers, because of their rigidity, possess long-lived, closed internal nanoscale pores that can serve as adsorption sites. The existence of amorphous polymer segments and internal nanovoids in glassy polymers results in HOC sorption by linear dissolution (partitioning/absorption) and by nonlinear hole-filling

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(adsorption) mechanisms (Xing & Pignatello 1997). Because of the dual sorption mechanisms, the sorptive behaviour of glassy polymers is normally described by the nonlinear Freundlich model ($q = K_f C_e^n$), where q is the amount of the compound sorbed per unit mass of solid, C_e the aqueous-phase concentration at equilibrium, K_f the Freundlich constant related to the capacity of the sorbent material to sorb the sorbate and n the Freundlich exponent and an indicator of the site energy distribution of a sorbent (i.e. sorbent heterogeneity increases as n decreases from 1; Carter *et al.* 1995). Absorptive partitioning into an organic matrix is characterized by a linear sorption model ($q = K_p C_e$), where K_p is the partition coefficient. Weber *et al.* (1992) showed that nonlinear behaviour may be masked at high aqueous-phase concentrations, but can actually control the overall sorption behaviour at low aqueous-phase concentrations. At low-phase concentrations (<1–1.5% of aqueous solubility), HOCs are sorbed most favourably by regions or components of SOM that have the strongest affinity for that compound (Chiou & Kile 1998). As the high-affinity regions (characterized by nonlinear sorption isotherms) become saturated, HOC sorption is limited to less strongly sorbing regions (characterized by linear sorption isotherms).

(a) *Model description*

Desorption of HOCs from plastics can be described by a one-compartment polymer diffusion model. The model assumes that the HOC desorption rate is limited by diffusion through a single polymer phase, and each sorbent particle is a homogeneous polymeric sphere. Fick's second law of diffusion can be used to express HOC diffusion from plastic particles. In radial coordinates, Fick's second law of diffusion yields equation (4.1):

$$\frac{\partial q}{\partial t} = \frac{D}{r^2}\frac{\partial}{\partial r}\left(r^2 \frac{\partial q}{\partial t}\right), \tag{4.1}$$

where D is the diffusion coefficient (L^2/T), q the solid-phase concentration (sorbed HOC mass/sorbent mass), r the radial position in the sorbent particle (L) and t the time.

To solve equation (4.1), two-dimensionless variables, T and R, are introduced:

$$T = \frac{Dt}{a^2} \quad \text{and} \quad R = \frac{r}{a}, \tag{4.2}$$

where a is the sorbent particle radius.

Therefore, the governing equation is transformed to:

$$\frac{\partial q}{\partial T} = \frac{1}{R^2}\frac{\partial}{\partial R}\left(R^2 \frac{\partial q}{\partial R}\right). \tag{4.3}$$

Initial and boundary conditions specific to the experimental method employed to estimate D are required to solve equation (4.3). For the initial condition, it was assumed that sorption equilibrium was attained prior to initiation of desorption and, therefore, the solid-phase concentration (q_0) was uniform throughout the sorbent at the beginning of desorption tests, i.e.

$$q = q_0 \quad \text{at} \quad T = 0 \quad \text{from} \quad 0 \le R \le 1. \tag{4.4}$$

The first boundary condition requires that symmetry is maintained at the particle centre at all times, i.e.

$$\frac{\partial q}{\partial R} = 0 \quad \text{at} \quad R = 0 \quad \text{and} \quad T \ge 0. \tag{4.5}$$

The second boundary condition specifies the solid-phase concentration at the external solid surface. For the results described here, in which volatile HOCs were tested, sorbents equilibrated with an aqueous phase were sparged continuously during the desorption test. Thus, the aqueous-phase HOC concentration was negligible (i.e. an infinite sink was approximated).

Assuming instantaneous equilibrium between the solid- and aqueous-phase concentrations at the external sorbent surface, the solid-phase concentration at the external particle boundary was therefore also zero, i.e.

$$q = 0 \quad \text{at} \quad R = 1. \tag{4.6}$$

A Crank–Nicholson finite-difference algorithm was developed to solve the one-compartment polymer diffusion model. The Newton–Raphson optimization routine was used to determine the diffusion coefficient (D) such that the mean square error between the model output and experimental data was minimized. The model requires the following input parameters: isotherm parameters (K_p for linear isotherms and K_f and n for nonlinear isotherms), particle radius, particle density, fractional uptake and initial estimate of D.

(b) *Model application*

The validity of the one-compartment polymer diffusion model to simulate desorption kinetics of HOCs in homogeneous plastics was tested using toluene, *o*-xylene and tetrachloroethylene (PCE) as model HOCs, high-density polyethylene (HDPE) as a model rubbery polymer and PVC as a model glassy polymer. As shown in figure 5, model results agreed well with *o*-xylene desorption data from HDPE and PVC. Similarly, model results agreed well with toluene (figure 6) and PCE (not shown) desorption data. Table 2 summarizes inputs to the desorption model and the estimated D values for the tested HOCs.

In general, HOC diffusivities in plastics were higher in HDPE ($D = 10^{-10}\ \mathrm{cm^2\,s^{-1}}$) and lower in PVC ($D = 10^{-13}$–$10^{-14}\ \mathrm{cm^2\,s^{-1}}$). The diffusivity of HOCs in PVC is reasonably consistent with the values observed by Berens (1989). For HDPE, the diffusivity of HOCs is an order of magnitude lower than values reported in literature, possibly because of differences in polymer composition and crystallinity, experimental conditions and uncertainties in estimates of diffusional length (Roger *et al.* 1960; Park *et al.* 1996; Sangam & Rowe, 2001; Joo *et al.* 2004). Typically, HDPE has 80–95 per cent crystallinity (Brandup *et al.* 1989) but the crystallinity of an HDPE geomembrane tested by Sangam & Rowe (2001) was only 47 per cent, which could account for the difference in D estimates. Moreover, uncertainties in diffusional length scale (i.e. film thickness or particle radius) could also affect the

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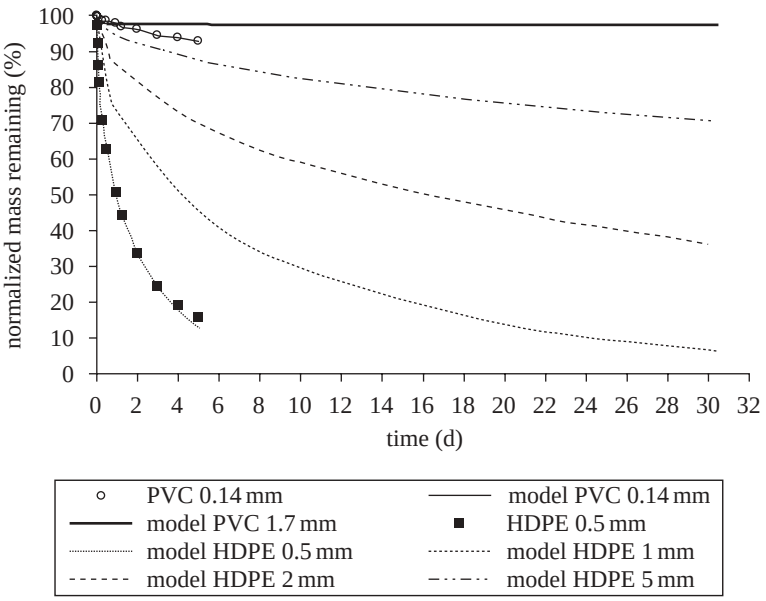

Figure 5. Comparison of *o*-xylene desorption data and one-compartment diffusion model fits as well as predictions of *o*-xylene desorption rates from PVC and HDPE spheres of different diameters. Desorption data were measured after six months of equilibration in ultrapure water.

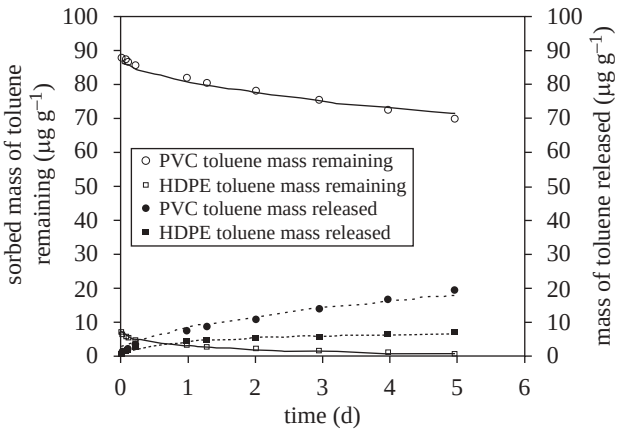

Figure 6. Effect of polymer type on sorbed toluene mass remaining and released per gram of sorbent. Lines represent model fits for sorption equilibrium liquid-phase concentration (C_e) of 100 $\mu g\,l^{-1}$, $K_p = 70.7$ ($\mu g\,kg^{-1}$)($l\,\mu g^{-1}$) for HDPE and $K_f = 1663$ ($\mu g\,kg^{-1}$)($l\,\mu g^{-1}$), and $n = 0.864$ for PVC (Wu *et al.* 2001). The particle diameters of HDPE and PVC were 0.5 and 0.14 mm, respectively.

calculated diffusion coefficients. When the diffusion coefficients are normalized by diffusional length scale (D/a^2), the normalized HOC diffusion coefficients determined for HDPE in this study have the same order of magnitude ($10^{-2}\,d^{-1}$) as those in previous reports (Sangam & Rowe 2001; Joo *et al.* 2004).

The difference in HOC desorption rates observed between HDPE and PVC is consistent with their rubbery and glassy states. The polymeric organic matrix of glassy polymers such as PVC is more rigid than that of rubbery polymers such as HDPE. Because the relaxation speeds of glassy polymers are slow, diffusion of solute molecules into and out of the condensed and highly cross-linked organic matter is slow, which explains the smaller HOC diffusivities in glassy polymers (Brusseau *et al.* 1991; Pignatello & Xing 1996; Huang & Weber 1997, 1998). Moreover, nanovoids within glassy polymer matrices provide strong adsorption sites, and desorption of HOCs from these sites is generally activated (Pignatello & Xing 1996).

For the same input parameters and estimate of D, figure 5 illustrates the effect of diffusional length scale on *o*-xylene desorption rates. The particle sizes, for which model predictions are shown in figure 5, are representative of the size range of plastic pellets and fragments collected from the marine environment (Colton *et al.* 1974; Mato *et al.* 2001; Rios *et al.* 2007). In agreement with the inverse proportionality between desorption rate and the square of the sorbent particle radius, the results in figure 5 illustrate that HOC desorption rates decreased dramatically as the diffusional length scale increased. Additional model predictions showed that the time required for 50 per cent desorption of toluene, *o*-xylene and PCE from rubbery plastics (e.g. polyethylene and polypropylene) with a 1 mm particle diameter was 2.8, 4.0 and 6.2 days, respectively. When the particle diameter was doubled (2 mm), the half-lives increased to 11.3, 16.1 and 25.3 days, respectively. For glassy plastics (e.g. PVC and polystyrene) with a particle diameter of 0.2 mm, the time required for 50 per cent desorption of toluene, *o*-xylene and PCE was 85 days, 2.3 years and 6.5 years, respectively. The predicted half-life of *o*-xylene and PCE in PVC was >100 years when the particle diameter of glassy polymers was increased to 1.7 mm.

Figure 6 compares the mass of toluene released per unit mass of HDPE and PVC, assuming $C_e = 100\,\mu g\,l^{-1}$ and using a particle diameter of 0.5 mm for HDPE and 0.14 mm for PVC. Although toluene diffuses three orders of magnitude faster in HDPE, the amount of toluene released from PVC is greater than that released from HDPE. This is because the mass of toluene sorbed to PVC at equilibrium is greater than the mass of toluene sorbed to HDPE (Wu *et al.* 2001).

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Table 2. Model input parameters and estimates of toluene, *o*-xylene and PCE diffusivities in HDPE and PVC.

			isotherm parameters						diffusion coefficient[a]		
			toluene[b]		*o*-xylene[b]		PCE[c]		toluene	*o*-xylene	PCE
material	particle density ($g\ cm^{-3}$)	mean particle diameter (μm)	K	n[d]	K	n[d]	K	n[d]	D	D	D
HDPE	0.962	500	70.7[e]	1.0	244.1	1.0	385	1.0	3.13×10^{-10}	2.20×10^{-10}	1.42×10^{-10}
PVC	1.4	140	1663[f]	0.864	4634	0.718	2951	0.918	4.33×10^{-13}	4.22×10^{-14}	1.50×10^{-14}

[a]$cm^2\ s^{-1}$.
[b]Values from Wu *et al.* (2001).
[c]Values from Wagner (2003).
[d]Dimensionless Freundlich exponent.
[e]K_p for HDPE ($\mu g\ kg^{-1}$)($l\ \mu g^{-1}$).
[f]K_f for PVC ($\mu g\ kg^{-1}$)($l\ \mu g^{-1}$)^n.

In summary, results from alkylbenzene and PCE desorption kinetic tests for glassy and rubbery polymers suggest that both sorbent and sorbate properties strongly influence HOC sorption uptake and desorption kinetics. Glassy polymers exhibit larger HOC sorption capacities and slower HOC release rates than rubbery polymers. Moreover, the size of the plastic pellet or fragment strongly affects the rate at which sorbed HOCs are released.

5. TYPES OF CONTAMINANTS DETECTED IN MARINE PLASTICS

(a) *Adsorption of contaminants to marine plastics from surrounding seawater*

Carpenter *et al.* (1972) first detected PCBs in polystyrene spherules collected from Niantic Bay (north-eastern Long Island Sound, USA). Although they suggested that the PCBs were adsorbed onto the plastic from the surrounding seawater, no supporting evidence was provided. After a 30 years break, Mato *et al.* (2001) triggered a series of systematic studies on toxic chemicals in marine plastics. They detected PCBs in polypropylene pellets from Japanese coasts with concentrations ranging from 4 to 117 ng g^{-1}. They conducted a field adsorption experiment using virgin polypropylene pellets and demonstrated a significant and consistent increase in PCB concentrations throughout the 6 day experiment. This indicated that the source of the PCBs was ambient seawater and that adsorption to the pellet surfaces was the mechanism of enrichment. In another adsorption experiment, Mato *et al.* (2002) subjected polyethylene and polypropylene pellets to seawater for two weeks and found that polyethylene pellets adsorbed four times more PCBs than polypropylene pellets, indicating that polyethylene has higher affinity for hydrophobic compounds. This is consistent with field observation and experimental work described later and literature (e.g. Karapanagioti & Klontza 2008). Comparison of PCB concentrations in marine plastic resin pellets with those in seawater suggested their high degree of accumulation (apparent adsorption coefficient of 10^5–10^6).

Subsequently, Endo *et al.* (2005) conducted a detailed study of PCBs in beached resin pellets. They analysed PCB concentrations in individual pellets and observed a large (i.e. two orders of magnitude) piece-to-piece variation in PCB concentrations among the pellets. Variation in PCB concentrations may be caused by various factors including difference in materials, weathering and residence time in the sea. Endo *et al.* (2005) found that polyethylene pellets tended to have higher concentrations of PCBs than polypropylene, consistent with the results of the above adsorption experiment. They also indicated that yellowing and/or fouled pellets had higher concentrations of PCBs. Yellowing is derived from oxidation of phenolic additives to quinone-type degradation products and, therefore, is an indication of environmental residence time of the pellets. Based on their findings, they proposed a monitoring methodology 'pellet watch' where beached yellowed and/or fouled polyethylene pellets are used to monitor coastal pollution by hydrophobic chemicals.

Based on the results, the monitoring was expanded to a global scale, named *International Pellet Watch* (http://www.tuat.ac.jp/~gaia/ipw/index.html). In this scheme, plastic resin pellets are collected on beaches by local volunteers and sent to the Tokyo University of Agriculture and Technology for analysis of a variety of HOCs. Through the activities of volunteers worldwide, and a network of scientists, 27 samples from 16 countries have been analysed. As shown in figure 7, PCB concentrations of beached plastic resin pellets were highest on the coasts of USA (San Francisco, Los Angeles and Boston), followed by Japan and Europe (The Netherlands, UK and Italy). In tropical Asia, Australia and southern Africa, PCB concentrations were much lower. This regional pattern reflects a difference in PCB usage, with larger amounts of PCBs used in the USA, western Europe and Japan, and minimal usage in tropical Asia, Australia and southern Africa. For example, more than half of the total global production of PCBs was used in the USA (Erickson 1997). Discharged PCBs have accumulated in coastal zones, particularly in sediments, which are likely to be resuspended into the water column. Correspondingly, higher concentrations of PCBs are still found in sediments, seawater and marine biota in the USA, western Europe and Japan. To examine the feasibility of a monitoring methodology employing plastic pellets, concentrations of PCBs in the pellets were compared with those in conventional biomonitoring organisms, i.e. mussels (green mussels, *Perna viridis*; blue mussels, *Mytilus galloprovincialis*) in the corresponding zones. Polychlorinated biphenyl concentrations in the beached resin pellets were highly

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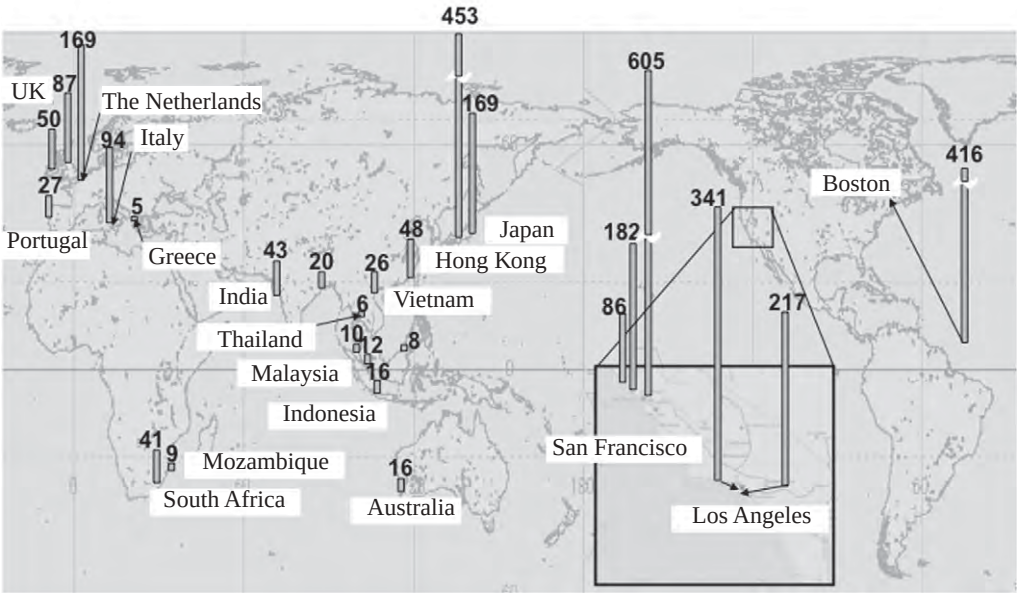

Figure 7. Concentrations of PCBs (ng g^{-1} pellet) in beached plastic pellets. Polychlorinated biphenyl concentration = sum of concentrations of CB nos 66, 101, 110, 149, 118, 105, 153, 138, 128, 187, 180, 170, 206.

correlated with those in mussels ($R^2 = 0.87$). These data clearly demonstrate that beached resin pellets can be used to monitor pollution by hydrophobic chemicals on a global scale.

The hydrophobic surfaces of the resin pellets accumulate other chemicals in addition to PCBs. So far, 2,2′-bis(*p*-chlorophenyl)-1,1,1-trichloroethane (DDT) and its metabolites (DDE and DDD), hexachlorinated hexanes (HCHs), PAHs and hopanes have been detected in beached resin pellets. Interesting, regional distributions were observed. For example, higher concentrations of HCHs and DDTs were detected in South Africa and northern Vietnam, respectively, probably because of the current use of these chemicals as pesticides in these areas. These regional differences in contaminant concentrations in the resin pellets imply that ecological risks associated with the contaminants adsorbed to marine plastics will also vary among the areas.

Considering potential effects on the marine ecosystem, plastic fragments, including microscopic fragments, are possibly more important than resin pellets because fragments are more abundant among marine plastic debris (e.g. Moore *et al.* 2001; McDermid & McMullen 2004; see also Barnes *et al.* 2009; Ryan *et al.* 2009). Similar to resin pellets, hydrophobic contaminants sorb to marine plastic fragments. Figure 8 shows the concentrations of various pollutants detected in plastic fragments collected from a beach near Tokyo, Japan and from the North Central Pacific Gyre (approx.1000 km off the west coast of the USA). The latter were floating plastic fragments and collected by a neuston net. The plastic fragments were sorted, and polyethylene fragments with various shapes and sizes ranging from 1 × 10 × 20 to 31 × 35 × 35 mm were extracted with dichloromethane by Soxhlet for analysis. To evaluate variability, several pools were analysed for each beach. As observed with the resin pellets, PCBs, DDE and PAHs were detected in plastic fragments from both areas.

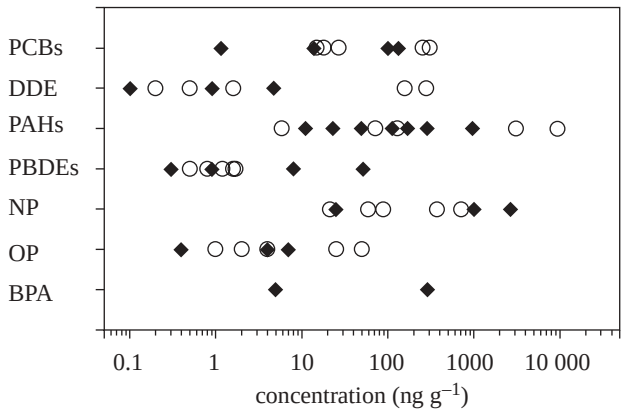

Figure 8. Concentrations of organic contaminants in marine plastic debris (fragments). Solid diamond: The North Pacific Central Gyre; open circle: Japanese coast of the Pacific Ocean. Polychlorinated biphenyls: sum of concentrations of CB nos 66, 101, 110, 149, 118, 105, 153, 138, 128, 187, 180, 170, 206; DDE: concentration of *p*, *p*′-dichlorodiphenyl dichloroethene; PAHs: sum of concentrations of phenanthrene, anthracene, methylphenanthrenes (substitution position: 3, 2, 9, 1), fluoranthene, pyrene, benz[a]anthracene, chrysene, benzo[b]fluoranthene, benzo[j]fluoranthene, benzo[k]fluoranthene, benzo[e]pyrene, benzo[a]pyrene, indeno[1,2,3-cd]pyrene, benzo[ghi]perylene and coronene; PBDEs: concentration of BDE nos 3, 7, 15, 17, 28, 71, 49, 47, 66, 77, 100, 119, 99, 85, 126, 154, 153, 138, 183; NP: concentration of nonylphenols; OP: concentration of octylphenol; BPA: concentration of bisphenol A.

Concentrations of contaminants in polyethylene fragments were higher on the Japanese coast (PCBs: 12–254 ng g^{-1}; DDE: 0.2–276 ng g^{-1}; PAHs: <60–9370 ng g^{-1}) than those in the Central Gyre (PCBs: 1–23 ng g^{-1}; DDE: 0.1–4.7 ng g^{-1}; PAHs: <100–959 ng g^{-1}). This difference can be explained by regional differences in seawater concentrations of the contaminants (i.e. coast versus open ocean), since these compounds enter the aquatic environment

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predominantly from terrestrial runoff. Rios *et al.* (2007) reported similar concentrations of PCBs, DDE and PAHs in marine plastics (both resin pellets and fragments) collected from a wide variety of Pacific Ocean locations including the North Pacific Gyre, California and Hawaii. They also detected DDT, DDD, HCHs and *n*-alkanes.

(b) *Additive-derived chemicals*

Plastics additives have also been measured on marine plastic debris. Nonylphenol, which forms from TNP degradation (§2), and is also present as an impurity in TNP (Gilbert *et al.* 1986), was detected in plastic resin pellets collected from 12 Japanese coasts and 7 Malaysian coasts (Mato *et al.* 2002). Nonylphenol concentrations ranged from 18 to 17 000 ng g^{-1}. Interestingly, no difference in NP concentrations was observed between Japanese and Malaysian coasts, although PCB concentrations in the pellets from Japanese coasts were much higher than those from the Malaysian coast, probably because of the difference in pollution levels of PCBs in the coastal waters. Polypropylene pellets tended to have higher amounts of NP than polyethylene pellets (Mato *et al.* 2002), which is consistent with a higher prevalence of additives in polypropylene pellets.

Since additives are also compounded into plastics during moulding, fragments of post-consumer plastics may have more additive-derived chemicals than resin pellets. Nonylphenol, OP and BPA were detected in marine plastic fragments, as shown in figure 8. Interestingly, higher concentrations of NP (24.9–2660 μg g^{-1}) were found in the polyethylene fragments from the open ocean (Central Gyre) than the Japanese coast, where higher concentrations of hydrophobic contaminants such as PCBs and PAHs were found. It is therefore likely that these NPs are derived from additives rather than adsorption from surrounding seawater. Significant concentrations of BPA (5–284 ng g^{-1}) were also detected in the plastic samples from the Central Gyre, whereas no significant BPA was detected in plastic fragments from the Japanese coast ($<$25 ng g^{-1}). Because only polyethylene plastic was sorted and analysed (i.e. no polycarbonate and epoxy resin was included), BPA detected in the marine plastic fragments is also likely to have been derived from an additive. Another important category of additive includes flame retardants such as polybrominated diphenyl ethers (PBDEs). Significant concentrations of PBDEs consisting of monobrominated to heptabrominated diphenyl ethers were detected in the plastic fragments both from the Japanese coast and the Central Gyre. Concentrations of PBDEs (sum of BDEs ranging from three to seven bromines) were 0.9–2.1 ng g^{-1} for the Japanese coast and 0.4–57 ng g^{-1} for Central Gyre. Since PBDEs are ubiquitous in coastal seawater and are hydrophobic, those on the plastic fragments may have sorbed from surrounding seawater. Alternatively, they may have been added to the plastics. In the plastic fragment samples from the Central Gyre, BD no. 183 was predominant. Since BD no. 183 is a major component of Octa-BDE, which is a flame retardant currently used in USA and not predominant in US coastal waters, this congener in the plastic fragment from the Central Gyre most probably originated from the flame retardant added to plastics. Polybrominated diphenyl ethers are suspected thyroid disruptors in wildlife and humans (WHO/IPCS 1994). Therefore, more research is needed to identify the sources of PBDEs in the plastic fragments.

6. TRANSFER OF PLASTIC-DERIVED CONTAMINANTS TO ORGANISMS

Uptake of contaminants by organisms occurs by a variety of pathways, most commonly inhalation, dermal sorption and ingestion. Contaminant transfer to organisms from plastics may occur by any of these routes, and the major transport route will vary according to the organism and the physico-chemical properties of the contaminant. For most species, the predominant route of transfer of contaminants from plastics is likely to occur via plastic ingestion.

More than 180 species of animals have been documented to ingest plastic debris, including birds, fish, turtles and marine mammals (Laist 1997). Small plastic pieces floating on the ocean surface are mistaken for food by both fish and birds, while turtles eat suspended plastic bags, which they may mistake for jellyfish. To some extent, plastic ingestion by marine mammals may also be indirect, occurring by ingestion of fish that have eaten plastic (Eriksson & Burton 2003). In addition, plankton trawls recover substantial quantities of plastic (Moore *et al.* 2001), and plankton-feeding mammals such as mysticetes (baleen whales) are also at risk of plastic ingestion (see Gregory 2009). Temporal studies suggest that plastic ingested by birds has been increasing since the 1960s (Robards *et al.* 1995; Vlietstra & Parga 2002; Ryan *et al.* 2009). Since few of these ingestion studies date later than the early 1990s, it is difficult to assess the effect of the enactment of later legislation such as MARPOL Annex V, which came into effect in 1989 and is intended to restrict further the disposal of plastics at sea (Derraik 2002). However, Cadée (2002) reported that 80 per cent of plastic debris on the Dutch coast had peck marks made by birds, indicating widespread sampling of plastic by birds. Ingestion of plastics has many detrimental consequences, including gastrointestinal blockages (Baird & Hooker 2000), ulceration (Fry *et al.* 1987), internal perforation and death (Mascarenhas *et al.* 2004).

A lesser-studied problem is the transfer of contaminants from ingested plastics to organisms. Sections 2 and 5 detail the presence of a selection of organic and organometallic contaminants in plastics. In addition, a number of platinum group metals have demonstrated a high affinity for plastics (Cobelo-Garcia *et al.* 2007). This suggests a high potential for plastics to transport such chemicals. The nature and environmental significance of this transport are then partly determined by the subsequent fate of the plastic. Sorption of contaminants to plastics may also inhibit contaminant biodegradation. For example, microbial degradation of phenanthrene has been shown to be reduced by a factor of 6 when associated with

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polyethylene (Hatzinger & Alexander 1997). Thus, plastics not only have the potential to transport contaminants, but they may also increase their environmental persistence. Ingestion of plastics with sorbed contaminants has been suggested as a possible exposure route for contaminants (Fry *et al.* 1987; Ryan *et al.* 1988; Mato *et al.* 2001; Thompson *et al.* 2004). In support of this hypothesis, the quantity of the contaminant desorbed from the plastic was greatly enhanced by the presence of surfactants and organic matter (Sakai *et al.* 2000), suggesting that increased leaching of contaminants will occur under gastric conditions. Furthermore, acidic gastric conditions may enhance desorption of metals bound to plastics. Finally, a positive correlation has been observed between the mass of ingested plastic and the PCB concentration in the fat tissue of birds (great shearwaters; *Puffinus gravis*) (Ryan *et al.* 1988). Since plastics are known to accumulate PCBs in the environment (§5), this correlation supports plastic-mediated transfer of contaminants to organisms.

7. TRANSFER OF CONTAMINANTS FROM PLASTICS TO LOWER TROPHIC ORGANISMS

Microscopic plastic debris as small as 20 μm (Thompson *et al.* 2004) also litters the global environment (Thompson *et al.* 2004; Ng & Obbard 2006; Reddy *et al.* 2006). This material enters the environment both directly from use as 'scrubbers' in cleaning products and as abrasive beads for cleaning ships, and indirectly from deterioration of brittle, weathered macroscopic plastics (Derraik 2002). To investigate the possibility for microscopic plastics to transport contaminants to benthic organisms, lugworms (*Arenicola marina*) were exposed to contaminant-sorbed plastics. Phenanthrene (a PAH), tetrabromodiphenyl ether (a PBDE), triclosan (antimicrobial) and NP were sorbed to microscopic PVC particles at environmentally relevant concentrations. Lugworms were exposed to sediment containing these contaminated plastics (5% w/w) for 10 days. At the end of the trial, contaminant concentrations in the lugworm tissue were significantly higher than that in the sediment. While this demonstrated that contaminants can be transferred from plastics to benthic organisms (R. C. Thompson 2004–2007, unpublished data), the mechanism remains unclear. Some plastic debris is comparable in size to sediment and suspended particulate matter and is ingested by invertebrates with varying feeding patterns. Sediment-ingesting lugworms, filter-feeding barnacles (*Semibalanus balanoides*) and amphipods (*Orchestia gammarellus*), which eat decaying organic matter, have all been shown to ingest plastic fragments (Thompson *et al.* 2004). Therefore, ingestion of contaminant-sorbed plastics may be a route of contaminant transport to these organisms. Indeed, digestive surfactants isolated from deposit feeders have been shown to enhance desorption of pollutants including PCBs and PAHs (Voparil & Mayer 2000; Ahrens *et al.* 2001).

Perhaps more important is the potential for invertebrates to uptake contaminants passively, as described by equilibrium partitioning. Contaminants accumulate in the tissues of some organisms as a consequence of the organism reaching equilibrium with their surrounding environment. Determination of the equilibrium concentration of a contaminant between the animal lipids and their environment (e.g. the surrounding water, sediment, soil) is an effective method for estimating contaminant burden in biota (Di Toro *et al.* 1991; European Chemicals Bureau 2003; Reichenberg & Mayer 2006). This method has been widely applied to explain accumulation of both organic and inorganic contaminants in sediment and soil-dwelling organisms (European Chemicals Bureau 2003). Uptake probably occurs through both the skin and the gut, but ultimately the organism reaches equilibrium with its environment.

Using the equilibrium partitioning model, the effect of contaminant uptake from plastic into the lugworm was examined (Teuten *et al.* 2007). Distribution coefficients (K_d) were measured *in vitro* for sorption of phenanthrene to a selection of sediments and plastics including polyethylene, polypropylene and PVC from seawater. Unsurprisingly perhaps, the measured K_d values were up to three orders of magnitude higher for sorption to plastic than to sediment, with uptake onto polyethylene exceeding that onto the other plastics by an order of magnitude (Teuten *et al.* 2007). These measurements were then used to estimate the amount of phenanthrene expected to accumulate in lugworm tissue in the presence of plastic. Addition of *clean* polyethylene to sandy sediment, at plastic concentrations reported in the environment, was predicted to *decrease* the amount of phenanthrene in lugworm tissue by 13 per cent, compared with accumulation in the absence of plastic (Teuten *et al.* 2007). This can be viewed as 'scavenging' of the phenanthrene by the highly sorbent plastic, which reduces the equilibrium concentrations in the other phases, including the organism. However, owing to the high uptake of contaminants onto plastics and the longevity of plastics in the environment, it is unlikely that plastic debris will remain 'clean' for any extended period of time. Hence any predicted beneficial effect of plastics in reducing contaminant concentrations in benthic organisms will likely be short-lived, if it operates at all (Teuten *et al.* 2007). Many plastics float in the sea-surface microlayer (SML), where contaminant concentrations can be highly enriched over concentrations in the bulk water column (Wurl & Obbard 2004). Further fouling of floating plastics causes them to become negatively buoyant (Ye & Andrady 1991), which may result in sedimentation, and thus act as a transporter of contaminants into offshore as well as strandline sediments.

Equilibrium partitioning was also used to examine the effect of scavenging of phenanthrene in the SML, followed by deposition of the contaminated plastics into sediments, on subsequent phenanthrene accumulation by lugworms living in the sediment. Figure 9 displays a schematic in which this transfer mechanism is compared with direct addition of clean plastic to the sediment, as described earlier. The calculations showed that only a small quantity of plastic would be required to sorb a significant amount of phenanthrene from the SML and transport it into the sediment. This would increase total phenanthrene in the benthic

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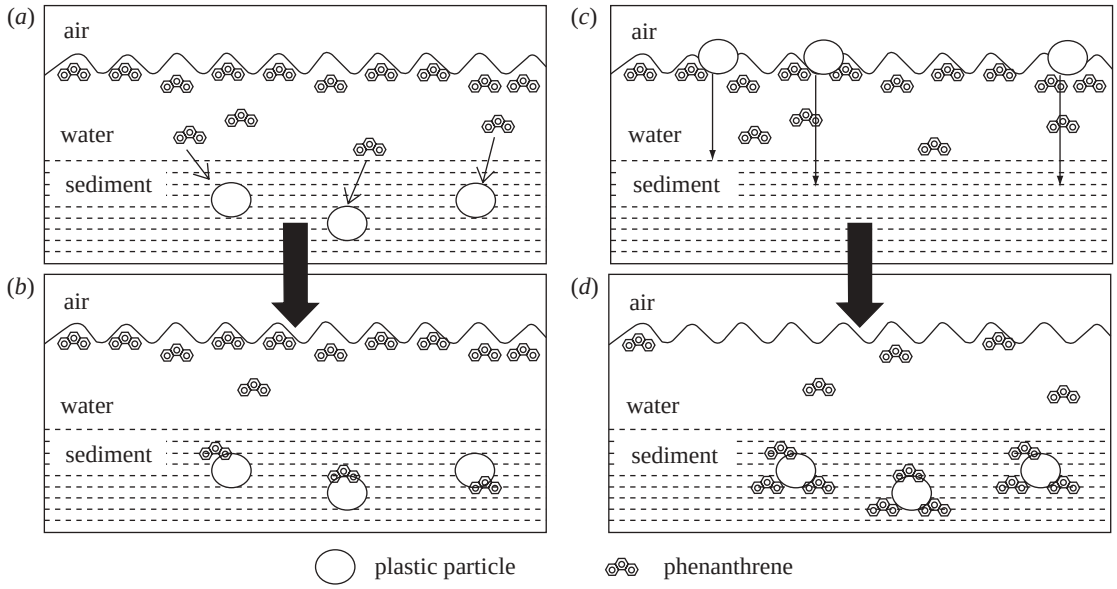

Figure 9. Schematic illustrating the additional effects of plastics in the transport of phenanthrene: (*a*) sorption of phenanthrene to clean plastic in sediment resulting in (*b*) subsequent accumulation of phenanthrene in the sediment, compared with (*c*) sorption of phenanthrene to plastic in the SML and subsequent sinking resulting in (*d*) accumulation of phenanthrene in the sediment. Note that (*c*)→(*d*) results in higher sediment phenanthrene concentrations than (*a*)→(*b*). Although not shown in the schematic, sorption to the sediment also occurs (reproduced with permission from Teuten *et al.* 2007). Copyright 2007 (American Chemical Society).

environment and was predicted to result in higher lugworm tissue concentrations of phenanthrene. Compared with plastic-free sediment, addition of only 1 ppm of phenanthrene-contaminated polyethylene to sandy sediment was predicted to increase the phenanthrene tissue concentration of the worms by 80 per cent. This, however, has yet to be verified by *in vivo* experiment.

Little is known about the effect of modification of plastics by 'weathering' on the sorption of contaminants to plastics, but photo-oxidation is likely to be important as photolytic reactions are known to modify plastic surfaces (Neidlinger & Schissel 1987). For example, polymer chain scission ultimately leads to cracking, thus increasing the surface area of the plastics (Neidlinger & Schissel 1987). This may then increase uptake of contaminants. Reaction with oxygen increases the surface polarity of some plastics (Neidlinger & Schissel 1987), decreasing affinity for hydrophobic contaminants. The results of a series of experiments, complementing the above work and designed to investigate the effect of photo-oxidation on sorption of contaminants to polymers, are illustrated in figure 9.

Granules (150–200 μm) of three high-production plastics (polyethylene, polypropylene and PVC) were artificially weathered using a filtered xenon lamp (Heraeus Suntest CPS), while maintaining the temperature at 25 ± 3°C in a glycerol bath connected to a Coolflow CFT-33 refrigerated recirculator. The spectrum of this lamp is similar to that of natural sunlight (Ali *et al.* 1995). The plastics were agitated daily to maximize exposure to radiation from the lamp. Calibration of the lamp by optometry indicated that 1.08 h of exposure time was equivalent to 1 day of Florida sunlight (30°N, 12 h) (West 2007). Plastics were exposed to the lamp for 7 and 16 days. This was determined to be equivalent to 208 and 460 days of exposure to sunlight at 30°N. The plastics did not appear visibly altered by exposure to the lamp. Aged polypropylene showed an additional absorption at approximately 1750 cm^{-1} as determined by Fourier transform infrared spectroscopy (FTIR), characteristic of the presence of a carbonyl group (C = O), presumably because of oxidation. A similar absorption was faintly visible in the FTIR spectrum of polyethylene aged for 16 days. Distribution coefficients (K_d) were determined for sorption of phenanthrene to these artificially weathered polymers, and tissue concentrations in lugworms were estimated using equilibrium partitioning, as described previously (Teuten *et al.* 2007).

Figure 10 shows the measured K_d for sorption of phenanthrene to plastics from seawater and the predicted amount of contaminated plastic required in sediment with 0.2 per cent organic carbon to give an 80 per cent increase in lugworm tissue, compared with plastic-free sediment. For polyethylene, a clear and statistically significant increase in K_d with length of exposure to light was observed (figure 10). The results for the other plastics were quite variable. However, in all cases K_d is lower for sorption to virgin plastic than to plastic exposed to the xenon lamp for 16 days, so that less of the weathered plastic would be required to transport phenanthrene to lugworms. The most likely explanation for the variation is that two competing surface changes occur on photo-oxidation, giving opposite effects. An increase in surface area will increase K_d, while an increase in surface oxidation might decrease K_d for hydrophobic organic compounds. The increase in oxidation is particularly evident for exposed polypropylene, for which FTIR analysis indicated the appearance of a carbonyl group arising from reaction of oxygen with

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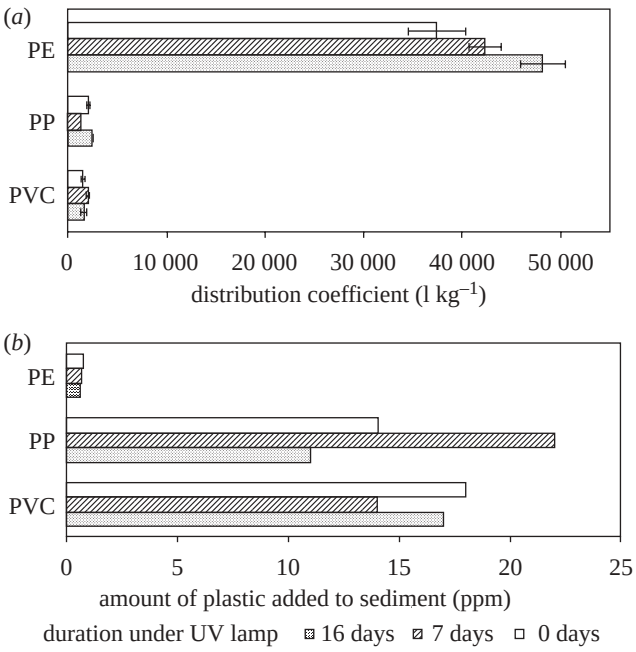

Figure 10. (*a*) Distribution coefficients (K_d) for sorption of phenanthrene to UV-treated plastics from seawater. (*b*) Amount of phenanthrene-sorbed plastic required in sediment (0.2% organic carbon) to increase lugworm tissue concentration by 80%, compared with plastic-free sediment, predicted using equilibrium partitioning as described previously (Teuten *et al.* 2007). Plastics were exposed to a UV lamp for 9 and 16 days, equivalent to 208 and 460 days in natural sunlight. Note that plastic concentrations in sediment are well below the maximum reported amount of 81 ppm (Reddy *et al.* 2006).

the polymer. This correlated with a decrease in K_d after 7 days light exposure, indicating a reduction in the amount of phenanthrene sorbed to polypropylene. Accurate determination of the surface area of the ultraviolet (UV)-exposed polymers was not possible with the amounts used. Further work involving light treatment would be advantageous. These variable results are consistent with work by Endo *et al.* (2005) who observed no significant relationship between PCB concentrations in beached plastic resin pellets and their carbonyl index.

The discussion mentioned earlier illustrates that transfer of contaminants from plastics to biota is a complex phenomenon relying on a variety of processes. For many lower-trophic-level organisms, the extent of contaminant uptake from plastics will be determined by equilibrium partitioning with the surrounding environment. This may be facilitated by active desorption of the contaminants *in vivo*, for example, by solubilization in digestive fluids. The nature and the history of the plastic will also be important in governing transfer of contaminants to organisms.

8. TRANSFER OF CONTAMINANTS TO HIGHER-TROPHIC-LEVEL ORGANISMS AND BIOMAGNIFICATION

In the previous sections, we have demonstrated that chemicals adsorbed to the plastics may be released to digestive fluid and can be transferred to the tissue of organisms. However, the situation for higher-trophic-level organisms is more complex because of 'biomagnification'. Tissue concentrations of hydrophobic and poorly metabolizable contaminants, such as PCBs, are amplified through the food web. Higher-trophic-level organisms (e.g. seabirds) are exposed to highly enriched concentrations of hydrophobic contaminants via their prey (e.g. fish). Therefore, ingested plastics (i.e. anthropogenic prey) compete with the natural prey in terms of contaminant burden to the predator. To assess the potential hazard of ingested plastic-derived chemicals to the predator, chemicals present in the natural prey should be considered. This section reports the preliminary results of a field experiment in which seabirds were fed with plastic resin pellets to examine the transfer of contaminants from the plastic to the seabird.

Chicks of streaked shearwater (*Calonectris leucomelas*) were fed with natural fish and kept in cages located at Mikurajima Island, a natural breeding ground of the bird. Eight 40-day-old chicks were used for the experiment. Among them, five individuals were fed with polyethylene resin pellets collected from Kasai seaside park in Tokyo Bay that contained significant amounts of PCBs as described later. Forty pellets (approx. 1 g) were mixed with the natural prey (Japanese sand lance: *Ammodytes personatus*) at the beginning of the experiment (day 0) and fed to each of the five chicks. Some of the pellets were not fed to the birds and were stored for PCB determination. As a control, three individuals were fed with the Japanese sand lance without plastic resin pellets. For both plastic-feeding and control settings, approximately 10–120 g wet of the fish was fed to each chick daily. Some of the fish were stored in a freezer until the PCB measurement. The experiment continued for 42 days and every 7 days preen gland oil, which is secreted from the preen gland located at the base of the tail feathers, was collected from the live chicks. Polychlorinated biphenyls in the resin pellets, the fish and the preen gland oil were analysed through solvent-extraction, chromatographic purification and a gas chromatograph equipped with an electron capture detector (GC-ECD). Twenty-four PCB congeners (IUPAC numbers 8/5, 18, 28, 52, 44, 66/95, 90/101, 110/77, 118, 132/153, 105, 138/160, 187, 128, 180, 170/190 and 206) were identified and quantified. The total concentration of these congeners is expressed as total PCB in the present study. The details of the analytical procedure for the plastic pellets and fish were described in Endo *et al.* (2005) and for preen gland oil in Yamashita *et al.* (2007).

In the pellets from Tokyo Bay, PCBs were detected at concentrations ranging from 51 to 562 ng g^{-1}, with a mean of 97 ng g^{-1}. Based on the number of pellets fed to the streaked shearwaters, each chick is estimated to have been exposed to approximately 100 ng of PCBs. Significant concentrations of PCBs were also detected in the fish fed to the chicks, because these fish bioconcentrate PCBs through their prey (e.g. copepods). Polychlorinated biphenyl concentrations in the fish ranged from 0.298 to 0.706 ng g^{-1} wet (mean: 0.451 ng g^{-1} wet). The PCBs in the fish should be compared with those in the pellets to

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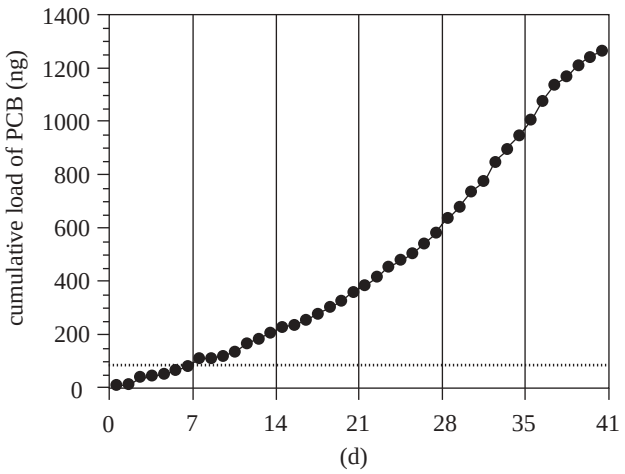

Figure 11. Loads of total PCBs in chicks. Closed circles: cumulative load from fish (Japanese sand lance: *Ammodytes personatus*); dotted line: load from plastic resin pellets.

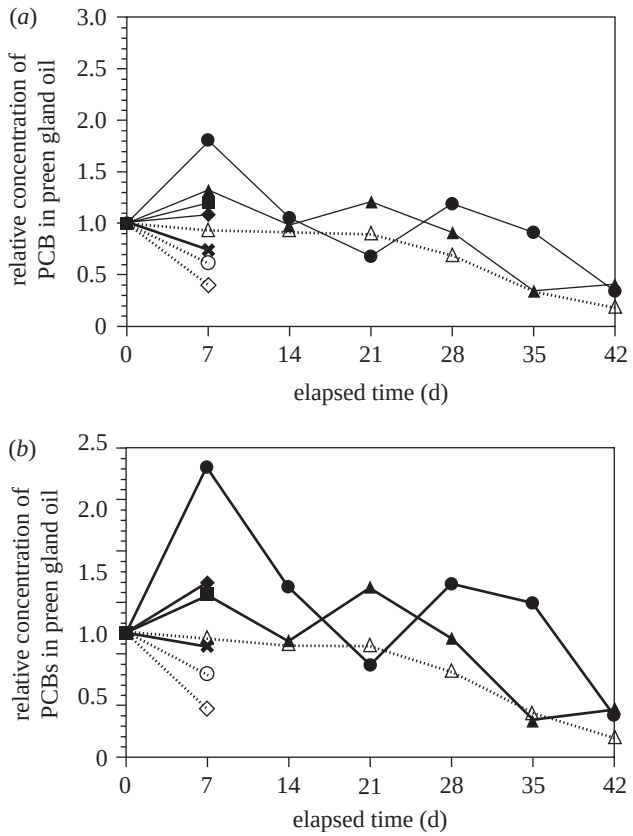

Figure 12. Time-course of PCBs in preen gland oil of the chicks during the feeding experiment: (*a*) total PCBs* and (*b*) lower chlorinated congeners**. *Total PCBs: sum of CB nos 8, 5, 28, 52, 44, 90, 101, 110, 77, 118, 132, 153, 138, 160, 187, 128, 180, 170, 190, and 206; **lower chlorinated congeners: sum of CB nos 8, 5, 18, 28, 52, 44, 66, 95. Polychlorinated biphenyl concentrations are normalized to those on day 0 on each series. Closed symbols and solid lines: plastic-feeding setting; open symbols and dotted lines: control setting. Replicate chicks no. 1 (closed square), 5 (cross), 8 (closed diamond), 10 (closed circle), 14 (closed triangle), 4 (open circle), 9 (open triangle), 18 (open diamond) were analysed. For chicks 1, 5, 8, 4, 18, samples were analysed on day 0 and day 7 only.

evaluate the contribution of the plastics to the total PCB burden of the seabird. Based on the weight of daily feeding of fish (approx. 10–120 g), daily exposure of PCBs from the fish is calculated to be approximately 15 ng. This 'daily' exposure (approx. 15 ng) is lower than those from the resin pellets (approx. 100 ng). However, plastic was fed once (day 0) during the experiment, while fish were fed daily throughout the experiment. Therefore, in the early stage of the experiment (days 0–7), PCB exposure from the pellets dominated those from the fish, whereas after Day 7, PCB exposure from the fish overcame that from the plastics (figure 11). This suggests that potential effects of fed-pellet-derived PCBs could be observed in the early stage of the experiment.

Figure 12 shows the preliminary results on the time-course of PCB concentrations in the preen gland oil from the streaked shearwater. It has been previously demonstrated that PCBs are accumulated in the preen gland oil owing to their hydrophobic nature and that the PCB concentrations in the oil reflect PCB levels in the internal tissue (i.e. abdominal adipose; Yamashita *et al.* 2007). Preen gland oil can be taken from a live bird without injury or stress. It provides a practical and useful tool to monitor PCB concentrations in biological tissue of 'live' seabirds over time. Figure 12 displays the relative concentrations of PCBs in the individual chicks. For each chick, the preen gland oil PCB concentration is normalized to that at day 0. The PCB concentrations in preen gland oil increased through day 7 for the plastic-feeding setting, whereas no such increase was observed in the control. On the other hand, in the later stage, PCB concentrations in preen gland oil showed no difference between the plastic-feeding and the control settings (figure 12*a*). This is consistent with a larger PCB dose from the plastics than from fish-only in the early stage of the experiment (through day 7). These results suggest that plastic-derived PCBs were transferred to the biological tissue of the seabird. However, the difference between the plastic-feeding chicks and the control was not statistically significant.

To obtain clearer evidence on the nature of the transfer, we focused on lower chlorinated congeners, because lower chlorinated congeners were relatively rich in plastic resin pellets compared with the fish (figure SI in the electronic supplementary material). The difference was probably because of biomagnification of higher chlorinated congeners in fish through the food web. Lower chlorinated congeners are more easily metabolized and selectively depleted when PCBs move through the food chain and, consequently, are less abundant in the fish tissue. Plastic resin pellets, however, just concentrate (partition) PCB congeners from seawater (not through the food web) and no metabolic process occurs, therefore lower chlorinated congeners that could be subject to biological degradation are not depleted. Lower chlorinated congeners can be regarded as a sensitive tracer to detect the contribution from plastic-derived PCBs. As illustrated in figure 12*b*, concentrations of lower chlorinated congeners in the plastic-feeding chicks increased up to three times from day 0 to day 7, whereas no increase in the congeners was observed for the control. This difference was statistically significant ($p < 0.05$, two-tailed

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t-test), demonstrating that transfer of PCBs, especially lower chlorinated congeners, occurs from ingested plastics to the biological tissue of the organisms that intake the plastics.

In higher-trophic-level organisms, plastic-derived contaminants compete with the biomagnified contaminants through the food web. However, marine plastics may act as a more important source of phenolic additive-derived chemicals (i.e. NP, OP and BPA) to higher-trophic-level organisms. Biomagnification of phenolic compounds through the food chain is unlikely as their hydrophilic group makes them easier to metabolize. Several studies suggest that biomagnification does not play an important role in the transfer of NP to animals and birds at higher trophic levels (e.g. Hu *et al.* 2005; Takeuchi *et al.* 2009). Thus, ingestion of marine plastics could be a direct and important route of phenolic chemicals to higher animals such as seabirds. Studies on the burden of the additive-derived chemicals to organisms, their transfer to organisms, and potential adverse effects are all needed.

9. CONCLUSIONS

This work reviews an expanse of literature addressing the role of plastics in the release and transport of environmental contaminants. Plastics can act as a source of environmental contaminants. For example, in the landfill environment, many additives and constitutional monomers leach out of plastics, and the discharged leachate can introduce plastic-derived contaminants into the environment. Concentrations of BPA and NP were significantly higher in landfill leachate than in wastewater effluent, and BPA accounted for more than half of the oestrogenic activity of the leachate.

Plastics also sorb and concentrate organic contaminants from the marine environment. The extent of uptake varies among plastics; polyethylene has a higher contaminant diffusivity and exhibits greater uptake of contaminants than other plastics including polypropylene and PVC. Ultraviolet weathering also appears to affect the uptake of contaminants by plastics. Unfortunately, our understanding of how to evaluate the risks associated with chemicals derived from marine plastics is poor. However, evidence is emerging that plastics with environmental contaminants can transport these compounds to organisms at various trophic levels.

Finally, these findings warn us that we should not underestimate the environmental impact of discarded plastics. As plastics production and usage continue to increase, particularly in economically developing countries, the environmental implications of their disposal should be carefully considered to avoid inadvertent release, magnification and transport of contaminants (Thompson *et al.* 2009*a*,*b*).

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PHILOSOPHICAL TRANSACTIONS OF THE ROYAL SOCIETY B

Phil. Trans. R. Soc. B (2009) **1**, 1–14
doi:10.1098/rstb.2009.0053

Review

Plastics, the environment and human health: current consensus and future trends

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Plastics have transformed everyday life; usage is increasing and annual production is likely to exceed 300 million tonnes by 2010. In this concluding paper to the Theme Issue on Plastics, the Environment and Human Health, we synthesize current understanding of the benefits and concerns surrounding the use of plastics and look to future priorities, challenges and opportunities. It is evident that plastics bring many societal benefits and offer future technological and medical advances. However, concerns about usage and disposal are diverse and include accumulation of waste in landfills and in natural habitats, physical problems for wildlife resulting from ingestion or entanglement in plastic, the leaching of chemicals from plastic products and the potential for plastics to transfer chemicals to wildlife and humans. However, perhaps the most important overriding concern, which is implicit throughout this volume, is that our current usage is not sustainable. Around 4 per cent of world oil production is used as a feedstock to make plastics and a similar amount is used as energy in the process. Yet over a third of current production is used to make items of packaging, which are then rapidly discarded. Given our declining reserves of fossil fuels, and finite capacity for disposal of waste to landfill, this linear use of hydrocarbons, via packaging and other short-lived applications of plastic, is simply not sustainable. There are solutions, including material reduction, design for end-of-life recyclability, increased recycling capacity, development of bio-based feedstocks, strategies to reduce littering, the application of green chemistry and revised risk assessment approaches. Such measures will be most effective through the combined actions of the public, industry, scientists and policymakers. There is some urgency, as the quantity of plastics produced in the first 10 years of the current century is likely to approach the quantity produced in the entire century that preceded.

Keywords: plastic; polymer; debris; endocrine disruption; phthalates; waste management

1. INTRODUCTION

Many of the current applications and the predicted benefits of plastic follow those outlined by Yarsley and Couzens in the 1940s. Their account of the benefits that plastics would bring to a person born nearly 70 years ago, at the beginning of this '*plastic age*', was told with much optimism:

> It is a world free from moth and rust and full of colour, a world largely built up of synthetic materials made from the most universally distributed substances, a world in which nations are more and more independent of localised naturalised resources, a world in which man, like a magician, makes what he wants for almost every need out of what is beneath and around him (Yarsley & Couzens 1945, p. 152).

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One contribution of 15 to a Theme Issue 'Plastics, the environment and human health'.

The durability of plastics and their potential for diverse applications, including widespread use as disposable items, were anticipated, but the problems associated with waste management and plastic debris were not. In fact the predictions were '*how much brighter and cleaner a world* [it would be] *than that which preceded this plastic age*' (Yarsley & Couzens 1945, p. 152).

This paper synthesizes current understanding of the benefits and concerns surrounding the use of plastics and looks to challenges, opportunities and priorities for the future. The content draws upon papers submitted to this Theme Issue on Plastics, the Environment and Human Health together with other sources. While selected citations are given to original sources of information, we primarily refer the reader to the discussion of a particular topic, and the associated references, in the Theme Issue papers. Here, we consider the subject from seven perspectives: plastics as materials; accumulation of plastic waste in the natural environment; effects of plastic debris in the

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environment and on wildlife; effects on humans; human usage, disposal and waste management solutions; biopolymers, degradable and biodegradable polymer solutions; and policy measures.

2. PLASTICS AS MATERIALS: AN OVERVIEW

Plastics are inexpensive, lightweight, strong, durable, corrosion-resistant materials, with high thermal and electrical insulation properties. The diversity of polymers and the versatility of their properties are used to make a vast array of products that bring medical and technological advances, energy savings and numerous other societal benefits (Andrady & Neal 2009). As a consequence, the production of plastics has increased substantially over the last 60 years from around 0.5 million tonnes in 1950 to over 260 million tonnes today. In Europe alone the plastics industry has a turnover in excess of 300 million euros and employs 1.6 million people (Plastics Europe 2008). Almost all aspects of daily life involve plastics, in transport, telecommunications, clothing, footwear and as packaging materials that facilitate the transport of a wide range of food, drink and other goods. There is considerable potential for new applications of plastics that will bring benefits in the future, for example as novel medical applications, in the generation of renewable energy and by reducing energy used in transport (Andrady & Neal 2009).

Virgin plastic polymers are rarely used by themselves and typically the polymer resins are mixed with various additives to improve performance. These additives include inorganic fillers such as carbon and silica that reinforce the material, plasticizers to render the material pliable, thermal and ultraviolet stabilizers, flame retardants and colourings. Many such additives are used in substantial quantities and in a wide range of products (Meeker *et al.* 2009). Some additive chemicals are potentially toxic (for example lead and tributyl tin in polyvinyl chloride, PVC), but there is considerable controversy about the extent to which additives released from plastic products (such as phthalates and bisphenol A, BPA) have adverse effects in animal or human populations. The central issue here is relating the types and quantities of additives present in plastics to uptake and accumulation by living organisms (Andrady & Neal 2009; Koch & Calafat 2009; Meeker *et al.* 2009; Oehlmann *et al.* 2009; Talsness *et al.* 2009; Wagner & Oehlmann 2009). Additives of particular concern are phthalate plasticizers, BPA, brominated flame retardants and anti-microbial agents. BPA and phthalates are found in many mass produced products including medical devices, food packaging, perfumes, cosmetics, toys, flooring materials, computers and CDs and can represent a significant content of the plastic. For instance, phthalates can constitute a substantial proportion, by weight, of PVC (Oehlmann *et al.* 2009), while BPA is the monomer used for production of polycarbonate plastics as well as an additive used for production of PVC. Phthalates can leach out of products because they are not chemically bound to the plastic matrix, and they have attracted particular attention because of their high production volumes and wide usage (Wagner & Oehlmann 2009; Talsness *et al.* 2009). Phthalates and BPA are detectable in aquatic environments, in dust and, because of their volatility, in air (Rudel *et al.* 2001, 2003). There is considerable concern about the adverse effects of these chemicals on wildlife and humans (Meeker *et al.* 2009; Oehlmann *et al.* 2009). In addition to the reliance on finite resources for plastic production, and concerns about additive effects of different chemicals, current patterns of usage are generating global waste management problems. Barnes *et al.* (2009) show that plastic wastes, including packaging, electrical equipment and plastics from end-of-life vehicles, are major components of both household and industrial wastes; our capacity for disposal of waste to landfill is finite and in some locations landfills are at, or are rapidly approaching, capacity (Defra *et al.* 2006). So from several perspectives it would seem that our current use and disposal of plastics is the cause for concern (Barnes *et al.* 2009; Hopewell *et al.* 2009).

3. ACCUMULATION OF PLASTIC WASTE IN THE NATURAL ENVIRONMENT

Substantial quantities of plastic have accumulated in the natural environment and in landfills. Around 10 per cent by weight of the municipal waste stream is plastic (Barnes *et al.* 2009) and this will be considered later in §6. Discarded plastic also contaminates a wide range of natural terrestrial, freshwater and marine habitats, with newspaper accounts of plastic debris on even some of the highest mountains. There are also some data on littering in the urban environment (for example compiled by EnCams in the UK; http://www.encams.org/home); however, by comparison with the marine environment, there is a distinct lack of data on the accumulation of plastic debris in natural terrestrial and freshwater habitats. There are accounts of inadvertent contamination of soils with small plastic fragments as a consequence of spreading sewage sludge (Zubris & Richards 2005), of fragments of plastic and glass contaminating compost prepared from municipal solid waste (Brinton 2005) and of plastic being carried into streams, rivers and ultimately the sea with rain water and flood events (Thompson *et al.* 2005). However, there is a clear need for more research on the quantities and effects of plastic debris in natural terrestrial habitats, on agricultural land and in freshwaters. Inevitably, therefore, much of the evidence presented here is from the marine environment. From the first accounts of plastic in the environment, which were reported from the carcasses of seabirds collected from shorelines in the early 1960s (Harper & Fowler 1987), the extent of the problem soon became unmistakable with plastic debris contaminating oceans from the poles to the Equator and from shorelines to the deep sea. Most polymers are buoyant in water, and since items of plastic debris such as cartons and bottles often trap air, substantial quantities of plastic debris accumulate on the sea surface and may also be washed ashore. As a consequence, plastics represent a considerable proportion (50–80%) of shoreline debris (Barnes *et al.* 2009). Quantities are highly variable in time and space, but there are reports of more than 100 000 items m^{-2} on some shorelines (Gregory 1978) and

up to 3 520 000 items km^{-2} at the ocean surface (Yamashita & Tanimura 2007). Gyres and oceanic convergences appear to be particularly contaminated, as do enclosed seas such as the Mediterranean (Barnes *et al.* 2009; Ryan *et al.* 2009). Despite their buoyant nature, plastics can become fouled with marine life and sediment causing items to sink to the seabed. For example, shallow seabeds in Brazil were more heavily contaminated than the neighbouring shorelines (Oigman-Pszczol & Creed 2007), indicating that the seabed may be an ultimate sink even for initially buoyant marine debris (Barnes *et al.* 2009). In some locations around Europe, it has been suggested that quantities on the seabed may exceed 10 000 items ha^{-1}, and debris has even been reported more than a 1000 m below the ocean surface, including accounts of inverted plastic bags passing a deep-sea submersible like an assembly of ghosts (Barnes *et al.* 2009; Gregory 2009; Ryan *et al.* 2009). Quantitative data on the abundance of debris on the seabed are still very limited, but there are concerns that degradation rates in the deep sea will be especially slow because of darkness and cold.

Monitoring the abundance of debris is important to establish rates of accumulation and the effectiveness of any remediation measures. Most studies assess the abundance of all types of anthropogenic debris including data on plastics and/or plastic items as a category. In general, the abundance of debris on shorelines has been extensively monitored, in comparison to surveys from the open oceans or the seabed. In addition to recording debris, there is a need to collect data on sources; for plastic debris this should include discharges from rivers and sewers together with littering behaviour. Here, the limited data we have suggest that storm water pulses provide a major pathway for debris from the land to the sea, with 81 g m^{-3} of plastic debris during high-flow events in the USA (Ryan *et al.* 2009). Methods to monitor the abundance of anthropogenic debris (including plastics) often vary considerably between countries and organizations, adding to difficulties in interpreting trends. As a consequence, the United Nations Environment Programme and the OSPAR Commission are currently taking steps to introduce standardized protocols (OSPAR 2007; Cheshire *et al.* 2009). Some trends are evident, however, typically with an increase in the abundance of debris and fragments between the 1960s and the 1990s (Barnes *et al.* 2009). More recently, abundance at the sea surface in some regions and on some shorelines appears to be stabilizing, while in other areas such as the Pacific Gyre there are reports of considerable increases. On shorelines the quantities of debris, predominantly plastic, are greater in the Northern than in the Southern Hemisphere (Barnes 2005). The abundance of debris is greater adjacent to urban centres and on more frequented beaches and there is evidence that plastics are accumulating and becoming buried in sediments (Barnes *et al.* 2009; Ryan *et al.* 2009). Barnes *et al.* (2009) consider that contamination of remote habitats, such as the deep sea and the polar regions, is likely to increase as debris is carried there from more densely populated areas. Allowing for variability between habitats and locations, it seems inevitable, however, that the quantity of debris in the environment as a whole will continue to increase—unless we all change our practices. Even with such changes, plastic debris that is already in the environment will persist for a considerable time to come. The persistence of plastic debris and the associated environmental hazards are illustrated poignantly by Barnes *et al.* (2009) who describe debris that had originated from an aeroplane being ingested by an albatross some 60 years after the plane had crashed.

4. EFFECTS OF PLASTIC DEBRIS IN THE ENVIRONMENT AND ON WILDLIFE

There are some accounts of effects of debris from terrestrial habitats, for example ingestion by the endangered California condor, *Gymnogyps californianus* (Mee *et al.* 2007). However, the vast majority of work describing environmental consequences of plastic debris is from marine settings and more work on terrestrial and freshwater habitats is needed. Plastic debris causes aesthetic problems, and it also presents a hazard to maritime activities including fishing and tourism (Moore 2008; Gregory 2009). Discarded fishing nets result in ghost fishing that may result in losses to commercial fisheries (Moore 2008; Brown & Macfadyen 2007). Floating plastic debris can rapidly become colonized by marine organisms and since it can persist at the sea surface for substantial periods, it may subsequently facilitate the transport of non-native or 'alien' species (Barnes 2002; Barnes *et al.* 2009; Gregory 2009). However, the problems attracting most public and media attention are those resulting in ingestion and entanglement by wildlife. Over 260 species, including invertebrates, turtles, fish, seabirds and mammals, have been reported to ingest or become entangled in plastic debris, resulting in impaired movement and feeding, reduced reproductive output, lacerations, ulcers and death (Laist 1997; Derraik 2002; Gregory 2009). The limited monitoring data we have suggest rates of entanglement have increased over time (Ryan *et al.* 2009). A wide range of species with different modes of feeding including filter feeders, deposit feeders and detritivores are known to ingest plastics. However, ingestion is likely to be particularly problematic for species that specifically select plastic items because they mistake them for their food. As a consequence, the incidence of ingestion can be extremely high in some populations. For example, 95 per cent of fulmars washed ashore dead in the North Sea have plastic in their guts, with substantial quantities of plastic being reported in the guts of other birds, including albatross and prions (Gregory 2009). There are some very good data on the quantity of debris ingested by seabirds recorded from the carcasses of dead birds. This approach has been used to monitor temporal and spatial patterns in the abundance of sea-surface plastic debris on regional scales around Europe (Van Franeker *et al.* 2005; Ryan *et al.* 2009).

An area of particular concern is the abundance of small plastic fragments or microplastics. Fragments as small as 1.6 μm have been identified in some marine habitats, and it seems likely there will be even smaller pieces below current levels of detection. A

recent workshop convened in the USA by the National Oceanic and Atmospheric Administration concluded that microplastics be defined as pieces <5 mm with a suggested lower size boundary of 333 μm so as to focus on microplastics that will be captured using conventional sampling approaches (Arthur *et al.* 2009). However, we consider it important that the abundance of even smaller fragments is not neglected. Plastic fragments appear to form by the mechanical and chemical deterioration of larger items. Alternative routes for microplastics to enter the environment include the direct release of small pieces of plastics that are used as abrasives in industrial and domestic cleaning applications (e.g. shot blasting or scrubbers used in proprietary hand cleansers) and spillage of plastic pellets and powders that are used as a feedstock for the manufacture of most plastic products. Data from shorelines, from the open ocean and from debris ingested by seabirds, all indicate that quantities of plastic fragments are increasing in the environment, and quantities on some shores are substantial (>10% by weight of strandline material; Barnes *et al.* 2009). Laboratory experiments have shown that small pieces such as these can be ingested by small marine invertebrates including filter feeders, deposit feeders and detritivores (Thompson *et al.* 2004), while mussels were shown to retain plastic for over 48 days (Browne *et al.* 2008). However, the extent and consequences of ingestion of microplastics by natural populations are not known.

In addition to the physical problems associated with plastic debris, there has been much speculation that, if ingested, plastic has the potential to transfer toxic substances to the food chain (see Teuten *et al.* 2009). In the marine environment, plastic debris such as pellets, fragments and microplastics have been shown to contain organic contaminants including polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons, petroleum hydrocarbons, organochlorine pesticides (2,2′-bis(*p*-chlorophenyl)-1,1,1 trichloroethane (DDT) and its metabolites; together with hexachlorinated hexane (HCH)), polybrominated diphenylethers (PBDEs), alkylphenols and BPA at concentrations ranging from ng g^{-1} to μg g^{-1}. Some of these compounds are added to plastics during manufacture while others can absorb into plastic debris from the environment. Work in Japan has shown that plastics can accumulate and concentrate persistent organic pollutants that have arisen in the environment from other sources. These contaminants can become orders of magnitude more concentrated on the surface of plastic debris than in the surrounding sea water (Mato *et al.* 2001). Teuten *et al.* (2009) describe experiments to examine the transfer of these contaminants from plastics to seabirds and other animals. The potential for transport varies among contaminants, polymers and possibly also according to the state of environmental weathering of the debris. Recent mathematical modelling studies have shown that even very small quantities of plastics could facilitate transport of contaminants from plastic to organisms upon ingestion. This could present a direct and important route for the transport of chemicals to higher animals such as seabirds (Teuten *et al.* 2007, 2009), but will depend upon the nature of the habitat and the amount and type of plastics present. For instance, the extent to which the presence of plastic particles might contribute to the total burden of contaminants transferred from the environment to organisms will depend upon competitive sorption and transport by other particulates (Arthur *et al.* 2009). The abundance of fragments of plastic is increasing in the environment; these particles, especially truly microscopic fragments less than the 333 μm proposed by NOAA (see earlier), have a relatively large surface area to volume ratio that is likely to facilitate the transport of contaminants, and because of their size such fragments can be ingested by a wide range of organisms. Hence, the potential for plastics to transport and release chemicals to wildlife is an emerging area of concern.

More work will be needed to establish the full environmental relevance of plastics in the transport of contaminants to organisms living in the natural environment, and the extent to which these chemicals could then be transported along food chains. However, there is already clear evidence that chemicals associated with plastic are potentially harmful to wildlife. Data that have principally been collected using laboratory exposures are summarized by Oehlmann *et al.* (2009). These show that phthalates and BPA affect reproduction in all studied animal groups and impair development in crustaceans and amphibians. Molluscs and amphibians appear to be particularly sensitive to these compounds and biological effects have been observed in the low ng l^{-1} to μg l^{-1} range. In contrast, most effects in fish tend to occur at higher concentrations. Most plasticizers appear to act by interfering with hormone function, although they can do this by several mechanisms (Hu *et al.* 2009). Effects observed in the laboratory coincide with measured environmental concentrations, thus there is a very real probability that these chemicals are affecting natural populations (Oehlmann *et al.* 2009). BPA concentrations in aquatic environments vary considerably, but can reach 21 μg l^{-1} in freshwater systems and concentrations in sediments are generally several orders of magnitude higher than in the water column. For example, in the River Elbe, Germany, BPA was measured at 0.77 μg l^{-1} in water compared with 343 μg kg^{-1} in sediment (dry weight). These findings are in stark contrast with the European Union environmental risk assessment predicted environmental concentrations of 0.12 μg l^{-1} for water and 1.6 μg kg^{-1} (dry weight) for sediments.

Phthalates and BPA can bioaccumulate in organisms, but there is much variability between species and individuals according to the type of plasticizer and experimental protocol. However, concentration factors are generally higher for invertebrates than vertebrates, and can be especially high in some species of molluscs and crustaceans. While there is clear evidence that these chemicals have adverse effects at environmentally relevant concentrations in laboratory studies, there is a need for further research to establish population-level effects in the natural environment (see discussion in Oehlmann *et al.* 2009), to establish the long-term effects of exposures (particularly due to exposure of embryos), to determine effects of exposure to contaminant mixtures and to establish the role of plastics as sources (albeit not exclusive sources) of these contaminants (see Meeker *et al.* (2009) for discussion of sources and routes of exposure).

5. EFFECTS ON HUMANS: EPIDEMIOLOGICAL AND EXPERIMENTAL EVIDENCE

Turning to adverse effects of plastic on the human population, there is a growing body of literature on potential health risks. A range of chemicals that are used in the manufacture of plastics are known to be toxic. Biomonitoring (e.g. measuring concentration of environmental contaminants in human tissue) provides an integrated measure of an organism's exposure to contaminants from multiple sources. This approach has shown that chemicals used in the manufacture of plastics are present in the human population, and studies using laboratory animals as model organisms indicate potential adverse health effects of these chemicals (Talsness *et al.* 2009). Body burdens of chemicals that are used in plastic manufacture have also been correlated with adverse effects in the human population, including reproductive abnormalities (e.g. Swan *et al.* 2005; Swan 2008; Lang *et al.* 2008).

Interpreting biomonitoring data is complex, and a key task is to set information into perspective with dose levels that are considered toxic on the basis of experimental studies in laboratory animals. The concept of 'toxicity' and thus the experimental methods for studying the health impacts of the chemicals in plastic, and other chemicals classified as endocrine disruptors, is currently undergoing a transformation (a paradigm inversion) since the disruption of endocrine regulatory systems requires approaches very different from the study of acute toxicants or poisons. There is thus extensive evidence that traditional toxicological approaches are inadequate for revealing outcomes such as 'reprogramming' of the molecular systems in cells as a result of exposure to very low doses during critical periods in development (e.g. Myers *et al.* 2009). Research on experimental animals informs epidemiologists about the potential for adverse effects in humans and thus plays a critical role in chemical risk assessments. A key conclusion from the paper by Talsness *et al.* (2009) is the need to modify our approach to chemical testing for risk assessment. As noted by these authors and others, there is a need to integrate concepts of endocrinology in the assumptions underlying chemical risk assessment. In particular, the assumptions that dose–response curves are monotonic and that there are threshold doses (safe levels) are not true for either endogenous hormones or for chemicals with hormonal activity (which includes many chemicals used in plastics) (Talsness *et al.* 2009).

The biomonitoring approach has demonstrated phthalates and BPA, as well as other additives in plastics and their metabolites, are present in the human population. It has also demonstrated that the most common human exposure scenario is to a large number of these chemicals simultaneously. These data indicate differences according to geographical location and age, with greater concentrations of some of these chemicals in young children. While exposure via house dust is extensive (Rudel *et al.* 2008), it would appear that at least for some phthalates (e.g. diethylhexyl phthalate, DEHP), foodstuffs and to a lesser extent use of oral drugs probably present major uptake pathways (Wormuth *et al.* 2006). Exposure data for BPA are similar but less extensive. While average concentrations of phthalates in selected populations worldwide appear quite similar, there is evidence of considerable variability in daily intake rates among individuals, and even within individuals (Peck *et al.* 2009). Exposures through ingestion, inhalation and dermal contact are all considered important routes of exposure for the general population (Adibi *et al.* 2003; Rudel *et al.* 2003). Koch & Calafat (2009) show that while mean/median exposures for the general population were below levels determined to be safe for daily exposure (USA, EPA reference dose, RfD; and European Union tolerable daily intake, TDI), the upper percentiles of di-butyl phthalate and DEHP urinary metabolite concentrations show that for some people daily intake might be substantially higher than previously assumed and could exceed estimated safe daily exposure levels. Current 'safe' exposure levels are typically based on the application of traditional toxicological assumptions regarding acute toxicants to calculate daily exposures for chemicals in a range of widely used plastic items. The toxicological consequences of such exposures, especially for susceptible subpopulations such as children and pregnant women, remain unclear and warrant further investigation. However, there is evidence of associations between urinary concentrations of some phthalate metabolites and biological outcomes (Swan *et al.* 2005; Swan 2008). For example, an inverse relationship has been reported between the concentrations of DEHP metabolites in the mother's urine and anogenital distance, penile width and testicular decent in male offspring (Swan *et al.* 2005; Swan 2008). In adults, there is some evidence of a negative association between phthalate metabolites and semen quality (Meeker & Sathyanarayana) and between high exposures to phthalates (workers producing PVC flooring) and free testosterone levels. Moreover, recent work (Lang *et al.* 2008) has shown a significant relationship between urine levels of BPA and cardiovascular disease, type 2 diabetes and abnormalities in liver enzymes, and Stahlhut *et al.* (2009) have reported that exposure of adults in the USA to BPA is likely to occur from multiple sources and that the half-life of BPA is longer than previously estimated, and the very high exposure of premature infants in neonatal intensive-care units to both BPA and phthalates is of great concern (Calafat *et al.* 2009). These data indicate detrimental effects in the general population may be caused by chronic low-dose exposures (separately or in combination) and acute exposure to higher doses, but the full extent to which chemicals are transported to the human population by plastics is yet to be confirmed.

Much has been learned about toxicological effects on humans from experiments using laboratory animals. This approach has been used to examine component chemicals used in plastic production. A summary of work on phthalates, BPA and tetrabromobisphenol A (TBBPA) is presented by Talsness *et al.* (2009). The male reproductive tract is particularly sensitive to phthalate exposure. However, most reproductive effects are not exerted by phthalate diesters themselves, but by their monoester metabolites, which are formed in the liver. The majority of these studies have been done using rats as a model organism,

with doses at least an order of magnitude higher than those to which humans are commonly exposed, but they have resulted in rapid, severe changes in the rat testis. Reproductive effects have also been described in mice and guinea pigs. Effects on pre- and early post-natal development are of particular concern, and recent animal studies have shown exposures to certain phthalates can result in severe disorders of the developing male reproductive system. It should be noted that most work on animals has used phthalate exposures much higher than estimated daily human exposures (see above), and researchers have only recently started to investigate possible biological effects within the range of median human phthalate exposure (Talsness *et al.* 2009). This is of critical importance because epidemiological studies have reported associations between phthalate levels and a number of adverse health effects in humans (Swan *et al.* 2005), suggesting that either humans are more sensitive to phthalates than experimental animals or that the testing paradigm used in traditional toxicological studies, which examines one phthalate at a time, has not served to accurately predict adverse effects from the mixture of phthalates to which humans are exposed (Andrade *et al.* 2006; NAS 2008).

For BPA, there is an extensive published literature showing adverse effects of exposure at very low doses, based on administration during development and to adult experimental animals. In particular, unlike the case for experimental animal research on phthalates, there are now hundreds of experiments on laboratory animals using doses within the range of human exposures (Vandenberg *et al.* 2007). The rate and extent to which BPA is metabolized affect the interpretation of these findings, but even very low doses of BPA have been shown to cause significant stimulation of insulin secretion followed by insulin resistance in mice, a significant decrease in sperm production by rats, a decrease in maternal behaviour in mice and disruption of hippocampal synapses, leading to the appearance of a brain typical of that seen in senility in both rats and monkeys. The greatest concerns with exposure to BPA are during development; BPA appears to affect brain development leading to loss of sex differentiation in brain structures and behaviour (Talsness *et al.* 2009). A further important observation regarding adverse responses to developmental exposures of animals to very low doses of BPA is that many relate to disease trends in humans. Less has been published on effects of the flame retardant TBBPA, but there is evidence of effects on thyroid hormones, pituitary function and reproductive success in animals (Talsness *et al.* 2009).

Despite the environmental concerns about some of the chemicals used in plastic manufacture, it is important to emphasize that evidence for effects in humans is still limited and there is a need for further research and in particular, for longitudinal studies to examine temporal relationships with chemicals that leach out of plastics (Adibi *et al.* 2008). In addition, the traditional approach to studying the toxicity of chemicals has been to focus only on exposure to individual chemicals in relation to disease or abnormalities. However, because of the complex integrated nature of the endocrine system, it is critical that future studies involving endocrine-disrupting chemicals that leach from plastic products focus on mixtures of chemicals to which people are exposed when they use common household products. For example, in a study conducted in the USA, 80 per cent of babies were exposed to measurable levels of at least nine different phthalate metabolites (Sathyanarayana *et al.* 2008), and the health impacts of the cumulative exposure to these chemicals need to be determined. An initial attempt at examining more than one phthalate as a contributor to abnormal genital development in babies has shown the importance of this approach (Swan 2008). Studies of mixtures of chemicals therefore also need to extend beyond mixtures of the same class of chemical, such as mixtures of different phthalates or of different PCBs. For example, PVC (used in a wide range of products in the home including water pipes) may contain phthalates, BPA, flame retardants such as PBDEs or TBBPA, cadmium, lead and organotins, all of which have been shown in animal studies to result in obesity (Heindel & vom Saal 2009). In addition, the monomer used to manufacture PVC plastic, vinyl chloride, is a known carcinogen and exposure can cause angiosarcoma of the liver among factory workers (Bolt 2005; Gennaro *et al.* 2008). PVC in medical tubing has also been shown to be a source of high DEHP exposure among infants in neonatal intensive-care nurseries (Green *et al.* 2005) and probably contributes to the high levels of BPA found in these babies since BPA is an additive in PVC plastic (Calafat *et al.* 2009).

Examining the relationship between plastic additives and adverse human effects presents a number of challenges. In particular, the changing patterns of production and use of both plastics, and the additives they contain, as well as the confidential nature of industrial specifications makes exposure assessment particularly difficult. Evolving technology, methodology and statistical approaches should help disentangle the relationships between these chemicals and health effects. However, with most of the statistically significant hormone alternations that have been attributed to environmental and occupational exposures, the actual degree of hormone alteration has been considered subclinical. Hence, more information is required on the biological mechanisms that may be affected by plastic additives and in particular, low-dose chronic exposures. Meanwhile we should consider strategies to reduce the use of these chemicals in plastic manufacture and/or develop and test alternatives (for example citrates are being developed as substitute plasticizers). This is the goal of the new field of green chemistry, which is based on the premise that development of chemicals for use in commerce should involve an interaction between biologists and chemists. Had this approach been in place 50 years ago it would probably have prevented the development of chemicals that are recognized as likely endocrine disruptors (Anastas & Beach 2007). There is also a need for industry and independent scientists to work more closely with, rather than against, each other in order to focus effectively on the best ways forward. For example, contrast comments

on BPA by Bird (2005) with those of vom Saal (2005), and contrast comments in this volume on the safety of plastic additives by Andrady & Neal (2009) with those by Koch & Calafat (2009), Meeker *et al.* (2009), Oehlmann *et al.* (2009) and Talsness *et al.* (2009).

6. PRODUCTION, USAGE, DISPOSAL AND WASTE MANAGEMENT SOLUTIONS

Accumulation of plastic debris in the environment and the associated consequences are largely avoidable. Considerable immediate reductions in the quantity of waste entering natural environments, as opposed to landfill, could be achieved by better waste disposal and material handling. Littering is a behavioural issue and some have suggested that it has increased in parallel with our use of disposable products and packaging. Perhaps increasing the capacity to recycle will help to reverse this trend such that we start to regard end-of-life materials as valuable feedstocks for new production rather than waste. To achieve this will require better education, engagement, enforcement and recycling capacity (figure 1*a–f*). Unfortunately, we were unable to source a contribution on education and public engagement, but it is evident that social research on littering behaviour could be very informative. A recent report by EnCams in the UK examined attitudes towards littering in 2001 and then again in 2006. This indicated that despite greater awareness among the general public about the problems of littering, the propensity to litter had actually increased; five key attitudes and behaviours were noted and these offer valuable insight for future research (EnCams 2006). There is evidence that appropriate education can influence behaviour. For example, pre-production plastic pellets (a feedstock for production of plastic products, also described as nurdles or mermaids tears) account for around 10 per cent, by number, of the plastic debris recorded on shorelines in Hawaii (McDermid & McMullen 2004) and substantial quantities have been recorded on shorelines in New Zealand (Gregory 1978). These pellets have entered the environment through spillage during transportation, handling and as cargo lost from ships. In the USA guidelines (Operation Clean-Sweep, figure 1*e*) on handling of resin pellets are reported to have reduced spillage during trials (Moore *et al.* 2005). Conservation organizations such as the UK Marine Conservation Society play an important role in education, and the annual beach cleans they organize can be a good way to raise public awareness and to collect data on trends in the abundance of debris on shorelines (see www.mcsuk.org and Ocean Conservancy, International Coastal Cleanup www.oceanconservancy.org). However, there is a pressing need for education to reduce littering at source (figure 1*d* and *e*). This is especially important in urban settings where increased consumption of on-the-go/fast food coupled, in some locations, with a reduction in the availability of bins as a consequence of concerns about terrorism is likely to result in increased littering. Where plastic debris enters watercourses as a consequence of dumping or littering a range of strategies including catch basin inserts, booms and separators can be used to facilitate removal (figure 1*f*).

Substantial quantities of end-of-life plastics are disposed of to landfill. Waste generation statistics vary among countries and according to the rationale for data collection. For instance, plastics are a small component of waste by weight but a large component by volume. Temporal and spatial comparisons can thus be confounded, and data on quantities of waste recycled can be skewed according to categorization of various wastes. However, in many locations space in landfill is running out (e.g. Defra *et al.* 2006). It has also been suggested that because of the longevity of plastics, disposal to landfill may simply be storing problems for the future (Barnes *et al.* 2009; Hopewell *et al.* 2009). For example, plasticizers and other additive chemicals have been shown to leach from landfills (Teuten *et al.* (2009) and references therein). The extent of this varies according to conditions, particularly pH and organic content. There is evidence, however, that landfills can present a significant source of contaminants, such as BPA, to aquatic environments. Efficient treatment approaches are available and are in use in some countries (Teuten *et al.* 2009).

From a waste management perspective, the three R's—*reduce, reuse* and *recycle* are widely advocated to reduce the quantities of plastic and especially plastics packaging the waste we generate (figure 1*a–c*). Hopewell *et al.* (2009) outline the benefits and limitations of these strategies. They show that to be effective we need to consider the three R's in combination with each other and together with a fourth 'R', *energy recovery.* Indeed we also need to consider a 5th 'R', molecular redesign, as an emerging and potentially very important strategy. Hence, the three R's become five: 'reduce, reuse, recycle, recover and redesign'. There are opportunities to 'reduce' usage of raw material by down gauging (figure 1*a*) and there are also some opportunities to 'reuse' plastics, for example, in the transport of goods on an industrial (pallets, crates; figure 1*b*) and a domestic (carrier bags) scale. However, there is limited potential for wide-scale reuse of retail packaging because of the substantial back-haul distances and logistics involved in returning empty cartons to suppliers. Some of the energy content of plastics can be 'recovered' by incineration, and through approaches such as co-fuelling of kilns, reasonable energy efficiency can be achieved. These approaches have benefits compared with disposal to landfill since some of the energy content of plastics is recovered. However, energy recovery does not reduce the demand for raw material used in plastic production, hence it is considered less energy efficient than product recovery via recycling (WRAP 2006; Defra 2007). In addition, concerns about emissions from incinerators (Katami *et al.* 2002) can reduce the appeal of this waste disposal option. There is now strong evidence to indicate significant potential lies in increasing our ability to effectively recycle end-of-life plastic products (WRAP 2006, 2008; Defra 2007; fig 1*c*). Although thermoplastics have been recycled since the 1970s, the proportion of material recycled has increased substantially in recent

Figure 1. Solutions include: (*a*) measures to reduce the production of new plastics from oil. This example shows how small changes in product packing reduced the weight of packaging required by 70%, while (*b*) re-useable plastic packing crates have reduced the packaging consumption of the same retailer by an estimated 30,000 tonnes per annum; and (*c*) recycling; here, bales of used plastic bottles have been sorted prior to recycling into new items, such as plastic packaging or textiles. Measures to reduce the quantity of plastic debris in the natural environment include: (*d*) educational signage to reduce contamination via storm drains and (*e*) via industrial spillage, together with (*f*) booms to intercept and facilitate the removal of riverine debris (photographs (*a*) and (*b*), and associated usage statistics) courtesy of Marks and Spencer PLC; (*c*) courtesy of P. Davidson, WRAP; (*d, e*, *f*) courtesy of C. Moore, Algalita Marine Research Foundation).

years and represents one of the most dynamic areas of the plastic industry today (WRAP 2006, 2008).

The recycling message is simple; both industry and society need to regard end-of-life items, including plastics, as raw materials rather than waste. At present our consumption of fossil fuels for plastic production is linear, from oil to waste via plastics. It is essential to take a more cyclical approach to material usage, but achieving this goal is complex (Hopewell *et al.* 2009). Greatest energy efficiency is achieved where recycling diverts the need for use of fossil fuels as raw materials (figure 1*c*); good examples being the recycling of old polyethylene terephthalate (PET) bottles into new ones (closed-loop recycling) or where low-density polyethylene bottles are converted into waste bins (semi-closed loop). In addition to benefits as a consequence of more sustainable material usage, a recent life cycle analysis calculated that use of 100 per cent recycled PET rather than virgin PET to produce plastic bottles could give a 27 per cent reduction on CO_2 emissions (WRAP 2008; Hopewell *et al.* 2009).

There are some very encouraging trends, with growth in mechanical recycling increasing at 7 per cent per annum in western Europe. However, there is considerable regional variation in recycling rates and globally only a small proportion of plastic waste is recycled (see Barnes *et al.* (2009) for US data; see Hopewell *et al.* (2009) for European data). Items made of a single polymer are easier and more efficient to recycle than composite items, films and mixed wastes. As a consequence, it is currently not possible to recycle a substantial proportion of the packaging in a typical shopping basket (Hopewell *et al.* 2009). On reading the account by Hopewell *et al.* (2009), the ingenuity of the separation procedures for recycling is evident (Fourier-transform near-infrared spectroscopy, optical colour separation, X-ray detection), but one cannot help but wonder why similar ingenuity has not been focused on designing products for better end-of-life recyclability. Historically, the main considerations for the design of plastic packaging have been getting goods safely to market and product marketing. There is an increasing urgency to also design products, especially packaging, in order to achieve material reduction and greater end-of-life recyclability. Public support for recycling is high in some countries (57% in the UK and 80% in Australia; Hopewell *et al.* 2009), and consumers are keen to recycle, but the small size and the diversity of different symbols to describe a product's potential recyclability, together with uncertainties as to whether a product will actually be recycled if it is offered for collection, can hinder engagement. In our opinion, what is needed is a simplification and streamlining of everyday packaging, to facilitate recyclability, together with clearer labelling to inform users. One option could be a traffic light system so that consumers can easily distinguish from printed product labelling between packages that use recycled content and have high end-of-life recyclability (marked with a green spot), those that have low end-of-life recyclability and are predominantly made of virgin polymer (red spot), and those which lie between these extremes (amber spot). With combined actions including waste reduction, design for end-of-life,

better labelling for consumers, increased options for on-the-go disposal to recycling and improved recycling capability, Hopewell *et al.* (2009) consider it could be possible to divert the majority of plastic from landfill over the next few decades (figure 1*a–c*). This will require consistency of policy measures and facilities among regions and will also require the cooperation of industry since ultimately there needs to be an acceptance of reduced usage and hence reduced income associated with the production of plastics from virgin polymer.

Molecular *redesign* of plastics (the 5th R) has become an emerging issue in green chemistry (Anastas & Warner 1998; Anastas *et al.* 2000; Anastas & Crabtree 2009) that should be incorporated within the design and life cycle analysis of plastics. In this context, green chemists aspire to design chemical products that are fully effective, yet have little or no toxicity or endocrine-disrupting activity; that break down into innocuous substances if released into the environment after use; and/or that are based upon renewable feedstocks, such as agricultural wastes. One of the fundamental factors limiting progress on all other R's is that the design criteria used to develop new monomers have rarely included specifications to enhance reusability, recyclability or recovery of plastic once it has been used. Typically, such assessments have only been made after a product entered the marketplace and problems involving waste and/or adverse health effects have begun to appear. Had the guiding principles of Green Chemistry (Anastas & Warner 1998) been available to inform the syntheses of polymers over the past century, perhaps some of the environmental and health concerns described in this Theme Issue would be more manageable. To date, the application of these design criteria to polymers has remained largely in the laboratory. Polylactic acid (PLA) (Drumright *et al.* 2000), a biodegradable polymer sourced from corn and potatoes, has entered the marketplace and has the potential to make a valuable contribution among other strategies for waste management. However, life cycle analyses are required to help establish the most appropriate usage, disposal (e.g. Song *et al.* 2009 illustrate relatively slow degradability of PLA in home composting) and hence labelling, of biopolymers such as this (WRAP 2009).

7. BIOPOLYMERS, DEGRADABLE AND BIODEGRADABLE POLYMER SOLUTIONS

Degradable polymers have been advocated as an alternative to conventional oil-based plastics and their production has increased considerably in recent decades. Materials with functionality comparable to conventional plastics can now be produced on an industrial scale; they are more expensive than conventional polymers and account for less than 1 per cent of plastics production (Song *et al.* 2009). Biopolymers differ from conventional polymers in that their feedstock is from renewable biomass rather than being oil-based. They may be natural polymers (e.g. cellulose), or synthetic polymers made from biomass monomers (e.g. PLA) or synthetic polymers made from synthetic monomers derived from biomass (e.g. polythene derived from bioethanol) (WRAP 2009). They are often described as renewable polymers since the original biomass, for example corn grown in agriculture, can be reproduced. The net carbon dioxide emission may be less than that with conventional polymers, but it is not zero since farming and pesticide production have carbon dioxide outputs (WRAP 2009). In addition, as a consequence of our rapidly increasing human population, it seems unlikely that there will be sufficient land to grow crops for food, let alone for substantial quantities of packaging in which to wrap it. One solution is to recycle waste food into biopolymers; this has merit, but will ultimately be limited by the amount of waste food available.

Biopolymers that are designed to breakdown in an industrial composter are described as 'biodegradables' while those that are intended to degrade in a domestic composter are known as 'compostable'. There are benefits of these biodegradable materials in specific applications, for example, with packaging of highly perishable goods where, regrettably, it can be necessary to dispose of perished unopened and unused product together with its wrapper. Song *et al.* (2009) show experimentally that degradation of biodegradable, as opposed to compostable, polymers can be very slow in home composters (typically less than 5% loss of biomass in 90 days). Degradation of these polymers in landfills is also likely to be slow and create unwanted methane emissions. Hence, the benefits of biopolymers are only realized if they are disposed of to an appropriate waste management system that uses their biodegradable features. Typically, this is achieved via industrial composting at 50°C for around 12 weeks to produce compost as a useful product.

Some biopolymers, such as PLA, are biodegradable, but others such as polythene derived from bioethanol are not. A further complication is that degradable, as opposed to biodegradable, polymers (also called 'oxo-biodegradable', 'oxy-degradable' or 'UV-degradable') can also be made from oil-based sources but as a consequence are not biopolymers. These degradable materials are typically polyethylene together with additives to accelerate the degradation. They are used in a range of applications and are designed to break down under UV exposure and/or dry heat and mechanical stress, leaving small particles of plastic. They do not degrade effectively in landfills and little is known about the timescale, extent or consequences of their degradation in natural environments (Barnes *et al.* 2009; Teuten *et al.* 2009). Degradable polymers could also compromise the quality of recycled plastics if they enter the recycling stream. As a consequence, use of degradable polymers is not advocated for primary retail packaging (WRAP 2009).

There is a popular misconception that degradable and biodegradable polymers offer solutions to the problems of plastic debris and the associated environmental hazards that result from littering. However, most of these materials are unlikely to degrade quickly in natural habitats, and there is concern that degradable, oil-based polymers could merely disintegrate into small pieces that are not in themselves any more degradable than conventional plastic (Barnes *et al.* 2009). So while biodegradable polymers offer some waste management solutions, there are limitations and considerable misunderstanding among the general

Table 1. Synthesis of current knowledge, uncertainty and recommended actions relevant to environmental and human health concerns arising from current production, use and disposal of plastics.

	established knowledge	concerns and uncertainty	recommendations for industry, research and policy
production and use	plastics are inexpensive, lightweight, versatile, water resistant and durable annual growth in plastic production is approximately 9% (currently >260 Mt yr^{-1}) around 8% of world oil production is used to make plastics plastics bring extensive societal, human health and environmental benefits >33% of production is used for disposable items of packaging	is our usage of hydrocarbons for plastics sustainable? to what extent could biopolymers replace oil-based plastics? is there sufficient arable land for production of biomass (crops) required for biopolymers? to what extent does use of plastic powders as cleaning abrasives, and scrubbers results in direct release of particles to environment?	increase/incentivize material reduction and reuse construct life cycle analysis of production, disposal/recycling of major polymers (including biopolymers, degradable and biodegradable polymers) and plastic products develop alternative monomers, polymers and additives using green chemistry approaches revise international standards for and introduce accurate/ informative labelling of recyclable, 'degradable', 'biodegradable' and compostable polymers
disposal: waste management	plastics are a substantial part of domestic and industrial wastes in landfill recycling of some polymers (e.g. PET) has increased considerably in recent years, but substantial quantities of plastic waste not compatible with recycling biodegradable polymers typically require industrial composting and do not degrade in landfill biodegradable plastics can compromise recycling	are current disposal strategies sustainable—lack of space in landfill? to what extent do chemicals leach from plastic in landfill? little is known about the degradability or environmental fate of additives used in biodegradable polymers	increase/incentivize product design towards use of recycled feedstock and increased end-of-life recyclability improve methods to collect and separate plastic waste for recycling investment in/incentivize recycling operations standardize labelling so consumers can identify products with high end-of-life recyclability (traffic light system) research and monitoring of leachates from landfills
disposal: littering and dumping	plastic debris is common in marine habitats worldwide, including poles and deep sea the abundance of plastic debris is increasing/stabilizing (not declining) plastic debris is fragmenting, with pieces <20 μm on shorelines and in water column	to what extent will breakdown of plastic debris increase the abundance of small fragments in the environment? rates of accumulation of debris on land, in freshwaters and in the deep sea are not certain do biodegradable or compostable plastics degrade in natural habitats?	education/incentives to promote the value of end-of-life plastics as a feedstock for recycling education and associated enforcement on the wasteful and adverse ecological effects of plastic spillage, dumping and littering develop standard protocols and monitoring to evaluate trends in the abundance of plastic debris across in natural habitats cleaning programmes in natural, urban and industrial locations research on breakdown of degradable and biodegradables
issues relating to wildlife	>260 species are known to ingest or become entangled in plastic debris ingestion is widespread in some populations (>95% of individuals) and can compromise feeding entanglement in plastic debris can lead to severe injury and death	does ingestion of, or entanglement in, plastic debris have effects at the population level or can such effects combine with other stressors to do so? to what extent do plastics transport/release chemicals to wildlife? what are the consequences of the accumulation of small plastic particles (e.g. abrasives from cleaning applications) in the environment?	research to establish the distribution, abundance and environmental consequences of micro- and nano-plastic fragments research to establish potential for plastics to transport chemicals to food chain research to establish population-level consequences of ingestion and entanglement education, monitoring and cleaning (see above)

(*Continued*)

Table 1. (*Continued*)

	established knowledge	concerns and uncertainty	recommendations for industry, research and policy
issues relating to human health	some plastics contain potentially harmful monomers and additive chemicals, including flame retardants and plasticizers adverse effects of additives evident in laboratory animals measurable levels of chemicals used as additives/monomers are present in the human population Canadian government declared BPA a toxic substance. USA National Toxicology Program expressed concern for adverse health effects	what are the effects of low-dose chronic exposure to chemicals or mixtures of chemicals used as plastic monomers or additives? dose–response curves may not be monotonic and so should not be extrapolated in risk assessment	independently conduct cumulative risk assessment/ management of plastic additives and monomers biomonitoring of body burdens of additives/monomers effects on susceptible subpopulations (babies, children) and on those with high-exposure risks evaluate effects of exposure to mixtures of additives/monomers design/validate appropriate species/protocols to assess chronic low dose exposures to additives/monomers by humans

public about their application (WRAP 2007). To gain the maximum benefit from degradable, biodegradable and compostable materials, it is, therefore, essential to identify specific uses that offer clear advantages and to refine national and international standards (e.g. EN 13432, ASTM D6400-99) and associated product labelling to indicate appropriate usage and appropriate disposal.

8. POLICY MEASURES

Our intention when preparing this Theme Issue was to focus on the science surrounding all aspects pertinent to plastics, the environment and human health. There are some omissions from the volume, such as input from social scientists on how best to convey relevant information to influence littering behaviour, consumer choice and engagement with recycling. These omissions aside, to be of greatest value the science herein needs to be communicated beyond a purely scientific audience (see recommendations in table 1). This is in part the role of a Theme Issue such as this, and the final invited contribution to the volume examines the science–policy interface with particular reference to policy relating to plastics. Shaxson (in press) considers this interface from the perspectives of industry, the scientist and the policymaker. She emphasizes the need for policy relating to plastic to weigh societal and economic benefits against environmental and health concerns. This is a diverse subject area that will require a range of policies to focus at specific issues, including polymer safety, material reduction, reuse, recycling, biopolymers, biodegradable and compostable polymers, littering, dumping and industrial spillage. There are a range of appropriate measures (National Research Council 2008) including information and recommendations (e.g. WRAP 2009), regulations (such as the Canadian Government restrictions on BPA in baby bottles), taxes (such as land fill tax, which incentivizes the diversion of waste from landfill to recycling), standards (such as EN 13432 covering compostable plastics) and allocation of funds for research, innovation and capacity building. However, the diversity of issues leads to an equally complex policy environment. In the UK, for example, there is not one, but many relevant policy interfaces and numerous policies. These activities are shared among several government departments, driven by national pressures, international obligations and European directives. In such a complex environment, even robust and clearly delivered information from the scientific community does not always have the most appropriate effects on the policy process.

Shaxson presents evidence from case studies on policies relating to plastic litter in the marine environment and land-based plastic waste. She indicates that many plastic-related polices fall into what are defined as unstructured or badly structured problems—in essence, problems that lack consensus and clarity in the relevant policy question and in some cases lack clarity in the relevant knowledge base to inform any decision. Shaxson suggests such circumstances will require a reflexive approach to brokering knowledge between industry, scientists and policymakers, and that scientists will need to be prepared to make and facilitate value judgements on the basis of best evidence. From a UK perspective, she advocates using the science within this volume to help develop a 'Plastics Road Map', similar to the recently completed Milk and Dairy Road Map (Defra 2008) to structure policy around plastics, the environment and human health and suggests that this be facilitated by appropriate and broad debate among relevant parties.

9. PLASTICS AND THE FUTURE

Looking ahead, we do not appear to be approaching the end of the 'plastic age' described by Yarsley and Couzens in the 1940s, and there is much that plastics can contribute to society. Andrady & Neal (2009) consider that the speed of technological change is increasing exponentially such that life in 2030 will be unrecognizable compared with life today; plastics will play a significant role in this change. Plastic materials have the potential to bring scientific and medical

advances, to alleviate suffering and help reduce mankind's environmental footprint on the planet (Andrady & Neal 2009). For instance, plastics are likely to play an increasing role in medical applications, including tissue and organ transplants; lightweight components, such as those in the new Boeing 787, will reduce fuel usage in transportation; components for generation of renewable energy and insulation will help reduce carbon emissions and smart plastic packaging will no doubt be able to monitor and indicate the quality of perishable goods.

In conclusion, plastics offer considerable benefits for the future, but it is evident that our current approaches to production, use and disposal are not sustainable and present concerns for wildlife and human health. We have considerable knowledge about many of the environmental hazards, and information on human health effects is growing, but many concerns and uncertainties remain. There are solutions, but these can only be achieved by combined actions (see summary table 1). There is a role for individuals, via appropriate use and disposal, particularly recycling; for industry by adopting green chemistry, material reduction and by designing products for reuse and/or end-of-life recyclability and for governments and policymakers by setting standards and targets, by defining appropriate product labelling to inform and incentivize change and by funding relevant academic research and technological developments. These measures must be considered within a framework of lifecycle analysis and this should incorporate all of the key stages in plastic production, including synthesis of the chemicals that are used in production, together with usage and disposal. Relevant examples of lifecycle analysis are provided by Thornton (2002) and WRAP (2006) and this topic is discussed, and advocated, in more detail in Shaxson (2009). In our opinion, these actions are overdue and are now required with urgent effect; there are diverse environmental hazards associated with the accumulation of plastic waste and there are growing concerns about effects on human health, yet plastic production continues to grow at approximately 9 per cent per annum (PlasticsEurope 2008). As a consequence, the quantity of plastics produced in the first 10 years of the current century will approach the total that was produced in the entire century that preceded.

We are indebted to James Joseph and Claire Rawlinson in the editorial office at the Royal Society. Without their guidance and patience this volume would not have been possible. We also thank Dr J. P. Myers, Environmental Health Sciences, for his help in preparing text for the section on Green Chemistry.

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PHILOSOPHICAL TRANSACTIONS OF THE ROYAL SOCIETY B

Phil. Trans. R. Soc. B (2009) **364**, 1–4
doi:10.1098/rstb.2009.0054

Introduction

Our plastic age

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Within the last few decades, plastics have revolutionized our daily lives. Globally we use in excess of 260 million tonnes of plastic per annum, accounting for approximately 8 per cent of world oil production. In this Theme Issue of *Philosophical Transactions of the Royal Society*, we describe current and future trends in usage, together with the many benefits that plastics bring to society. At the same time, we examine the environmental consequences resulting from the accumulation of waste plastic, the effects of plastic debris on wildlife and concerns for human health that arise from the production, usage and disposal of plastics. Finally, we consider some possible solutions to these problems together with the research and policy priorities necessary for their implementation.

Keywords: plastic; polymer; debris; endocrine disruption; phthalates; waste management

The term plastics applies to a wide range of materials that at some stage in manufacture are capable of flow such that they can be extruded, moulded, cast, spun or applied as a coating. Synthetic polymers are typically prepared by polymerization of monomers derived from oil or gas, and plastics are usually made from these by addition of various chemical additives. There are currently some 20 different groups of plastics, each with numerous grades and varieties (APME 2006). Plastics are incredibly versatile materials; they are inexpensive, lightweight, strong, durable, corrosion-resistant, with high thermal and electrical insulation properties. The diversity of polymers and the versatility of their properties facilitate the production of a vast array of plastic products that bring technological advances, energy savings and numerous other societal benefits (Andrady & Neal 2009). The first truly synthetic polymer, *Bakelite*, was developed by Belgian chemist Leo Baekeland in 1907, and many other plastics were subsequently developed over the next few decades. It was not until the 1940s and 1950s, however, that mass production of everyday plastic items really commenced. On the opening page of their book '*Plastics*', Yarsley & Couzens (1945; first published in 1941) consider that '*the possible applications* [of plastics] *are almost inexhaustible*'. At that time, global production was less than a million tonnes per annum, but plastics were already widely used in products ranging from cups and saucers to components for cars and aeroplanes. The final chapter of their book anticipates the ways that plastics will influence the life of someone born 70 years ago at the start of our 'plastic age'. (Yarsley & Couzens 1945)

> This [imaginary] plastic man will come into a world of colour and bright shining surfaces where childish hands find nothing to break, no sharp edges, or corners to cut or graze, no crevices to harbour dirt or germs .... The walls of his nursery, his bath ... all his toys, his cot, the moulded light perambulator in which he takes the air, the teething ring he bites, the unbreakable bottle he feeds from [all plastic]. As he grows he cleans his teeth and brushes his hair with plastic brushes, clothes himself with in plastic clothes, writes his first lesson with a plastic pen and does his lessons in a book bound with plastic. The windows of his school curtained with plastic cloth entirely grease- and dirt-proof ... and the frames, like those of his house are of moulded plastic, light and easy to open never requiring any paint.
>
> (Yarsley & Couzens 1945, p.149)

The text continues through extensive use of plastics for furniture and interior design, for beauty and leisure, in industry and in transport by road, sea and air. Until in old age plastic man:

> wears a denture with silent plastic teeth and spectacles with plastic lenses ... until at last he sinks into his grave in a hygienically enclosed plastic coffin
>
> (Yarsley & Couzens 1945, p.152)

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One contribution of 15 to a Theme Issue 'Plastics, the environment and human health'.

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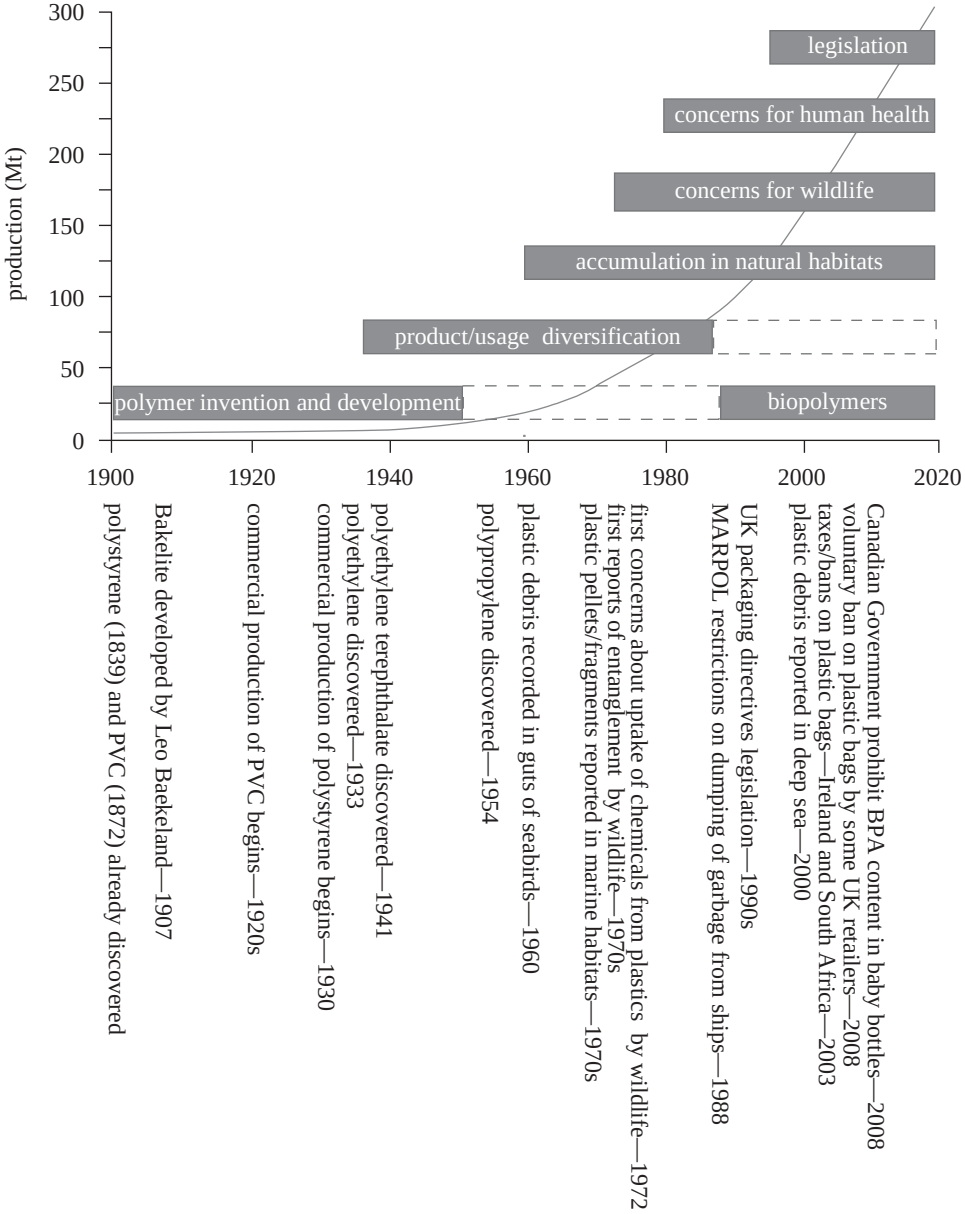

Figure 1. Summary illustrating historical stages in the development, production and use of plastics together with associated concerns and legislative measures (numerous sources). Solid red line shows plastic production in millions of tonnes (Mt). Reproduced with permission from APME (2006). BPA, bisphenol A; PVC, polyvinyl chloride.

This Theme Issue explores the evidence and the diversity of scientific opinion surrounding our use of plastics at the start of the twenty-first century. To set the present day perspective into context, we have included a historical overview summarizing the development and production of plastic, together with associated concerns, regulatory measures and some potential future trends (figure 1). Many of these topics are considered in detail within the Theme Issue, and we have included a selection of quotes from these papers to illustrate the diversity of subject matter, scientific opinions and conclusions therein (table 1).

The series of papers starts with a review of the history of polymer development together with some of their applications, past, present and future (Andrady & Neal 2009). This includes the use of lightweight plastic components in cars and aeroplanes to reduce fuel usage; the use of inexpensive plastic casings to make information technology and electrical goods far more readily accessible than would otherwise have been possible; and the use of plastics for sterile dressings and medical products. The most substantial use of plastics today, accounting for well over a third of production, is, however, for disposable items of packaging, most of which are discarded within a year or so of manufacture (Barnes *et al.* 2009; Hopewell *et al.* 2009).

The durability and increasing usage of plastics create a major waste management problem with plastic accounting for approximately 10 per cent of the waste we generate. Some of this is recycled, but a substantial proportion is disposed of to landfill (Barnes *et al.* 2009; Hopewell *et al.* 2009). A range of terms are used to describe the waste that is produced by modern society, these include trash, garbage, rubbish, litter and debris; usage varies according to the type and origin of the waste and according to regional differences in terminology. Usage of these terms is considered to be interchangeable in the papers within this volume; the

Table 1. Selected quotes reflecting the diversity of content and some of the scientific conclusions of authors in this Theme Issue. Phthalates, BPA, PBDE and tetrabromobisphenol A (TTBPA) are chemical additives, and in the case of BPA, a monomer used in the production of plastics.

'Any future scenario where plastics do not play an increasingly important role in human life therefore seems unrealistic' (Andrady & Neal 2009).
'One of the most ubiquitous and long-lasting recent changes to the surface of our planet is the accumulation and fragmentation of plastics' (Barnes *et al.* 2009).
'Monitoring is crucial to assess the efficacy of measures implemented to reduce the abundance of plastic debris, but it is complicated by large spatial and temporal heterogeneity in the amounts of plastic debris and by our limited understanding of the pathways followed by plastic debris and its long-term fate' (Ryan *et al.* 2009).
'The environmental, cultural, aesthetic, commercial and other problems arising from pelagic plastics in particular and varied marine debris items in general are manifold, widely acknowledged and often difficult to address' (Gregory 2009).
'As plastics production and usage continue to increase, particularly in economically developing countries, the environmental implications of their disposal should be carefully considered to avoid inadvertent release, magnification and transport of contaminants' (Teuten *et al.* 2009).
'Phthalates and BPA have been shown to affect reproduction in all studied animal groups, to impair development in crustaceans and amphibians and to induce genetic aberrations. Molluscs, crustaceans and amphibians appear to be especially sensitive to these compounds, and biological effects are observed at environmentally relevant exposures in the low ng l^{-1} to $\mu g\, l^{-1}$ range' (Oehlmann *et al.* 2009).
'PBDE and TTBPA have been shown to disrupt thyroid hormone homeostasis while PBDEs also exhibit anti-androgen action. Experimental investigations in animals indicate a wide variety of effects associated with exposure to these compounds, causing concern regarding potential risk to human health' (Talsness *et al.* 2009).
'Studies also are needed to identify the phthalate metabolites and BPA species relevant to human health, paying special attention to potentially vulnerable segments of the population (e.g. children, women of reproductive age, minorities)' (Koch & Calafat 2009).
'... small changes in hormone levels resulting from exposure may be of public health importance when considering the prevalence of exposure to plastic additives and endocrine disrupting compounds among entire populations' (Meeker *et al.* 2009).
'Around 4 per cent of world oil and gas production, a non-renewable resource, is used as feedstock for plastics and a further 3–4% is expended to provide energy for their manufacture. A major portion of plastic produced each year is used to make disposable items of packaging or other short-lived products that are discarded within a year of manufacture. These two observations alone indicate that our current use of plastics is not sustainable. In addition, because of the durability of the polymers involved, substantial quantities of discarded end-of-life plastics are accumulating as debris in landfills and in natural habitats worldwide. Recycling is one of the most important actions currently available to reduce these impacts and represents one of the most dynamic areas in the plastics industry today. Recycling provides opportunities to reduce oil usage, carbon dioxide emissions and the quantities of waste requiring disposal' (Hopewell *et al.* 2009).
'Bioplastic polymers have great potential to contribute to material recovery, reduction of landfill and use of renewable resources. Widespread public awareness of these materials and effective infrastructure for stringent control of certification, collection, separation and composting will be crucial to obtaining these benefits in full' (Song *et al.* 2009).
'... there is an opportunity to address many of these issues simultaneously by using the science in this issue to help develop an enhanced Road Map for policy around plastics, the environment and human health in the UK' (Shaxson 2009).
'... plastic production continues to grow at approximately 9 per cent per annum .... As a consequence, the quantity of plastics produced in the first 10 years of the current century will approach the total that was produced in the entire century that preceded.' (Thompson *et al.* 2009).

reader should focus on the types, sources, accumulation, disposal and effects of the waste.

Plastic debris has accumulated in natural habitats from the poles to the equator (Barnes *et al.* 2009); it is a very conspicuous component of the debris that is present in the marine environment, and most of the literature on the accumulation of plastic in the environment and the associated problems for wildlife has come from marine habitats (Gregory 2009). Monitoring represents an important step towards quantifying spatial and temporal trends in the abundance of all types of debris, including plastic. Numerous national and international schemes have been initiated to record quantities and categories (uses, sources, material types, sizes), and in some cases to facilitate debris removal (Ryan *et al.* 2009).

Substantial quantities of plastic debris already contaminate marine habitats from remote shorelines and inaccessible areas of the deep sea to heavily populated coastlines. The ubiquity of this debris in the marine environment has resulted in numerous accounts of species ingesting and becoming entangled in plastic. As a consequence of the durability of plastics, these encounters typically result in injury or impaired movement and can ultimately result in death (Gregory 2009). There is evidence that plastics are fragmenting in the environment and, as a consequence, will become available for ingestion by a wider range of organisms (Barnes *et al.* 2009). In addition to these physical effects, there has been speculation for over 30 years that the ingestion of plastic debris could lead to the transfer of toxic chemicals to wildlife. Recent publications have raised new concerns around this issue (Mato *et al.* 2001; Thompson *et al.* 2004; Arthur *et al.* 2009); Teuten *et al.* (2009) present a summary of current evidence together with new data on the accumulation of chemicals from plastic by wildlife.

In parallel with concerns for wildlife, there is a rapidly growing body of evidence relating to public health issues arising from current use of plastics. A range of chemicals are added to plastics during manufacture, to enhance the performance of plastics.

These additives can be referred to as plasticizers and include flame retardants, stabilizers, antioxidants and other chemicals such as antimicrobials that give each type of plastic unique properties. There is concern that potentially harmful chemical additives including phthalates, bisphenol A (BPA) and polybrominated diphenyl ethers (PBDE) could be transferred to humans directly from plastics, for example from flexible toys mouthed by toddlers, or indirectly, for example via food and drink that is packaged or transported via tubing in plastics containing these additives (e.g. Wagner & Oehlmann 2009). Evidence relating to this is considered from three perspectives: human body burdens of chemicals used in plastic manufacture (Koch & Calafat 2009); experimental studies on animals (Talsness *et al.* 2009) and the effects of these chemicals on humans (Meeker *et al.* 2009). These papers present stark evidence and consider some possible solutions.

Looking to the next few decades, it seems inevitable that humankind will become more reliant on plastics; therefore, the Theme Issue examines potential solutions for waste management of used plastics. For packaging applications, in particular, biodegradable plastics have been advocated as an approach that uses renewable biomass and/or facilitates decomposition at the end of a product's lifetime. Song *et al.* (2009) consider the potential applications and subsequent degradation of these materials, presenting new data on biodegradation in domestic composting conditions. The 'three Rs' (reduce, reuse and recycle) have been, extensively and lyrically (Johnson 2006; the three Rs: from the album Sing-A-Longs and Lullabies for the Film Curious Ge), advocated as solutions to the wasteful nature of our society. These strategies together with a 'fourth and fifth R' ('energy recovery' and 'molecular redesign') are considered by Hopewell *et al.* (2009) and Thompson *et al.* (2009), who describe current trends and examine the limitations to recycling of plastics.

Having considered the benefits of plastics, the problems associated with production and the use of plastics past and present and some solutions, the Theme Issue also examines the science–policy interface where appropriate directions will be determined by governments and where policies will be implemented to restrict activities and incentivize change (Shaxson 2009). Finally, the guest editors and contributors synthesize the work presented in the Theme Issue as a whole to give a summary of current understanding together with priorities for research, innovation and policy that are required to guide our future use of plastics in relation to the environment and human health (Thompson *et al.* 2009).

We are indebted to James Joseph and Claire Rawlinson in the editorial office at the Royal Society. Without their guidance and patience, this issue would not have been possible.

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ENVIRONMENT & ENERGY COMMISSION

City of Columbia & County of Boone
City Hall, Conference Room 1A

December 18, 2014

Mayor McDavid and Council Members,

The Environment and Energy Council has voted unanimously to foprward the attached report and proposed regulations concerning plastic bags. Please review the report and consider these recommendations.

Respectfully Yours,

Lawrence Lile, PE
Chair
Environment and Energy Commission

REPORT ON PLASTIC BAG RESTRICTIONS
ENVIRONMENT & ENERGY COMMISSION

City of Columbia & County of Boone
City Hall, Conference Room 1A

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Recommendations

The Environment and Energy Commission (EEC) recommends the City Council adopt an ordinance that restricts the use of plastic bags at grocery, pharmacy, and convenience stores. Main points of the proposed ordinance include:

- Stores cannot give out single-use plastic bags at the point-of-sale;
- Stores must charge a fee for paper bags at the point of sale;
- Stores may reward customers for bringing their own bags;
- The fee shall be waived for certain low income customers;
- Exemptions are allowed for restaurants, produce bags, and when bags are used for sanitation.

The full proposed ordinance starts on page 11 of this report.

Introduction & Purpose

This report addresses issues concerning plastic shopping bags from an environmental perspective.

Plastic is one of the most useful inventions of the 20th century. It is obvious that nearly every consumer product we buy these days has some plastic content, and many would be impossible without this technology. Plastic materials can be durable, inexpensive, and useful. Plastic is usually made out of some kind of nonrenewable resource, either oil or natural gas.

Using a piece of plastic once, for a few minutes, then throwing it away is a short-sighted way to waste an incredibly useful resource. Even if the bag is used again to pick up animal waste or line a trash can, the useful life of a plastic bag is quite short, squandering a nonrenewable resource. We believe that plastics should be reserved for uses that have a longer lifetime, preserving the nonrenewable resources they are generally made out of for a longer time.

Single use plastic bags have several particularly objectionable environmental negatives. The poet Harryette Mullen calls them "Urban Tumbleweeds". Like the tumbleweed, lightweight, single use plastic bags are carried by the wind to litter the landscape. The get entangled in bushes, trees, and fences. They show up as litter in rivers and streams, and entangle wildlife. They end up in ocean garbage gyres, where they are ingested by fish and birds. They are harmful when eaten by livestock and wildlife. They don't biodegrade, eventually photodegrading into smaller and smaller toxic bits, but do not disappear. In addition, scientific studies have not been able to show that chemicals used in the manufacture of plastics are safe for humans, and indicate the need to decrease the exposure to these chemicals (Anderson, Kuriyama, Taylor, Talsness, & vom Saal, 2009). Particles derived from the decomposition of plastic waste can enter the food chain of living organisms (Dorigo, 2011).

In addition, recycling rates for single use plastic bags is very low. According to the EPA, "only 9 percent of the total plastic waste generated in 2012 was recovered for recycling." The agency also states, ". . . in 2012, the category of plastics which includes bags, sacks, and wraps was recycled at about 12 percent." (EPA, n.d.) The City of Columbia does not accept grocery bags for recycling, because they don't have enough space to store enough plastic bags to recycle them in a cost-effective way. Solid waste customers in Columbia that mistakenly place plastic bags in their "blue bags" for recycling may

inadvertently cause machinery jams if the bags get caught in sorting machinery at the Material Recovery Facility. It is better to reduce the number of bags used to begin with.
The Environment & Energy Commission believes that the impact on retailers from the proposed legislation is small, that the mandatory bag fee compensates retailers fairly for their trouble, and will not reduce retail sales. We do not believe the convenience that single use bags provide is worth the associated environmental risks and costs, and unanimously recommends the adoption of an ordinance restricting the use of plastic bags.

Feature Analysis

Ordinances enacted in cities in the U.S. vary widely in feature and scope. The Environment & Energy Commission has examined many of these and determined that the following features are either unnecessary or ineffective:

- A plastic bag fee
  The Commission doesn't believe a fee alone will give enough incentive for consumers to stop using plastic bags. In fact most of the major grocery stores in the area already offer money back for customers using their own bags. This, for practical purposes, is already a plastic bag fee. We do not believe that implementing a mandatory bag fee without a ban would reduce the number of plastic bags used. In some jurisdictions consumers have become accustomed to the fee and any initial reduction in plastic bag use gradually rose back to previous levels. One of the major inducements for a ban on plastic bags is to reduce so called "white pollution" from bags blowing and ending up in aquatic environments. We believe that if bags are still distributed, a fee based ordinance doesn't do enough to get plastic bags out of the environment.

- A bag tax
  We do not recommend a bag tax. These type of ordinances require that retailers return a portion of the fee collected for both paper and plastic bags to a governing body. These funds are then used for cleanups, education and awareness programs, and even free reusable bag distribution. While admirable, we believe a tax adds complexity for retailers and legal ramifications for the city. An additional tax would not be welcomed by the public. The EEC is not familiar with the regulatory subtleties and restrictions on imposing a tax, and therefore did not pursue ordinances that included a tax.

- A phased implementation
  A phased implementation is not necessary. Some ordinances we examined included a phased implementation had 2-3 year timelines; too long, in our opinion. We could not see any benefit to allowing some stores a longer time to comply with the ordinance. A six month implementation seems sufficient for stores to change signage and procedures.

- An ordinance that affects all stores
  We discussed an ordinance that affects all retailers. The majority of plastic bags come from stores selling food items. One visit to a grocery store can net many bags compared to a store that sells only clothing or household items. We believe this scope will significantly reduce the number of bags in the environment, with the least amount of disruption to retailers. The EEC

also believes that in a few years' time, the ordinance can be amended to widen the scope, if needed.

- Allowing compostable plastic bags
  Compostable bags do not solve most of the problems caused by plastic bags. Some ordinances we examined allowed stores to distribute bags that claim to be compostable. Our research found that these types of bags are much higher in cost to retailers. It is difficult to distinguish a compostable bag from a biodegradable bag or a plain plastic bag. Although there may be ASTM regulations that define these bag types, they all look the same. While compostable plastic bags do eventually biodegrade, they still would be found in the environment as litter and pose an entanglement/strangling danger to livestock and wildlife until the bags biodegrade, which could take months or years. The Commission does not recommend an ordinance that allows stores to give compostable plastic bags to their customers.

- Allowing biodegradable plastic bags.
  Biodegradable bags are hard to define. Some ordinances we examined allowed stores to distribute bags that claim to be biodegradable. The Commission found that information indicating the labeling of these bags is frequently misleading. Bags that claim to be biodegradable are often really "fast-degrading" meaning there is an additional chemical added to the composition of these bags that make the plastic fall apart faster than normal. However, they still degrade into small bits of plastic – it just happens faster. Because of the points below, the Commission does not recommend passing an ordinance that allows stores to give so-called biodegradable bags. Included below is an analysis of the difference between fast-degrading and biodegradable plastic bags.

    - "Degradation" is the process of a material breaking down either a) from a more complex compound to a less complex compound, or, as often the case for plastic bags (#4 polyethylene), from larger pieces into smaller pieces.

    - "Fast-degrading" can mean faster breakdown into less complex compounds or into smaller pieces (reduction in mass) or both. Increased breakdown times to smaller pieces would likely be worse for the environment than conventional plastic bag degradation.

    - "Fast-degrading" plastics are also called "oxo-plastics." These plastics have additives that increase abiotic oxidative degradation with exposure to light and are not compostable. ASTM D6954 is the standard defining oxo-degradable plastics.

    - "Biodegradation" is the process by which organisms decompose biodegradable plastics into water, carbon dioxide and biomass via aerobic or anaerobic processes. Biodegradable = Compostable. ASTM D6400 – "Standard Specification for Labeling of Plastics Designed to be Aerobically Composted in Municipal or Industrial Facilities" governs these types of bags.

    - Some products marketed as biodegradable are not; they are fast-degrading. There is money in getting into the "biodegradable" market share. There may be issues with

policing these differences. As mentioned before, it may be hard to distinguish one from another.

- Biodegradable/compostable bags may provide less long-term impact to the environment. However, switching from conventional to biodegradable bags does not address the volume of bags entering the waste stream. This type of substitution also does not create consumer awareness or foster changes in behaviors that affect our natural environment. These bags may be just as bad as normal plastic bags in that they blow around the landfill, causing extra labor to pick them up, or get caught in material sorting machinery, causing breakdowns. They can still endanger wildlife, causing entanglements or being ingested, before they break down.

Public Education and Outreach

Although there is no language in the proposed ordinance requiring education and outreach, the EEC believes it is a necessary component to inform and educate the public prior to the effective date of the ordinance. One of the unintended consequences of bag regulation may be on out-of-town visitors, who may not bring shopping bags. We recommend that the City use outreach programs to provide reusable bags with a prominent City of Columbia logo to places that may be frequented by visitors, such as hotels, the airport, conferences, Mo-Ex, and so on. This will help these consumers enjoy their experience in Columbia.

Common Ordinance Structure

Nationally we are aware of 173 jurisdictions that have imposed regulations on single-use plastic bags. Of these, 153 ban plastic single use bags at the point-of sale; 108 impose both a ban and a fee; 45 ban plastic bags, but do not impose any kind of fee; and just 5 implement a bag fee but no ban. 88 have a 10c fee, which was most common, the lowest is 5c and the highest is $1.00 in Brownsville, Texas. As can be seen, the most common regulation by far is a plastic bag ban with a 10c fee for paper bags. This is the type of regulation recommended by the EEC.

Many jurisdictions exempt certain retailers or customers from bag bans and fees. Common exemptions include:

a. An exemption for WIC or consumers on other assistance programs
b. Exemptions for farmers markets
c. Exemptions for charities and non-profits
d. Exemptions for those retailers who can prove that implementing the ban will cause an undue hardship
e. An exemption for stores that are under a particular number of square feet
f. An exemption for stores that do not belong to a chain
g. Exemptions for dine-in and take-out restaurants
h. Exemptions for other types of plastic bags, such as bags used for
    i. unpackaged bulk items, such as fruit, vegetables, nuts, grains, candy

ii. small hardware items;
iii. frozen foods, meat, fish, whether packaged or not
iv. flowers, potted plants, or other items where dampness may be a problem
v. unwrapped prepared foods or bakery goods
vi. Pharmaceuticals
vii. laundry-dry cleaning
viii. garbage waste, pet waste, or yard waste
ix. Newspapers

An examination of cities and communities that have passed an ordinance banning single use plastic bags reveals four basic structures in these ordinances.

1. A partial (not all stores) ban of single use plastic bags, a fee for paper bags, and a fee for compostable plastic bags that meet the (ASTM) International Standard Specification for Compostable Plastics D6400 (commercially compostable)
"Partial" bans apply to a particular segment of the retail market, such as grocery/food stores to ban single use plastic bags, and mandates a nominal fee for paper bags. These bans can also include one or more of the following elements:

   a. A "reward" to customers who bring their own bags, usually 10 cents per bag used is returned to the customer
   b. A requirement that plastic bags have a minimum thickness
   c. A requirement that paper bags contain at least 40% recycled content.
   d. A requirement for stores to administer an in-store reuse program for customers to return clean plastic bags for reuse by customers.
   e. A requirement for stores to keep records of bag sales and numbers of customers brining their own bags, and a requirement to submit those statistics to the governing authority.
   f. A city sponsored bag recycling program or free reusable bag distribution program, or education program. These are funded by the paper bag fee; the retailer keeps part of the fee and submits the rest of the fee to the city to administer the program(s).
   g. Establishment of river clean-up fund with proceeds from bag "tax"

   Advantages of this ban: Though the scope of the ban is partial (does not affect all stores), it still produces a significant reduction in single-use bags. Fees for paper bags reduces incentive for their use and induces consumers to switch to reusable bags. This ban allows retailers to provide paper bags as a convenience to customers and allows an additional small revenue stream in the form of a fee. Many different optional features allow communities to create an ordinance customized for the community.

   Disadvantages of this ban: A partial ban is not as effective as a full ban in reducing the use of plastic bags. Compostable bags still have significant environmental negatives.

2. A full ban of single use plastic bags and a fee for paper bags, fee for compostable plastic bags that meet the (ASTM) International Standard Specification for Compostable Plastics D6400 (commercially compostable).
"Full" bans generally apply to all retailers, and can contain any combination of features or exceptions. In addition, the following elements may also be included:

a. A staged implementation: different types of stores implement the ban on different timelines. For example, grocery stores implement first, then six months later, convenience stores, then six months later all other retailers.
b. A staged timeline for implementation that gives smaller stores additional time to implement the ban.

Advantages of this ban: Full scope bans on single use plastic bags has the most profound effect at eradicating bags in the environment. Fees for paper bags and compostable bags reduces incentive for their use and induces consumers to switch to reusable bags. Many different optional features allows communities to create an ordinance customized for their community.

Disadvantages of this ban: Compostable bags still have significant environmental negatives.

3. A full ban of single-use plastic bags and a ban on paper bags that contain less than 40% recycled content. No fee.

"Full" ban generally applies to all retailers, with any of the other features or exceptions common to other types of bans.

Advantages of this ban: May be easier to get public support since no fees are collected.

Disadvantages of this ban: Grocery stores may object to extra costs. Most paper grocery bags are already being made with more than 40% recycled content. Without a fee for paper bags, an unintended consequence of this type of ban is that consumers may shift to using these 40% recycled paper bags rather than reusable bags.

4. A fee for single-use plastic carry-out bags and paper carry-out bags, but no ban

These bans allow single-use plastic bags and paper bags to be distributed for a fee, and generally applies to all retailers.

Advantages of this ban: Reduces the use of single use plastic bags somewhat. Allows consumers more choices. Quiets those consumers who claim they need bags for trash can liners and dog waste.

Disadvantages of this ban: After the initial drop in bag use, consumers may grow used to the fees for plastic bags and rebound. Bag use directly correlates to bag cost. Currently many stores already charge a 10c fee, with little effect on plastic bag use, so using a fee-only system doesn't change the status quo.

Scope of Bag Regulations

The scope of retailers to which a fee or ban applies is an important consideration. Chicago, for instance, has banned single-use plastic bags in essentially all stores. Other cities have passed regulations affecting only grocery stores or retailers above a certain physical size. Several examples have been gleaned from the example ordinances by other jurisdictions. These include:

Applying regulations to all retailers regardless of type or size

- Most effective at reducing single use plastic bags in the environment
- Consumers may see this as draconian; it may be harder for cities to gain public acceptance

Applying regulations to stores over a certain size

- Larger stores give out more bags. This type of scope, while not the most effective measure, would still reduce plastic bag use significantly.
- This can seem arbitrary and unfair, particularly to retailers near the cut-off.
- This type of scope works best when the ordinance contains additional options for bags, such as those that require retailers to offer compostable plastic bags in addition to paper bags .
- Applying the ordinance only to stores over a certain size relieves the financial burden on small businesses or farmer's market vendors that don't have the resources of larger businesses.

Applying regulations to stores that sell only certain types of goods, such as perishable foods.

- Because of the nature of these stores, they tend to give out the highest number of single use plastic bags. Other retailers, like clothiers and retailers of other soft goods, give heavier gauge plastic bags that stand up to more uses, and fewer of them. Limiting the ban to a segment like this is a good starting point to an eventual full ban at all retailers. It should be easier to gain public acceptance.
- Limiting the ban to stores that sell a certain type of goods, like perishable foods is not as effective as including all stores in a ban.

Exceptions for various types of goods for sanitation

- Produce bags, bulk food bags, meat wrapping, and materials are commonly excepted for health reasons. The EEC agrees that bags used for sanitation should be allowed.
- Other specific types of single use bags may be needed by specific industries. Regulations should be flexible enough to allow exceptions.

Common Objections to Bag Bans

Paper bags have a larger cost and carbon footprint than plastic bags

Framing this as a "Paper vs. Plastic" debate is missing the point. These regulations are meant to reduce the use of all types of single use shopping bags. Just switching from plastic to paper is not the intended result. In addition, this argument focuses on the cost to make the bags, but does not account for the environment damage done post-use by consumer which includes possible toxicity to soil and water, dangers to wildlife and livestock, litter or so-called "white pollution", costs to the city to pick up litter caused by plastic bags. Clearly, paper bags are the winner when looking at the environmental fate of bags distributed to consumers.

Plastic bags are recyclable, so what's the problem?

It is better to eliminate the production of a product, from an environmental standpoint, than to spend money and energy recycling it. Recycling rates on single use plastic bags are very low, although many consumers also reuse them at least once around the house for various uses. Plastic bags are not recycled by Columbia's recycling program. They can cause major problems with sorting equipment at the recycling facility. People who need a bag to pick up after their dog will still have many opportunities to get free bags at the produce counter, bulk counter, or stores not regulated by this proposal.

Paper bags cost the retailer more than paper

The 10 cent fee should be more than enough to compensate retailers for the cost of paper bags, which is about 6 cents per bag. Over time, as consumers get used to bringing their own bags, retailers should be able to reduce the amount of bags they purchase to supply to consumers.

Bag bans reduce retail sales

Some studies, with various levels of bias, have tended to show that some types of bans or fees reduce retail sales, causing shoppers to go elsewhere. The EEC has considered retail sales in analyzing this proposed legislation, and has determined that allowing paper bags at the point-of-sale would not affect retailers negatively, in fact this is the current policy at many Columbia stores already. Would shoppers go elsewhere? There are very few shopping opportunities within a short driving distance of the city limits. We do not expect many consumers to drive to Ashland, Fulton, or Boonville, with less choice in grocery stores, just to avoid a nominal bag fee. We believe that these regulations will have no effect on retail sales in Columbia in Columbia.

Bag fees hurt the poor

The Commission recommends that the proposal waive the fee for paper bags for those customers who are using public assistance funds, such as SNAP or WIC to make their purchases.

Bag fees annoy tourists and visitors

On the Columbia Visitors and Convention Bureau website, Columbia is touted as a "cool neighborhood. . . .with a young vibe". A tagline on the site says, "What you Unexpect". (About Columbia, n.d.) Columbia is like no other place in Missouri, and we advertise that fact, proudly.

A bag ban may turn out to be just another one of those things that makes visiting Columbia like visiting another country, and experiencing something new and unexpected. Visitors to Californian or European cities (who already ban plastic bags) would hardly complain about the shortage of plastic shopping bags. The City can and should aggressively promote reusable shopping bags with Columbia Missouri logos highlighted prominently on the outside, through hotels, the airport, Mo-Ex shuttles, new students, and any other avenue which may help out-of-town visitors enjoy their experience in Columbia.

Bag fees are a tax

The fee in the proposed ordinance is a charge for the supplied paper bag, i.e., cash for goods, and not a tax. Other cities have opted to require retailers to return a portion of the fee to a specific city department to fund education efforts about their bag ban and to fund a free reusable bag program. We do not have sufficient expertise in tax law to determine if this would be possible or not. The Environment & Energy Commission has no opinion on this option, which would in essence be a tax, if adopted, but is not included in the ordinance, as currently proposed.

Environment & Energy Commission Proposed Ordinance

The Environment & Energy Commission recommends the following ordinance be considered by the City Council.

WHEREAS, the City of Columbia seeks to protect our waterways, our local streams including Hinkson Creek, the Missouri River, and ultimately our oceans; and

WHEREAS, single-use plastic carryout bags pollute our local streams and rivers; these bags ultimately break down into smaller bits that contaminate soil and waterways and enter into the food supply that humans, animals, and aquatic life ingest; and

WHEREAS, the City Council of the City of Columbia seeks to reduce the toxicity of waste materials in the solid waste stream that are directed to resource recovery and sanitary landfill facilities, and to maximize the removal of plastic carryout bags from the waste stream; and

WHEREAS, the City of Columbia signed Resolution 160-06A, endorsing the Mayors Climate Protection Agreement striving to meet or exceed Kyoto Protocol targets for reducing global warming pollution by taking actions in our community; and

WHEREAS, the City of Columbia has shown an interest in being a leader in initiatives with a positive effect on climate change through its curbside recycling program, methane gas bioenergy facility, solar energy arrays, LEED certified city buildings, increased use of renewable energy; and

WHEREAS, the production of plastic bags worldwide requires the use of more than 12 million barrels of oil per year, which also has a significant environmental impact; and

WHEREAS, many chemicals in plastic products are now known to cause harm, but the chemicals present in plastic bags are not disclosed to the public and are not required to be tested for health effects, so chemicals in plastic bags cannot be assumed to be safe.

BE IT ORDAINED BY THE CITY COUNCIL OF THE CITY OF COLUMBIA:

The Municipal Code of Columbia is hereby amended by inserting a new ordinance as follows:

SINGLE-USE PLASTIC BAG ORDINANCE

Definitions.
As used in this ordinance:

"Customer" means any person purchasing goods from a store.

"Operator" means the person in control of, or having the responsibility for, the operation of a store, which may include, but is not limited to, the owner of the store.

"Single-use plastic bag" means any bag made predominantly of plastic which is provided to a customer at the point of sale.

"Single-use plastic bag" includes

1) Degradable plastic bags, and
2) Biodegradable plastic bags

"Single-use plastic bag" does not include

1. Reusable bags,
2. Produce bags, or product bags.

"Degradable" plastic is plastic that

1. Breaks down into less complex compounds from a more complex compound or
2. Contains additives that increase degradation with exposure to light to cause faster breakdown times or
3. Breaks down from larger into smaller pieces

"Biodegradable" plastic is plastic that

1. Decomposes into water, carbon dioxide and biomass via aerobic or anaerobic processes and
2. Adheres to ASTM D6400 – "Standard Specification for Labeling of Plastics Designed to be Aerobically Composted in Municipal or Industrial Facilities"

"Produce bag" or "product bag" means any bag without handles used exclusively to carry produce, meats, or other food items to the point of sale inside a store.

"Recyclable" means material that can be sorted, cleansed, and reconstituted using available recycling collection programs for the purpose of using the altered form in the manufacture of a new product. As used in this definition, the term "recycling" does not include burning, incinerating, converting, or otherwise thermally destroying solid waste.

"Recyclable paper bag" means a paper bag that:

1) contains no old growth fiber
2) contains a minimum of forty percent (40%) post-consumer recycled material

"Post-consumer recycled material" means a material that would otherwise be destined for solid waste disposal, having completed its intended end use and product life cycle. "Post-consumer recycled material" does not include materials and byproducts generated from, and commonly reused within, an original manufacturing and fabrication process.

"Reusable bag" means a bag with handles that is specifically designed and manufactured for reuse and meets all of the following requirements:

1) is machine washable or is made from a material that can be cleaned or disinfected;
2) does not contain lead, cadmium, or any other heavy metal in toxic amounts;
3) if made of plastic, is a minimum of at least 2.25 mils thick.

"Store" means a grocery, drug, convenience, or retail establishment which includes any of the following:

1. A self-service retail store that sells a line of dry grocery, canned goods, or non-food items and some perishable items
2. A store that sells a limited line of goods that includes milk, bread, soda, or perishable snack foods.

Section 1: Single-use plastic bags prohibited.
No store shall provide a single-use plastic bag to any customer for the purpose of enabling the customer to carry away goods from the point of sale.

Section 2: Exemptions
The use of plastic bags is allowed in the following circumstances:

1. For carryout from dine-in or take-out restaurants; and
2. For purposes of sanitation.

Section 3: Types of carryout bags allowed.
Each operator may provide reusable bags, or recyclable paper bags, or any combination thereof, to customers for the purpose of enabling the customer to carry away goods from the point of sale. This section does not apply if the operator does not provide carryout bags of any type to its customers.

Section 4: Fee for recyclable bags
Operators shall charge at least 10 cents for each recyclable paper bag provided to a customer. Store receipt will show recyclable paper bag count and charges on the printed receipt provided to the customer. Customers using food assistance programs such as Women, Infant, and Children (WIC) and Supplemental Nutrition Assistance Program (SNAP) at the point of sale are not required to pay the fee for paper bags at checkout.

Section 5: Customer Reward Programs
Retailers that reward customers for supplying their own carry out bags with a rebate or charitable contribution may continue.

Section 6: Operative date
This ordinance shall take effect six (6) months after adoption.

Section 7: Permitted bags
Nothing in this ordinance shall be construed to prohibit customers from using bags of any type that the customer brings to the store.

Section 8: Penalties and enforcement
Any store that violates Section 1 shall be subject to a fine of $500 for each offense. Each day that a violation continues shall constitute a separate and distinct offense.

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REPORT ON PLASTIC BAG RESTRICTIONS
APPENDIX
ENVIRONMENT & ENERGY COMMISSION

City of Columbia & County of Boone
City Hall, Conference Room 1A

Appendix A - Example Ordinances

These ordinances have been passed by other cities, and also serve as typical examples of the various approaches to this problem that have been used by other jurisdictions.

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ORDINANCE NO. ORD-11-0009

AN ORDINANCE OF THE CITY COUNCIL OF THE CITY OF LONG BEACH AMENDING THE LONG BEACH MUNICIPAL CODE BY ADDING CHAPTER 8.62 RELATING TO REGULATING THE USE OF PLASTIC CARRYOUT BAGS AND RECYCLABLE PAPER CARRYOUT BAGS AND PROMOTING THE USE OF REUSABLE BAGS

The City Council of the City of Long Beach ordains as follows:

Section 1. Chapter 8.62 is added to the Long Beach Municipal Code to read as follows:

Chapter 8.62

PLASTIC CARRYOUT BAGS

8.62.010 Definitions.

A. "Customer" means any person or persons, purchasing goods from a store.

B. "Farmer's Market" has the meaning provided in Subsection 3.80.180.G of the Long Beach Municipal Code.

C. "Operator" means the person in control of, or having the responsibility for, the operation of a store, which may include, but is not limited to, the owner of the store.

D. "Person" means any natural person, firm, corporation, partnership, or other organization or group however organized.

E. "Plastic carryout bag" means any bag made predominantly of plastic derived from either petroleum or a biologically-based source, such

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as corn or other plant sources, which is provided to a customer at the point of sale. Plastic carryout bag includes compostable and biodegradable bags but does not include reusable bags, produce bags or product bags.

F. "Postconsumer recycled material" means a material that would otherwise be destined for solid waste disposal, having completed its intended end use and product life cycle. Postconsumer recycled material does not include materials and by-products generated from, and commonly reused within, an original manufacturing and fabrication process.

G. "Produce bag" or "product bag" means any bag without handles used exclusively to carry produce, meats, or other food items to the point of sale inside a store or to prevent such food items from coming into direct contact with other purchased items.

H. "Recyclable" means material that can be sorted, cleansed, and reconstituted using available recycling collection programs for the purpose of using the altered form in the manufacture of a new product. Recycling does not include burning, incinerating, converting, or otherwise thermally destroying solid waste.

I. "Recyclable paper carryout bag" means a paper bag that meets all of the following requirements: (1) contains no old growth fiber; (2) is one hundred percent (100%) recyclable overall and contains a minimum of forty percent (40%) postconsumer recycled material; (3) is capable of composting, consistent with the timeline and specifications of the American society of Testing and Materials (ASTM) Standard D6400; (4) is accepted for recycling in curbside programs in the City; (5) has printed on the bag the name of the manufacturer, the location where the bag was manufactured, and the percentage of postconsumer recycled material used; and (6) displays the word "Recyclable" in a highly visible manner on the outside of the bag.

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J. "Reusable bag" means a bag with handles that is specifically designed and manufactured for multiple reuse and meets all of the following requirements: (1) has a minimum lifetime of one hundred twenty-five (125) uses, which for purposes of this Section, means the capability of carrying a minimum of twenty-two (22) pounds one hundred twenty-five (125) times over a distance of at least one hundred seventy-five (175) feet; (2) has a minimum volume of fifteen (15) liters; (3) is machine washable or is made from a material that can be cleaned or disinfected; (4) does not contain lead, cadmium, or any other heavy metal in toxic amounts, as defined by applicable state and federal standards and regulations for packaging or reusable bags; (5) has printed on the bag, or on a tag that is permanently affixed to the bag, the name of the manufacturer, the location where the bag was manufactured, a statement that the bag does not contain lead, cadmium, or any other heavy metal in toxic amounts, and the percentage of postconsumer recycled material used, if any; and (6) if made of plastic, is a minimum of at least 2.25 mils thick.

K. "Store" means any of the following retail establishments located within the City of Long Beach:

1. A full-line, self-service retail store with gross annual sales of two million dollars ($2,000,000), or more, that sells a line of dry grocery, canned goods, or nonfood items and some perishable items;

2. A store of at least ten thousand (10,000) square feet of retail space that generates sales or use tax pursuant to the Bradley-Burns Uniform Local Sales and Use Tax Law (Part 1.5 (commencing with Section 7200) of Division 2 of the Business and Professions Code) and that has a pharmacy licensed pursuant to Chapter 9 (commencing with Section 4000) of Division 2 of the Business and Professions Code; or

3. A drug store, pharmacy, supermarket, grocery store,

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convenience food store, food mart, or other entity engaged in the retail sale of a limited line of goods that includes milk, bread, soda, and snack foods, including those stores with a Type 20 or 21 license issued by the Department of Alcoholic Beverage Control.

8.62.020 Plastic carryout bags prohibited.

A. No store shall provide to any customer a plastic carryout bag.

B. This prohibition applies to bags provided for the purpose of carrying away goods from the point of sale and does not apply to produce bags or product bags.

C. No person shall distribute a plastic carryout bag or any paper bag at the Long Beach Farmers' Markets, except produce bags or product bags.

8.62.030 Permitted bags.

All stores shall provide or make available to a customer only recyclable paper carryout bags or reusable bags for the purpose of carrying away goods or other materials from the point of sale, subject to the terms of this Chapter. Nothing in this Chapter prohibits customers from using bags of any type that they bring to the store themselves or from carrying away goods that are not place in a bag, in lieu of using bags provided by the store.

8.62.040 Regulation of recyclable paper carryout bags.

A. Any store that provides a recyclable paper carryout bag to a customer must charge the customer ten cents ($0.10) for each bag provided, except as otherwise provided in this Chapter.

B. No store shall rebate or otherwise reimburse a customer any

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portion of the ten cent ($0.10) charge required in Subsection A, except as otherwise provided in this Chapter.

C. All stores must indicate on the customer receipt the number of recyclable paper carryout bags provided and the total amount charged for the bags.

D. All monies collected by a store under this Chapter will be retained by the store and may be used only for the following purposes:

1. Costs associated with complying with the requirements of this Chapter;

2. Actual costs of providing recyclable paper carryout bags; or

3. Costs associated with a store's educational materials or education campaign encouraging the use of reusable bags, if any.

E. All stores shall keep complete and accurate records or documents, for a minimum period of three years from the date of sale, of the total number of recyclable paper carryout bags provided, the total amount of monies collected for providing recyclable paper carryout bags, which record shall be available for inspection at no cost to the City during regular business hours by any City employee authorized to enforce this Section. Unless an alternative location or method of review is mutually agreed upon, the records or documents shall be available at the store's address. A responsible agent or officer of the store shall confirm that the information provided is accurate and complete.

F. The provision of false information, including incomplete records or documents, to the City shall be a violation of this Section, and such store shall be subject to the fines set forth in Section 8.62.080.

8.62.050 Use of reusable bags.

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A. All stores must provide reusable bags to customers, either for sale or at no charge.

B. Each store is strongly encouraged to educate its staff to promote reusable bags and to post signs encouraging customers to use reusable bags.

C. Each store is strongly encouraged to provide to the City and to the public a summary of any efforts a store has undertaken to promote the use of reusable bags by customers.

8.62.060 Exempt customers.

All stores must provide at the point of sale, free of charge, either reusable bags or recyclable paper carryout bags or both, at the store's option, to any customer participating either in the California Special Supplemental Food Program for Women, Infants, and Children pursuant to Article 2 (commencing with Section 123275) of Chapter 1 of Part 2 of Division 106 of the Health and Safety Code or in the Supplemental Food Program pursuant to Chapter 10 (commencing with Section 15500) of Part 3 of Division 9 of the Welfare and Institutions Code.

8.62.070 Operative date.

This Chapter shall become operative on August 1, 2011, for stores defined in Subsections K.1. and K.2. of Section 8.62.010. For stores defined in Subsection K.3. of Section 8.62.010, this Chapter shall become operative on January 1, 2012. For Farmers' Markets as defined in Subsection B of Section 8.62.010, this Chapter shall become operative on January 1, 2012.

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8.62.080 Enforcement and violation – penalty.

A. The City Manager has primary responsibility for enforcement of this Chapter. The City Manager or designee is authorized to promulgate regulations and to take any and all other actions reasonable and necessary to enforce this Chapter, including, but not limited to, investigating violations, issuing fines and entering the premises of any store during business hours. If the City Manager or designee determines that a violation of this Chapter has occurred, he/she will issue a written warning notice to the operator of a store that a violation has occurred and the potential penalties that will apply for future violations.

B. Any store that violates or fails to comply with any of the requirements of this Chapter after a written warning notice has been issued for that violation shall be guilty of an infraction.

C. If a store has subsequent violations of this Chapter that are similar in kind to the violation addressed in a written warning notice, the following penalties will be imposed and shall be payable by the operator of the store:

1. A fine not exceeding one hundred dollars ($100) for the first violation after the written warning notice is given;

2. A fine not exceeding two hundred dollars ($200) for the second violation after the written warning notice is given; or

3. A fine not exceeding five hundred dollars ($500) for the third and any subsequent violations after the written warning notice is given.

D. A fine shall be imposed for each day a violation occurs or is allowed to continue.

E. All fines collected pursuant to this Chapter shall be used to assist with the implementation and enforcement of the requirements of this Chapter.

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F. Any store operator who receives a written warning notice or fine may request an administrative review of the accuracy of the determination or the propriety of any fine issued, by filing a written notice of appeal with the City Manager no later than thirty (30) days after receipt of a written warning notice or fine, as applicable. The notice of appeal must include all facts supporting the appeal and any statements and evidence, including copies of all written documentation and a list of any witnesses that the appellant wishes to be considered in connection with the appeal. The appeal will be heard by a hearing officer designated by the City Manager. The hearing officer will conduct a hearing concerning the appeal within forty-five (45) days from the date that the notice of appeal is filed, or on a later date if agreed upon by the appellant and the City, and will give the appellant ten (10) days prior written notice of the date of the hearing. The hearing officer may sustain, rescind, or modify the written warning notice or fine, as applicable, by written decision. The hearing officer will have the power to waive any portion of the fine in a manner consistent with the decision. The decision of the hearing officer is final and effective on the date of service of the written decision, is not subject to further administrative review, and constitutes the final administrative decision.

8.62.090 Severability.

If any section, subsection, sentence, clause, or phrase of this ordinance is for any reason held to be invalid by a decision of any court of competent jurisdiction, that decision will not affect the validity of the remaining portions of the ordinance.

8.62.100 No conflict with federal or state law.

Nothing in this ordinance is intended to create any requirement,

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power or duty that is in conflict with any federal or state law.

Section 2. The City Clerk shall certify to the passage of this ordinance by the City Council and cause it to be posted in three (3) conspicuous places in the City of Long Beach, and it shall take effect on the thirty-first (31st) day after it is approved by the Mayor.

I hereby certify that the foregoing ordinance was adopted by the City Council of the City of Long Beach at its meeting of May 24, 2011, by the following vote:

Ayes: Councilmembers: Garcia, Lowenthal, DeLong, Johnson, Neal.

Noes: Councilmembers: O'Donnell, Schipske, Gabelich.

Absent: Councilmembers: Andrews.

City Clerk

Approved: 6/3/11 (Date)

Mayor

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Chicago, IL

City of Chicago

Office of the City Clerk

Document Tracking Sheet

SO2014-1521

Meeting Date:	3/5/2014
Sponsor(s):	Moreno (1) Cardenas (12) Austin (34) O'Shea (19)
Type:	Ordinance
Title:	Amendment of Municipal Code Chapter 11-4 by adding Article XXIII to prohibit retail establishments from providing customers with plastic carryout bags from waste stream
Committee(s) Assignment:	Committee on Health and Environmental Protection

SUBSTITUTE

ORDINANCE

WHEREAS, The City Council of the City of Chicago seeks to reduce the toxicity of waste materials in the solid waste stream that are directed to resource recovery and sanitary landfill facilities, and to maximize the removal of plastic carryout bags from the waste stream; and

WHEREAS, The City Council finds that plastic carryout bags are not biodegradable, which means that these bags ultimately break down into smaller bits that contaminate soil and waterways and enter into the food supply that animals and marine life ingest; and

WHEREAS, The production of plastic bags worldwide requires the use of more than 12 million barrels of oil per year, which also has a significant environmental impact; now, therefore,

BE IT ORDAINED BY THE CITY COUNCIL OF THE CITY OF CHICAGO:

SECTION 1. Chapter 11-4 of the Municipal Code of Chicago is hereby amended by inserting a new Article XXIII, as follows:

ARTICLE XXIII. RETAIL BAG USE

11-4-4000 Definitions.

As used in this Article XXIII:

"Chain store organization" means (1) three or more stores having common ownership, or (2) any store, regardless of ownership, that is part of a franchise.

"Commissioner" means the commissioner of health of the City of Chicago.

"Compostable plastic bag" means any bag that (1) when placed with other biodegradable or decaying natural materials will be broken down into simple organic components by biological processes, and (2) meets the current American Society for Testing and Materials (ASTM) D6400 Standard Specification for Compostable Plastics, and (3) has been certified as a compostable product by the Biodegradable Products Institute ("BPI") or other independent, testing organization approved by the commissioner, as evidenced by the presence on the bag of the BPI

Commercial Logo or other logo approved by the commissioner, indicating that the bag is "commercially compostable" or "compostable in an industrial composting facility."

"Customer(s)" means any person purchasing goods from a store.

"Operator" means the person in control of, or having the responsibility for, the operation of a store, which may include, but is not limited to, the owner of the store.

"Plastic carryout bag(s)" means any bag made predominantly of plastic derived from either petroleum or a biologically based source, such as corn or other plant sources, which is provided to a customer at the point of sale. The term "plastic carryout bag" includes (1) degradable plastic bags, and (2) biodegradable plastic bags that are not commercially compostable. The term "plastic carryout bag" does not include (i) commercially compostable plastic bags, (ii) reusable bags, (iii) produce bags, or (iv) product bags. As used in this definition: The term "produce bag" or "product bag" means any bag without handles used exclusively to carry produce, meats or other food items to the point of sale inside a store or to prevent such food items from coming into direct contact with other purchased items.

"Recyclable" means material that can be sorted, cleansed, and reconstituted using available recycling collection programs for the purpose of using the altered form in the manufacture of a new product. As used in this definition, the term "recycling" does not include burning, incinerating, converting or otherwise thermally destroying solid waste.

"Recyclable paper bag(s)" means a paper bag that meets all of the following requirements: (1) contains no old growth fiber; (2) is one hundred percent (100%) recyclable overall and contains a minimum of forty percent (40%) post-consumer recycled material; (3) is capable of composting, consistent with the timeline and specifications of the American Society for Testing and Materials (ASTM) D6400 Standard Specification for Compostable Plastics; (4) is accepted for recycling in curbside programs in the county; (5) has printed on the bag the name of the manufacturer, the location (country) where the bag was manufactured, and the percentage of post-consumer recycled material used; and (6) displays the word "Recyclable" in a highly visible manner on the outside of the bag.

"Post-consumer recycled material" means a material that would otherwise be destined for solid waste disposal, having completed its intended end use and product life cycle. The term "post-consumer recycled material" does not include materials and by-products generated from, and commonly reused within, an original manufacturing and fabrication process.

"Reusable bag" means a bag with handles that is specifically designed and manufactured for multiple reuse and meets all of the following requirements: (1) has a minimum lifetime of

125 uses which, for purposes of this definition, means the capability of carrying a minimum of 22 pounds125 times over a distance of at least 175 feet; (2) has a minimum volume of 15 liters; (3) is machine washable or is made from a material that can be cleaned or disinfected; (4) does not contain lead, cadmium, or any other heavy metal in toxic amounts; (5) has printed on the bag, or on a tag that is permanently affixed to the bag, the name of the manufacturer, the country where the bag was manufactured, a statement that the bag does not contain lead, cadmium, or any other heavy metal in toxic amounts, and the percentage of postconsumer recycled material used, if any; and (6) if made of plastic, is a minimum of at least 2.25 mils thick.

"Store" means a retail establishment, other than an establishment set forth in Section 11-4-4010, that sells perishable or non-perishable goods, including, but not limited to, clothing, food and personal items, directly to the customer.

11-4-4010 Exemptions.

This Article XIII shall not apply to (1) any dine-in or take-out restaurant, or (2) any store that is not a chain store organization.

11-4-4020 Plastic carryout bags prohibited.

No store shall provide a plastic carryout bag to any customer for the purpose of enabling the customer to carry away goods from the point of sale.

11-4-4030 Types of carryout bags required.

Each operator shall provide reusable bags, recyclable paper bags or commercially compostable plastic bags, or any combination thereof, to customers for the purpose of enabling the customer to carry away goods from the point of sale. Provided, however, that this section shall not apply if the operator does not provide carryout bags of any type to its customers.

11-4-4040 Permitted bags.

Nothing in this Article XXIII shall be construed to prohibit customers from using bags of any type that the customer brings to the store for their own use or from carrying away from the store goods that are not placed in a bag provided by the store.

11-4-4050 Penalties - Enforcement.

(a) Any person who violates Section 11-4-4020 shall be subject to a fine of not less than $300.00 nor more than $500.00 for each offense. Each day that a violation continues shall constitute a separate and distinct offense.

(b) Any person who violates Section 11-4-4030 shall be subject to a fine of not less than $100.00 nor more than $300.00 for each offense. Each day that a violation continues shall constitute a separate and distinct offense.

(c) The department of public health and department of business affairs and consumer protection are authorized to enforce this section.

SECTION 2. Section 7-30-020 of the Municipal Code of Chicago is hereby amended by inserting the language underscored, as follows:

7-30-020 Recycling program requirements.

(a) ~~Every~~ Except as otherwise provided in subsection (e) of this section, every operator shall establish an in-store recycling program that shall include, but need not be limited to, the following:

(Omitted text is unaffected by this ordinance)

(b) ~~Each~~ Except as otherwise provided in subsection (e) of this section, each operator or its designee shall maintain a copy of the annual report submitted to the Department in accordance with the requirements of Section 7-30-020(c).

(c) ~~Each~~ Except as otherwise provided in subsection (e) of this section, each operator or its designee shall submit an annual report, sworn by an affidavit, to the Department covering the preceding calendar year, beginning with a report covering calendar year two thousand nine (2009) which shall state the following:

(Omitted text is unaffected by this ordinance)

(e) This section shall not apply to the operator of any store meeting the requirements set forth in Sections 11-4-020 and 11-4-030.

SECTION 3. Upon its passage and publication, this ordinance shall take full force and effect, as follows:

(a) With regard to individual stores subject to this ordinance with a floor area whose square footage exceeds 10,000 feet, this ordinance shall take full force and effect on August 1, 2015.

(b) With regard to individual stores subject to this ordinance with a floor area whose square footage is 10,000 feet or less, this ordinance shall take full force and effect on August 1, 2016.

Alderman George A. Cardenas, 12th Ward	Alderman Carrie Austin, 34th Ward
Alderman Joe Moreno, 1st Ward	Alderman Matthew O'Shea, 19th Ward

ORDINANCE NO. 20498

AN ORDINANCE ENCOURAGING THE USE OF REUSABLE BAGS, BANNING SINGLE-USE PLASTIC CARRYOUT BAGS, AND ADDING SECTIONS 6.850, 6.855, 6.860, AND 6.865 TO THE EUGENE CODE,1971.

THE CITY OF EUGENE DOES ORDAIN AS FOLLOWS:

Section 1. Sections 6.850, 6.855, 6.860, and 6.865 of the Eugene Code, 1971, are added to provide as follows:

6.850 Plastic Bag Use - Definitions. For purposes of sections 6.850 to 6.865 of this code, the following words and phrases mean:

ASTM standard. The American Society for Testing and Materials (ASTM)'s International D-6400.

Carryout bag. Any bag that is provided by a retail establishment at the point of sale to a customer for use to transport or carry away purchases, such as merchandise, goods or food, from the retail establishment. "Carryout bag" does not include:

(a) Bags used by consumers inside retail establishments to:
  1. Package bulk items, such as fruit, vegetables, nuts, grains, candy or small hardware items;
  2. Contain or wrap frozen foods, meat, fish, whether packaged or not;
  3. Contain or wrap flowers, potted plants, or other items where dampness may be a problem;
  4. Contain unwrapped prepared foods or bakery goods; or
  5. Pharmacy prescription bags;

(b) Laundry-dry cleaning bags or bags sold in packages containing multiple bags intended for use as garbage waste, pet waste, or yard waste bags;

(c) Product bags.

City sponsored event. Any event organized or sponsored by the city or any department of the city.

Customer. Any person obtaining goods from a retail establishment or from a vendor.

Food provider. Any person in the city that provides prepared food for public consumption on or off its premises and includes, without limitation, any retail establishment, shop, sales outlet, restaurant, grocery store, delicatessen, or catering truck or vehicle.

Grocery store. Any retail establishment that sells groceries, fresh, packaged, canned, dry, prepared or frozen food or beverage products and similar items and includes supermarkets, convenience stores, and gasoline stations.

Pharmacy. A retail use where the profession of pharmacy by a pharmacist licensed by the state of Oregon's Board of Pharmacy is practiced and where prescription medications are offered for sale.

Product bag. Any bag provided to a customer for use within a retail establishment to assist in the collection or transport of products to the point of sale within the retail establishment. A product bag is not a carryout bag.

Recyclable paper bag. A paper bag that meets all of the following requirements:
(a) Is 100% recyclable and contains a minimum of 40% recycled content;
(b) Is capable of composting consistent with the timeline and specifications of the ASTM Standard as defined in this section.

Retail establishment. Any store or vendor located within or doing business within the geographical limits of the city that sells or offers for sale goods at retail.

Reusable bag. A bag made of cloth or other material with handles that is specifically designed and manufactured for long term multiple reuse and meets all of the following requirements:
(a) If cloth, is machine washable; or
(b) If plastic, has a minimum plastic thickness of 4.0 mils.

Vendor. Any retail establishment, shop, restaurant, sales outlet or other commercial establishment located within or doing business within the geographical limits of the city, which provides perishable or nonperishable goods for sale to the public.

Single-use plastic carryout bag. Any plastic carryout bag made available by a retail establishment to a customer at the point of sale. It does not include reusable bags, recyclable paper bags, or product bags.

Undue hardship. Circumstances or situations unique to the particular retail establishment such that there are no reasonable alternatives to single-use plastic carryout bags or a recyclable paper bag pass-through cannot be collected.

6.855 **Plastic Bag Use - Regulations**. Except as exempted in section 6.865 of this code:
(a) No retail establishment shall provide or make available to a customer a single-use plastic carryout bag;
(b) No person shall distribute a single-use plastic carryout bag at any city facility, city managed concession, city sponsored event, or city permitted event.

6.860 **Plastic Bag Use - Cost Pass-Through.** When a retail establishment makes a recyclable paper bag available to a customer at the point of sale pursuant to section 6.865(b) of this code, the retail establishment shall:
(a) Charge the customer a reasonable pass-through cost of not less than 5 cents per recyclable paper bag provided to the customer; and
(b) Indicate on the customer's transaction receipts the total amount of the recyclable paper bag pass-through charge.

6.865 **Plastic Bag Use - Exemptions.** Notwithstanding sections 6.855 and 6.860 of this code:
(a) Single-use plastic carryout bags may be distributed to customers by food providers for the purpose of safeguarding public health and safety during the

transportation of hot prepared take-out foods and prepared liquids intended for consumption away from the food provider's premises.

(b) Retail establishments may distribute product bags and make reusable bags available to customers whether through sale or otherwise.

(c) A retail establishment shall provide a reusable bag or a recyclable paper bag at no cost at the point of sale upon the request of a customer who uses a voucher issued under the Women, Infants and Children Program established in the Oregon Health Authority under ORS 409.600.

(d) Vendors at retail fairs such as a farmers' market or holiday fair are not subject to indicating on the customer's transaction receipt the total amount of the recyclable paper bag pass through charge required in section 6.860(b) of this code.

(e) The city manager or the designee may exempt a retail establishment from the requirement set forth in sections 6.855-6.860 of this code for a period of not more than one year upon the retail establishment showing, in writing, that this code would create an undue hardship or practical difficulty not generally applicable to other persons in similar circumstances. The decision to grant or deny an exemption shall be in writing, and the city manager's or designee's decision shall be final.

Section 2. The City Recorder, at the request of, or with the consent of the City Attorney, is authorized to administratively correct any reference errors contained herein, or in other provisions of the Eugene Code, 1971, to the provisions added, amended or repealed herein.

Section 3. Enforcement of this Ordinance shall begin six months after its adoption to allow time for community education and to allow businesses sufficient time to implement the program.

Passed by the City Council this

22nd day of October, 2012

Beth Forrest

City Recorder

Approved by the Mayor this

29 day of October, 2012

Kitty Piercy

Mayor

Ordinance - Page 3 of 3

ARTICLE 7. - CARRYOUT BAGS.

§ 15-6-121 - DEFINITIONS.

In this article:

(1) BUSINESS ESTABLISHMENT means any commercial enterprise that provides carryout bags to its customers, including sole proprietorships, joint ventures, partnerships, corporations, or any other legal entity whether for profit or not for profit and includes all employees of the business and any independent contractors associated with the business.

(2) CARRYOUT BAG means a bag provided by a business establishment to a customer typically at the point of sale for the purpose of transporting purchases.

(3) REUSABLE CARRYOUT BAG means a carryout bag that is specifically designed and manufactured for multiple reuse, and meets the following criteria:

(a) displays in a highly visible manner on the bag exterior, language describing the bag's ability to be reused and recycled, as prescribed by rule;

(b) except as provided in subsection (d) below, has a handle;

(c) is constructed out of either:

(i) Cloth, other washable fabric, or other durable materials whether woven or non-woven,

(ii) Recyclable plastic, with a minimum thickness of 4.0 mil and containing only the types of plastic resin as prescribed by rule, or

(iii) Recyclable paper; and

(d) handles are not required for carryout bags constructed out of recyclable paper with a height of less than 14 inches and width of less than 8 inches.

(4) SINGLE-USE CARRYOUT BAG means a carryout bag that is not a reusable carryout bag.

Source: Ord. 20120301-078.

§ 15-6-122 - REGULATIONS.

(A) Beginning on the effective date of this ordinance, the City will engage in a public education campaign to inform business establishments and citizens of the requirements regarding carryout bags.

(B) Beginning March 1, 2013, no person may provide single-use carryout bags at any City facility, City-sponsored event, or any event held on City property.

(C) Beginning March 1, 2013, a business establishment within the City limits may not provide single-use carryout bags to its customers or to any person.

(D) Beginning March 1, 2013, a business establishment within the City limits must provide prominently displayed signage advising customers of the benefit of reducing, reusing and recycling and of the need to use reusable carryout bags. The language and placement of signs under this Section shall be as prescribed by rule.

(E) A business establishment within the City limits may provide or sell reusable carryout bags to its customers or any person. A person may provide or sell reusable carryout bags at any City facility, City-sponsored event, or any event held on City property.

Source: Ord. 20120301-078.

§ 15-6-123 - EXEMPTIONS.

This article does not apply to:

(1) Laundry dry cleaning bags, door-hanger bags, newspaper bags, or packages of multiple bags intended for use as garbage, pet waste, or yard waste;

(2) Bags provided by pharmacists or veterinarians to contain prescription drugs or other medical necessities, only if the bags are recyclable within the City of Austin residential recycling program;

(3) Bags used by restaurants to take away prepared food, only if the bags are recyclable within the City of Austin residential recycling program; and

(4) Bags used by a consumer inside a business establishment to:

(a) Contain bulk items, such as produce, nuts, grains, candy, or small hardware items,

(b) Contain or wrap frozen foods, meat, or fish, whether or not prepackaged,

(c) Contain or wrap flowers, potted plants or other items to prevent moisture damage to other purchases, or

(d) Contain unwrapped prepared foods or bakery goods; and

(5) Bags used by a non-profit corporation or other hunger relief charity to distribute food, grocery products, clothing, or other household items.

Source: Ord. 20120301-078.

§ 15-6-124 - HARDSHIP VARIANCE.

(A) The Director may grant a variance from a requirement of this article only after determining that:

(1) application of this article would cause undue hardship based upon unique circumstances, or

(2) application of this article would deprive a person or business enterprise of a legally protected right.

(B) The request for variance shall be submitted on a form prescribed by rule.

(C) A variance granted under this Section must be the minimum departure necessary to address the hardship.

(D) The Director shall prepare written findings to support the grant or denial of a variance request under this Section.

Source: Ord. 20120301-078.

Bill No. 8-11
Concerning: Taxation – Excise Tax – Disposable Carryout Bags
Revised: 4-12-11 Draft No. 2
Introduced: March 15, 2011
Enacted: May 3, 2011
Executive: May 11, 2011
Effective: January 1, 2012
Sunset Date: None
Ch. 6, Laws of Mont. Co. 2011

COUNTY COUNCIL
FOR MONTGOMERY COUNTY, MARYLAND

By: Council President at the Request of the County Executive

AN ACT to:

(1) impose an excise tax on certain carryout bags provided to customers at certain retail establishments;
(2) require certain retail establishments to collect the carryout bag tax and remit the tax to the County;
(3) authorize payment of a fee to compensate certain retail establishments for collecting the carryout bag tax;
(4) allow certain exemptions from the carryout bag tax;
(5) authorize imposition of an estimated carryout bag tax under certain circumstances;
(6) authorize the imposition of interest and penalties for failure to collect and remit the carryout bag tax; and
(7) generally amend County law to provide for an excise tax on certain bags.

By adding
Montgomery County Code
Chapter 52, Taxation
Article XIV, Carryout Bag Tax
[[Sections 52-101, 52-102, 52-103, 52-104, 52-105, 52-106, and 52-107]]

Boldface	*Heading or defined term.*
Underlining	*Added to existing law by original bill.*
[Single boldface brackets]	*Deleted from existing law by original bill.*
Double underlining	*Added by amendment.*
[[Double boldface brackets]]	*Deleted from existing law or the bill by amendment.*
* * *	*Existing law unaffected by bill.*

The County Council for Montgomery County, Maryland approves the following Act:

1 **Sec. 1. Article XIV is added to Chapter 52 as follows:**

2 **Article XIV. Carryout Bag Tax.**

3 **52-101. Definitions.**

4 In this Article, the following terms have the following meanings:

5 **Director** means the **Director** of the Department of Finance.

6 **Carryout bag** means a paper or plastic bag provided by a **retail**
7 **establishment** to a customer at the point of sale, pickup, or delivery to carry
8 purchased items. **Carryout bag** does not include:

9 (1) a bag provided by a pharmacist that contains a prescription drug;

10 (2) any newspaper bag or bag intended for initial use as a dry cleaning,
11 garbage, pet waste, or yard waste bag;

12 (3) a bag provided at the point of sale **[[**at a stand**]]** at a seasonal event, such
13 as a farmers market, street fair, or yard sale, or by an **occasional**
14 **retailer**; **[[**or**]]**

15 (4) a paper bag that a **restaurant** gives a customer to take prepared or
16 leftover food or drink from the **restaurant**; or

17 (5) a bag used to package a bulk item or to contain or wrap a perishable
18 item.

19 **Occasional retailer** means a **retail establishment** that engages in the retail
20 sale of goods no more than 6 days in any calendar year.

21 **Restaurant** means any lunchroom, cafe, or other establishment located in a
22 permanent building for the accommodation of the public, equipped with a
23 kitchen containing facilities and utensils for preparing and serving meals to the
24 public, and outfitted with a public dining area. A **restaurant** does not include
25 any area of a supermarket, department store, or other **retail establishment**
26 beyond the kitchen and public dining area.

27 **Retail establishment** means any person engaged in the retail sale of goods.
28 **Retail establishment** includes any supermarket, convenience store, shop,
29 service station, or restaurant, and any other sales outlet where a customer can
30 buy goods.

31 **52-102. Tax imposed.**

32 (a) A tax in the amount of 5 cents is levied and imposed on each customer
33 for each **carryout bag** that a **retail establishment** provides to the
34 customer.

35 (b) Each **retail establishment** that provides a **carryout bag** to a customer
36 must collect the amount of the tax imposed under subsection (a) when
37 the customer makes any payment for goods in person, through the
38 Internet, by telephone, by facsimile, or by any other means. The **retail**
39 **establishment** must hold the taxes required to be collected under this
40 Section in trust for the County until remitted as required under Section
41 52-103.

42 (c) Each **retail establishment** may retain 1 cent from each 5-cent tax that
43 the **retail establishment** collects to cover the administrative expense of
44 collecting and remitting the tax to the County.

45 (d) A **retail establishment** must indicate on the customer's transaction
46 receipt the number of **carryout bags** that the **retail establishment**
47 provided to the customer and the total amount of tax levied under this
48 Section.

49 **52-103. Remittance.**

50 (a) Except as provided in subsection (b), on or before the 25th day of each
51 month, each **retail establishment** must remit the full amount of the tax
52 collected for all **carryout bags** provided to a customer during the
53 previous month, less the amount retained under Section 52-102 (c).

54 (b) A **retail establishment** is only required to remit the taxes to the
55 **Director** when the cumulative taxes collected under Section 52-102(a)
56 since the previous remittance, if any, exceed $100.
57 (c) Each remittance must be accompanied by a report of all transactions
58 that involve bags subject to the tax. The report must be on a form
59 supplied by the **Director** and must contain the number of bags supplied
60 or provided to customers, the amount of tax required by this Section to
61 be collected and any other information the **Director** requires to assure
62 that the proper tax has been remitted to the County.
63 (d) (1) If the **retail establishment** does not file a required report by the
64 deadline established under subsection (a), the **Director** may
65 estimate the amount of tax due. The **Director** may base the
66 estimate on a reasonable projection of bags supplied or provided
67 and may consider taxes reported by other **retail establishments**.
68 (2) The **Director** may send a notice of the estimated tax due,
69 including interest and penalty, to the **retail establishment**'s last
70 known address. The **retail establishment** must pay the estimated
71 tax, including any interest and penalty assessed by the **Director**,
72 within 10 days after the notice is sent.
73 (e) Each **retail establishment** must preserve for 3 years all records
74 necessary to determine the amount of the tax due under this Section.
75 (f) The **Director** may inspect any records required to be kept under this
76 Section at any reasonable time.
77 (g) The **Director** must deposit all taxes remitted under this Section into the
78 stormwater management fund created under Section 19-35, after
79 deducting the cost of administering this Article.
80 **52-104. Interest and penalties.**

81 (a) If a **retail establishment** does not remit to the **Director** any tax owed
82 under Section 52-103 when due, the **retail establishment** is liable for:
83 (1) interest on the collections at the rate of one percent per month for
84 each month or part of a month after the remittance is due; and
85 (2) a penalty of 5 percent of the amount of the collections per month
86 or part of a month after the remittance is due, up to 25 percent of
87 the tax collected.
88 (b) The **Director** must collect any interest and penalty as part of the
89 remittance due.
90 **52-105. Prohibited conduct.**
91 A **retail establishment** must not:
92 (a) neglect or refuse to collect or remit the tax levied under this Article;
93 (b) file an incomplete, false, or fraudulent report to the **Director**;
94 (c) neglect or refuse to keep complete and accurate records; or
95 (d) refuse to allow the **Director** to inspect and audit the **retail**
96 **establishment**'s records.
97 **52-106. Regulations.**
98 The County Executive may adopt regulations under method (2) to implement
99 this Article.
100 **52-107. Enforcement.**
101 Any violation of this Article is a Class A civil violation. Each violation is a
102 separate offense. A conviction under this Section does not relieve a **retail**
103 **establishment** from paying a remittance owed to the County.
104 **Sec. 2. Effective Date.**
105 This Act takes effect on January 1, 2012.

106 *Approved:*

107

108 Valerie Ervin — 5/5/2011

Valerie Ervin, President, County Council — Date

109 *Approved:*

110

111 Isiah Leggett — May 11, 2011

Isiah Leggett, County Executive — Date

112 *This is a correct copy of Council action.*

113

114 Linda M. Lauer — May 16, 2011

Linda M. Lauer, Clerk of the Council — Date

Staff Summary of Public Comments from Public Forum on Proposed Single-Use Plastic Bag Ordinance. February 3, 2015

In all, 26 residents spoke at the Public Forum on the Proposed Single-Use Plastic Bag Ordinance. One speaker simply had a neutral question: "If we buy clothes at a store like Target that also sells food, can we use plastic bags?"

Ten speakers spoke against the potential ordinance. General themes for reasons the ordinance should not be passed centered on freedom of choice, suggestions to focus instead on education and littering, the idea that paper bags are more harmful to the environment than plastic bags, the need for a more precise baseline of current bag use and litter, measurable reduction goals and evidence of results from other communities.

Fifteen speakers spoke in support of the ordinance. However, please note that one of the speakers was Dr. Vom Saal, who was also a presenter. General themes for reasons the ordinance should be passed were the high rates of plastic bag litter in Missouri water ways, high rates of plastic bag litter in the City of Columbia, health impacts related to plastic bag waste in the environment, the thought that many other communities are enacting similar policies, and the idea of good stewardship for the Columbia community. Below is a shorthand summary of all the comments made for or against the ordinance.

Pro	Con
1. Have we heard back from grocery stores? I've gotten some resistance from baggers when I use my cloth bags, but I suggest using a fold out bottom for the bags that makes them easier to use. **Same speaker, later in forum**: And what about all the plastic bag litter behind Wal-Mart near Hinkson Creek? We don't see what's happening in the environment because we are not there; there are no trails that go along the Hinkson Creek.	1.Should be statewide regulations, if any at all. Percentage of trash that is plastic bags is so little compared to other trash. Don't like nanny-state rules. Would be better to go after littering.
2. Previously in the solid waste industry. Education only goes so far.	2. Should be an educational program. Refresh the litterbug campaign. Should be about choices and my freedom to choose
3. Stream team coordinator who works from Stadium Blvd. to the confluence of the Hinkson with Flatbranch would argue that plastic bags are the number 1 item his team picks up	
4.Stream cleanup coordinator. Sees a lot of plastic bags in the streams. Education doesn't have the same impact as policy change. This policy helps promote good change and stewardship of our community	3.If plastic bottles are everywhere, why don't we attack the plastic bottles instead of the plastic bags? If plastic is so dangerous, why hasn't the federal government shut down all the plastic companies? I recycle and encourage my tenants to. I am against the ban
5. I am a cancer survivor. I do not want to eat or drink out of anything in plastic and do not want plastic in my water. Toxics in plastic do come up the food chain and we're all exposed to them.	

6. A number of years ago I ran a study on trash on road sides. We looked at the number of items, the weight of items, and the surface area of items. Cigarette butts were number one item in amount picked up. Plastic bags were the number 1 item in surface area. Car parts (tires and pieces of metal), were the number one item in weight. When industry talks about this, they say that plastic bags weigh hardly anything, they only make up a couple of tenths of % of the waste going to the landfill, but that is because they are very light weight. But in streams and water ways they make up the most surface area.	4. New education campaigns are needed. This is not the way we should be going about encouraging this--it should be phased in
7. I think this is extremely important issue. There is a need for additional education on where our waste goes.	
8.Worked for gathering petition signatures to ban plastic bags. A lot of other U.S. communities and countries have banned plastic bags - they are also experiencing a lot of stormwater issues.	
9.Plastic bags are a concern because they're so mobile - because their lightweight. Visited the Hinkson Creek near the landfill for research and saw plastic bags everywhere, lined up against the fence, in the trees, in the stream. It's obvious that the landfill is failing to capture them.	
10. Have changed my own behaviors pretty easily once I made the choice to stop using plastic bags. It will make a big impact on the world as a whole	5. Served on the first city-wide and university recycling committees with a focus to reduce, recycle, and reuse. However, we should not be moving people towards paper bags - which are worse for the environment (they take more resources to produce, are heavier and take more fuel to transport, and if they end up in the landfill they produce methane). The plastic bags are not one-use bags, you've already addressed many of the ways that they are used. Don't legislate people. Let them make good decisions.
11. **Dr. Vom Saal Response:** Regarding the discussion on the environmental impact of paper bags, there is good data from other states that impose even a minor cost, such as 10 cents, that folks won't just go from paper to plastic. Data shows the fee will actually greatly reduce the use of paper bags, as well as plastic.	6. No doubt, plastic is one of the greatest inventions, and we are now seeing the unforeseen consequences of it. However, the presentation did a poor job of connecting the plastic bag to the problems that were shown in the video. I think we're starting in the wrong place. This regulation should be submitted to the voters and let them decide what they want.

12. I think this ordinance is excellent. Should be stronger, but we should take what we can get. The idea that paper bags are worse than plastic bags, which leach neuro-toxins, is incorrect, in my opinion. As far as education and litter enforcement being a better approach, it's not a zero sum game, it's not one or the other. Plastic bottles are a problem, but that's no reason we can't address the problem of plastic bags.	7. I do recycle, however, I am against this ban. Take it to a vote of the people. I came with an open mind and was not convinced by this presentation. Would like to see a case study on the impacts of the communities that have enacted these bans.
13. I am a member of the Stream Team and urge the city council to pass this.	8. The plastic bag ban is not going to effect the places that people litter from: Taco bells, etc. Those people are still going to litter - they're already littering. The ban will punish people like me who don't litter and recycle. Having it not be applied to all stores is confusing for the customers.
14. DC has a similar ban that is focused on pride in the community and cleaning up	9.This is a disadvantage to the poor because the plastic bags make easier to carry more groceries on the busses for example. What is the metric for measuring the GHG emissions reduced by the ban? The political ill-will caused by this is only going to do harm to the community. Measure impacts - educate - THEN consider banning.
15. Rather than have the attitude that we shouldn't focus on this, but we need to focus on this instead; we should have the attitude that we need to start somewhere. There's also a lot of talk about re-using the plastic bags, but where do those bags go after they've been re-used, as opposed to the bags that can be used over and over again? A better mind-set is to focus on the future. I personally can't wait until the day when we can say, "Remember when we used to use those single-use plastic bags? Boy, what were we thinking?	10. We are all concerned about our environment - why was the proposal discriminating on those who are not poor by exempting WIC and SNAP recipients from paying the fee?

Sheela Amin <skamin@gocolumbiamo.com>

Fwd: Additional Information/Materials Associated with B51-15 (Single Use - Plastic Bag Ordinance)

1 message

Sheela Amin <skamin@gocolumbiamo.com> Mon, Mar 2, 2015 at 10:44 AM
To: City Council <City_Council@gocolumbiamo.com>
Cc: "Buffaloe, Barbara" <babuffal@gocolumbiamo.com>, Leah Christian <lcchrist@gocolumbiamo.com>, "Cole, Heather" <hlcole@gocolumbiamo.com>, "Matthes, Michael" <mematthe@gocolumbiamo.com>

Hi all,

I failed to provide you the petition that was dropped off by a representative of the Sierra Club. You have the cover letter as part of the council meeting packet, but do not have the actual signature pages due to it being received late on Thursday, the day the packet went out.

The letter along with the signatures/petition pages are attached.

Thanks,
Sheela

--------- Forwarded message ---------
From: **Sheela Amin** <skamin@gocolumbiamo.com>
Date: Mon, Mar 2, 2015 at 10:18 AM
Subject: Additional Information/Materials Associated with B51-15 (Single Use - Plastic Bag Ordinance)
To: City Council <City_Council@gocolumbiamo.com>
Cc: "Buffaloe, Barbara" <babuffal@gocolumbiamo.com>, Leah Christian <lcchrist@gocolumbiamo.com>, "Cole, Heather" <hlcole@gocolumbiamo.com>, "Matthes, Michael" <mematthe@gocolumbiamo.com>

Hi all,

Ms. Hoppe asked that the Council be provided the following additional information associated with B51-15 as you all will potentially discuss the single use plastic bag ordinance this evening.

Attached is the report from the Environment and Energy Commission, which you have seen previously.

Below is the link to the video of the public information session that had been held on that topic on February 3 in the City Council Chamber:
https://www.youtube.com/user/columbiacitychannel
or
https://www.youtube.com/watch?v=iVTQHDF_bLQ

Thank you,
Sheela

--
Sheela Amin
City Clerk

City Hall
701 E. Broadway
2nd Floor
Columbia, MO 65201

573.874.7207
skamin@gocolumbiamo.com

--

Sheela Amin
City Clerk

City Hall
701 E. Broadway
2nd Floor
Columbia, MO 65201

573.874.7207
skamin@gocolumbiamo.com

2 attachments

EEC Report - REP11-15.pdf
1565K

Petition Provided by Sierra Club re single use plastic bags.pdf
4429K

To: Mayor McDavid and all Columbia City Council Members

From: Osage Group of the Sierra Club

Date: February 26, 2015

Subject: Petitions in support of the proposed ordinance pertaining to the regulation of single-use plastic bags

Please find enclosed the signed petitions from the citizens of Columbia, Missouri as well as from people who live near the city limits and shop in Columbia. 899 signatures from Columbia residents and 120 signatures from individuals outside the city limits are included.

The individuals who have signed the petition are in support of a single-use plastic bag ban in Columbia, Missouri. We encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

In the event that the proposed ordinance is tabled during the upcoming council meeting, we request that the Council direct the city staff to conduct an education campaign to raise awareness of the negative impact of plastic bags on littering, the storm water system, wildlife and the environment. An educational campaign will enable the citizens to participate in an informed discussion of the merits of a single-use plastic bag ordinance in the future.

Any questions regarding this petition may be addressed to Carolyn Amparan, Chair of the Osage Group of the Sierra Club. Carolyn can be contacted at 417-793-8600 or Carolyn.amparan@gmail.com

Carolyn Amparan

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Columbia Missouri Residents Only

Name	Address	Zip Code	Email address (optional)
Sonya Nicholson	308 Manor drive	65203	Sonya Nicholson
Ruth E.W. Walker	3716 Joan Ct.	65201	Ruth E.W. Walker

2

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Columbia Missouri Residents Only

Name	Address	Zip Code	Email address (optional)
Liberty Hunter	704 Clinkscales Rd	65203	
Erin King	1614 Hinkson Ave.	65201	
Caroline Hargrove	1009 Coats St.	65201	
Steve Rosenberg	118 Tracy Dr.	65203	
Lori McCurdy	201 West Sexton Rd	65203	
Alan Saunders	214 Saint Joseph	65201	
PATRICK COVERT	2905 Creekside	65202	
Kathe Freeling	707 Cook Ave	65203	

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Columbia Missouri Residents Only

Name	Address	Zip Code	Email address
Jessica VanGuren	402 Kentucky Blvd Columbia, MO		
Bailey Walter	511 E Rollins Columbia MO	65201	
Grant Paris	211 Hitt St Apt 207 Columbia	65201	ggp944@mail.missouri.edu
Caleb Smith	1811 Gordon St.	65201	
Kelsey Grant	414 W Walnut St Columbia MO	65203	—
Christiane Quinn	719 W Broadway	65203	
Cheryl Price	511 Parkade Blvd.	65202	
Dawn Zetterberg	608 Hunt	65202	
Nathaniel Braun	3601 W. Broadway #27-201	65203	
Allen Tarter	1108 Princeton	65203	
Amanda Jetty	1013 E Walnut Suite 102	65201	
Emily Waggoner	1800 N Waterfront Dr C	65202	
BRIAN CHERVITZ	301 CAMPUS VIEW DR APT 156C	65201	
Evan Chiarelli	222 Elliott dr	65201	

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Columbia Missouri Residents Only

Name	Address	Zip Code	Email address
Mary Denson	1020 Yale	65203	Maryc.denson@gmail.com
AKilah Stanton	1806 Mayberry	65203	akilaRStanton@yahoo.com
Orlando Winbush	2012 W Ash St Apt A4		Alex Orlandowinbush gmail.com
Margaret Grace Vega	3008 Middlebush Dr., Columbia, MO 65203		
Dean Andersen	814 Timbers Ct, Columbia, MO 65201		andersendc@health.missouri.edu
Allie Blaylock	1109 Sunset Dr.	65203	allie.blaylock@gmail.com
Caitlyn Gallip	2015 College	65201	cgallip387@gmail.com
Rachel Cullen	2510 Lenox PL	65203	rcjonasgate@gmail.com
Billie Coachman	3302 Talent Dr. Columbia, Mo.	65203	
Steven Chaffin	905 Hitt Street	Columbia, MO,	65201
Emily Waters	515 S 5th street	Columbia MO	65201
Miguel Rodriguez	309 Waugh St Apt C-1	Columb. MO	65201
[illegible]	1913 Princeton Dr.	Columbia Mo	65203
Ann Rogers	112 E Burnam Rd.	Columbia, MO	65203

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Columbia Missouri Residents Only

Name	Address	Zip Code	Email address
Tony Bell	1617 Windsor St	65201	—
out of town Isabella Padicelli	108 La Vista	65583	122ypaoli7@gmail.com
Sarah Trott	1005 Tiger Ave.	65201	—
Marisa Arz	511 E. Rollins St	65201	—
Erin Pearson	208 St Joseph St	65201	—
Laurel Goodman	408 N. 9th St		—
Toli Olofintuyi	821 i Southwest Blvd	65201	—
Lance Mallette	1107a Rangeline St.	65201	LLYaf@mail.missouri.edu
Margaret Miller	102 E Green Meadows Rd #6	65203	
Martha Stevens	903 Ash Street	Columbia, MO	
[illegible]	3150 N. Rt. Z	65202	
[illegible]	804 Wilson	65201	
[illegible]	1708 Pennington, St. Mo		

(1 out) (12 CC)

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Columbia Missouri Residents Only

Name	Address	Zip Code	Email address
Jonathan Sessions	115 Aldrich Ave	65203	jsesh@jsesh.com
Debra Howerstine	5065 N. Clearview	65202	howerstined@missouri.edu
Edward Duqum	502 Campus Lot	65201	ed@gmail.com
Danielle Lely	1221 E. Walnut	65201	dlfly6@mail.missouri.edu
Anne Wright	3814 Aspen Heights	65201	amawyb@mail.missouri.edu
Bill Allen	1011 Elston Dr., Columbia	65203	allenwg@missouri.edu
Nina Furstenau	201 Gentry, Apt MO	65211	furstenaun@missouri.edu
Brittany Marchus	1222 E. Walnut	65201	bmrmf@mail.missouri.edu
Paula Moscoso	501 Westminster Av.	65251	paumoscoso91@gmail.com
Karen Davis	2645 Silverwing Tr.	65201	karendavis13@yahoo.com
Lindsey Miller	503 Kentucky Blvd, Columbia MO	65201	LRM62D@mail.missouri.edu
Marilyn K. Cameron	2 Eubanks Ct. Columbia, MO	65203	
Glenn Cameron	2 Eubanks Ct Columbia, MO	65203	

Petition to the Columbia City Council

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Please write legibly. Columbia Missouri Residents Only

Name	Address	Zip Code	Email address
out of town Brad Allgood	1725 Irving St NW	20010	bradallgoodnew@yahoo.com
Joann Flom	1133 Ashland 610	65201	suddenly.appear@gmail.com
SCOTT WILSON	101 S. FIFTH, #6	COMO 65201	SWILSON@TRANQUILITY.NET
Caitlin Campbell	2503 Gabrianna Ct	65203	cmjd7d@mail.missouri.edu
Stuart P. Derry	2502 LONGVIEW DR	65203	
Lu Edgerly	2502 Longview Dr	65203	
Emma Jean McKinin	703 Englewood Dr	65201	
DONALD R JOHNSON	5034 N'ONIAL Rd	COLUMBIA MO	
Angie WOOD	503 N. Brookline Dr.	65203	angelinewd@gmail.com
Phyllis Roller	3905 THORNWOOD Dr	65202	prollersphr@yahoo.com
Betty Wilson	1719 University	65201	
Manuela Wistal	2205 N Country Club Dr.	65201	
Scott Curtis	2205 N. Country Club Dr	65201	
Sally Kessell	715 Lyons St.	65201	

(1 out) (13 col)

Petition to the Columbia City Council

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Please write legibly. Columbia Missouri Residents Only

Name	Address	Zip Code	Email address
Nadey Waqe	1849 Timber creek dr.	65202	UWasena@gmail.com
Jacob McKinley	1402 Bass Ave 2	65201	
Emily Haghighi	403A N. 8th St.	65201	medart20@gmail.com
Laura Fenke	903 Hitt	65201	csfdmb@mail.missouri.edu
Jerry Schorna	4913 Royal Lyft	65203	jerryschorna@hotmail.
Jennifer Anderson	3605 Calvinte Ct.	65201	
Joyce Randle	5000 Hatteras Dr Col.	65202	
Nancy Malugani	1034 Westwinter	65203	Izedg Malugani
Laura Sattlowski	1408 Pratt ~~Ave~~ St	65201	
JARED WHEAR	700 McBAINE AVE	65203	jaredcharlesw@gmail
Olivia Liu	Schurz Hall	65202	olk55@mail.missouri.edu
Elizabeth Long	1617 Windsor	65201	elizabethflong@gmail.com
Erin Smith	1842 Cliff Dr.	65201	
Allyson Junker	215A ~~Cotton~~ St. Joseph Street	65201	
Isaac Pasley	1015 College Park Dr.	65203	ilpd45@mail.missouri.edu

Petition to the Columbia City Council

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Please write legibly. Columbia Missouri Residents Only

Name	Address	Zip Code	Email address
Ban Plastic (Category)	SUPPORT effort STO create Biodegradable Alternatives		honubabu@gmail.com
Caleb [illegible]	8810 TOMLIN HILL RD	65201	
Melissa Camper	2204 Iris Dr.	65202	melissacam@aol.com
Michelle Long	1700 Forum Blvd. #3201	65203	mlong426@hotmail.com
Nicholas Jackson	104 E Stewart Rd	65203	npjgvmo@gmail.com
Lindsay Smith	3300 Greenridge Rd.	65202	Lnzsmith@gmail.com
Alex Mitchell	1021 Ashland Rd	75201	amnd5d@mail.missouri.edu
Judy Schermer	4913 Royal Gytham Dr.	65203	judyschermer@hotmail.com
Mary Bifulco	403 N. College Ave	65201	MBIFULCO10@YAHOO.COM
Day Polacco	1034 Westwinds Ct	65203	polaccoj@missouri.edu
Robert Hemmelgarn	805 Alton Ave.	65201	ROBERTDAVIDH@GMAIL.COM
Eric Meyer	3820 Village Park Dr	65201	eric--meyer@hotmail.com
Y Moore	118 tracy dr.	6203	ycmoore9@mail.
Courtney Smith	1811 Gordon St.	65201	courtjoc@aol.com

256

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and mandate a small charge of at least 10 cents for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Columbia, Missouri residents only.

Name	Address	Email (optional)
Jeff Frey	300 Maplewood Dr.	flagdoulus@me.com

Petition to the Columbia City Council

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Please write legibly. Columbia, Missouri residents only.

Name	Address	Email (optional)
Ashley N. White	1619 Hershey Ct 65202	anwvy4@mail.missouri.edu
Peter Holmes		pfholmes63@gmail.com
Scott Powell	5700 E. OSAGE RIDGE COLUMBIA, MO	
Jack Thomas	2616 HILLSKIRT DR.	JAThomas1996@gmail.com

Petition to the Columbia City Council

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Please write legibly. Columbia, Missouri residents only.

Name	Address	Email (optional)
Brianna Westervelt	2501 Northampton Dr.	
Jackson Harrison	408 N College Ave, Apt 101 Columbia, MO 65201	jkhsnc@mail.missouri.edu
Garrett McKee		
Lori McCurdy	201 W Sexton Rd 65203	lori@columbiaurbanag.org
Debra Blakely	3906 Cameron Ct, 65203	
Paul Blakely	3906 Cameron Ct 65203	
Marlys Jacobs	2607 Eastwood Dr, 65202	
Ken Midkiff	1005 Bellevue Ct	12midkiff@centurylink
Alex Prentice	11 East Phyllis Ave, 65202	
Elizabeth Larmore	5812 Winingstar Ct. Columbia 65203	
BROCK FRIEDRICH	204 PERSHING RD 65201	
Sarah Horton	1133 Ashland Rd Apt 210 Columbia	
Christopher Oldfather	1133 Ashland Rd Apt 210 Columbia	

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and mandate a small charge of at least 10 cents for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Columbia, Missouri residents only.

Name	Address	Email (optional)
Jeff Barnow	1007 COATS ST 65201	jeffrey.barnow@gmail.com
Paul W Rice	1214 London Columbia	
John T. Murray	911 EDGEWOOD 65203-2823	john@johntmurraylaw.com
Falke B. Reed	911 Edgewood Columbia MO 65203-2823	amstown@johntmurraylaw.com
Denise Ferguson	2290 Benfield Subdivision 65201	
Daniel Cullimore	715 Lyons Street 65201	
	1904 KATY LANE	legacyshelterse@mchsi.com
ML Lehmann	2601 S Providence 65203	
James Lind	1619 Hershey Ct	
ERIC PHERIGO	103 PINEWOOD DR.	
Emily Pherigo	103 PINEWOOD Dr.	emily-emily-k@yahoo.com
HUNTER MARET	821 E. WALNUT ST.	
Hayley Baker	2501 Northampton Dr.	

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 Cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Columbia Missouri Residents Only.

Name (Print + ~~Write~~ Sign)	Address	Zip Code
Carolyn Amparan	4804 Shale Oaks Ave	65203
Gabe Amparan	4804 Shale Oaks Ave	65203
David Finke	1106 Maplewood Dr.	65203
Ingrid Clark	208 E Burnwood Ln	65203
Jessica Stone	308 S. 10th St. Apt 413	65201
Laura McCann	2205 Kyle Drive	65203
RYAN LANMAN	2609 ~~2069~~ EAST WOOD DR	65203
~~Jennifer Stoker~~	~~104 McNielle~~	~~65256~~
Ashley Holt	1414 Grindstone	65201
Rick Meredith	116 W Forest Ave	65203
PHIL WRIGHT	2306 BROOKVIEW	65203
Kim Gill	1601 E. Nancy Ct.	65201

Petition to the Columbia City Council

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Please write legibly. Columbia, Missouri residents only.

Name	Address	Email (optional)
Pat Fowler	606 N Sixth St	fowlerpatj@gmail.com
Vincent Gallo	411 N. 9th St. Co-Mo.	
Christina Easley	2709 W. Rollins Rd. Apt 2. 65203	
Megan Steen	31003 Blue Cedar Ln Como 65203	
Chip Cabra	1625 Wilson Ave.	
Mara Aruguete	1625 Wilson Ave	
Aileen & Larry Garvin	500 N. Brookline Dr. 65203	
Emma O'Connell	2557 S. Roby Fm 65279	
Scott O'Connell	" " " "	
Lawren Askinosie	14 Aldiah Ave.	—
Kyle Cook	404 N Tenth Como 65201	
Scott Dye	2222 Bluff Blvd 65201	

out of town

out of town

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Petition to the Columbia City Council

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Please write legibly. Columbia, Missouri residents only.

Name	Address	Email (optional)
Michael P Clark	304 Ridgeway Ave, Columbia, MO	Pontiacporpoise@gmail.com
Shawn Palmer	307 Melbourne St Columbia MO 65201	
Alicia Pigg	1600 Kathy Dr, Columbia, MO 65202	
Mitchell Guthrie	105 So Greenwood Ave Col MO 65203	
Jeanie Scott-Pillen	3804 Godfrey Dr., Columbia, MO 65203	
Ambrosia McCord		
Gabriella McCord		gabriellamccord@hotmail.com
Paul Bosg	813 Colyott, Co Mo 65203	
Christine Mezines	3800 Faurot Dr Como 65203	
KENNETH HAMMANN	1808 TRELLIS LN COLUMBIA MO 65201	
CAROL HAMMANN	1808 TRELLIS LN COLUMBIA MO 65201	
Andrew Mueller	2601 Ridgefield Road	Columbia MO 65203
Melinda Farhangi	2602 Luan Ct	Columbia, MO 6530

Nov 23 Holiday Parade

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 Cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Name	Address	Zip Code
Patricia J. Fowler	606 N Sixth St.	65201
Dana Tanly	905 S. Glenwood Ave	65203
Pam Forbes	707 Donnelly Ave	65203
Pamforbes98@yahoo.com		
John Bat	johnbat@gmail.com	65201
Dee Dokken	804 Again	65203
Fred Emmscal	4679 S. Bearfield	65201
Marta Velez	2905 Franklin	65203
Ann Midkiff	1005 Bellview St, Apt.	65203
Glenda Kelly	500 Overland Ct	65203
Verna Kelly	500 Overland CT	65203
[illegible]	5200 Forest Glen Dr.	65203

add to comm.

Sasha Mizzen

11

DIER

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 Cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Name	Address	Zip Code
[illegible]	1607 Bluffdale Dr.	65201
Mike Sears	" "	"
Kathy Love	1623 University	65201

3

Petition to the Columbia City Council

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Please write legibly. Columbia Missouri Residents Only.

Name	Address	Zip Code
Lea Langdon	1301 Overhill Ct.	65203
Emily Fisher	5981 S. Hwy KK	65203
Nancy Rezabek	3805 Conlect ct	65203
Clint Shannon	Schurz on College Av,	65201
Rebecca De Courley	813 Colgate	65203
Rose Neal	3605 Berrywood	65201
Cody Stenckhoff	3605 Berrywood Dr.	65201
DEBBIE KNESS	7661 BELL ROAD COL MO	65202
~~[illegible]~~	~~[illegible]~~	
P. B. MacPherson	1100 Sunset LN Col MO	65203
Lori McCurdy	201 West Sexton Rd	65203
David Sautner	3200 Northland Dr.	65202

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Please write legibly. Columbia Missouri Residents Only.

Name	Address	Zip Code
Victoria Boyd-Kennedy	504 Manor Dr. Col.	65203
SCOTT WIRMAIER	915 W. ROLLINS, COL	65203
Vicky Woodward	1703 Garrison PL	65203
Mary Sol	2775 W Shagbark Co	"
Kathryn Kendall	7 E Lathrop Rd	65203
[illegible]	1901 Chapel Ridge Rd	65203
Norma Feik	6 E Stewart Rd	65203
Jane Crawford	15 Austin Av	65203
[illegible]	1115 Keegan Ct	65203
Angela Bourne	1515 Liberty St.	65201
Susan B Jones	2409 Lynnwood Dr.	65203
John R Jones	2409 Lynnwood Dr	65203

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 Cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Columbia Missouri Res[illegible] Only.

Name	Address	Zip Code
Paula Harmon	403 Alexander	65203
Raquel Ortiz	4606 Lake Valley	65203
Debbie Cahow	405 Manor Dr	65203
Michael Bushmann	1902 Katy Wood Ct	65203
Marcus Wilkins	701 West Blvd.	65203
Katie Burnham Wilkins	"	"
Julie Nardy	2711 N. Willowbrook Rd.	65202
Kalle Wm	1703 Madison Park Ct	65203
Diane Gingles	917 Stratford Ct	65203
Jennifer Somerberg	3060 S. Maple Bluff	65203
Patty Clover	915 W. Rollins	65203

11

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 Cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Columbia Missouri Residents Only.

Name	Address	Zip Code
Summer Foote	3304 Oakside Ct.	65203
Mr + Mrs Chris Polo	3401 Woods Edge	65203
Anne Smith	3100 W. Old field Rd.	65203
Kate Kruenewald	802 S. Fairview Rd	65203
Nellie Boyt	609 West Blvd N	65203
Roberta Carter	1506 Gary S	65203
SEAN FOLEY	804 JEWELL AVE	65203
Jaylynn Gentry	407 N. 9th St.	65201
Lindsey Woodcock	1700 Forum Blvd Apt 3303	65203
Andrea Gerdke	5508 Arrowwood Dr	65202
Rachel Pinnow	1700 Forum Blvd	65203

Bag Ban Petition

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Columbia Missouri Residents Only

6

Name	Address	Zip Code	Email address (optional)
~~Becky Maitlack~~	~~3321 MSB Bridge Ln~~	~~Jeff City 65109~~	
CHRIS Kuhlmann	~~3240~~ 111 Bicknell,	Columbia, MO	65203
Rachel Kesterson	" "	"	" "
Steve Ambsoger	201 4th Ave	Columbia, Mo	65203
Emily Phenigo	103 Pinewood Dr.	65203	emily_emily_k@yahoo.com
Louise Slogh	1411 Stone St	Co Mo	65201
Mary Stilwell	207 Alexander Ave	Columbia MO	65203

6 Col.

1 out

⇒ Bag Ban Petition

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Columbia Missouri Residents Only

Name	Address	Zip Code	Email address (optional)
[redacted]			
Cody Struckholt	308 Loch Ln.	65203	hoffstruck2014@yahoo.com
Alma Hopkins	805 Stratford	65203	hopkinsal4@yahoo.com
[redacted]			
NINA STAWSKI	2100 E. CEDAR TREE LN	65039	nstawski@stephens.edu
[redacted]			
Anthony Pettit	801 Alton	65201	anthonysyotr@gmail.com
Paula Harmon	403 Alexander	65203	pharmonpharmon@yahoo
~~DAVE BANDY~~	~~1015 CS PLEN Col. MO 65203~~		~~dave.roger.bandy@gmail~~
Ellie Prentice	11 E Phyllis Ave	65202	ewaterburyprentice@gmail.com
Alex Prentice	11 E Phyllis Ave	65202	al-prentice@yahoo.com

out of city limits

from a 4 yr. old who apparently supports banning the bag!

4 out

16 Col

Sorry — I crossed out out-of-towners before I knew there was an out of town petition!

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 Cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Columbia Missouri Residents Only

Name	Address	Zip Code
Tim Nigx	801 Alton 65201	65201
Anna Wilson	1004 Otto Ct. 65201 →	65201
Stefanie Mcdollum	2008 Bear Creek	65202
Ruslan Grigoriev	2301 Primrose Dr #6A	65202
DORY COLBERT	700 Westwood	65203
Melissa Poelling	4009 Defoe Dr	65203
Joy Cole	4408 JORDAN CT	65203
Jimmie Markey	1406 A Richardson ST	65201
Sunyata Markey	1406 A Richardson St	65201
Karen [illegible]	PO Box 30762 C.I. MO	65205
Robert E. [illegible]	403 N. Ann. St. #9	65203
Samantha Anderson	1301 Old 63S	65201

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Columbia Missouri Residents Only

Name	Address	Zip Code	Email address (optional)
Nathan O'Neil	522 Pear Tree Cir	65203	
Jessie Starbuck	809 N. 8th St	65201	
Katie Connette	1700 Forum Blvd 2403	65203	
Grant Connette	1700 Forum Blvd, #2403	65203	
Irene Serano	609 Hardin St.	65203	
ROGER GILES	4028 GRACE ELLEN	65202	
Kevin Renfro	117 Aldech Ave	65203	
ERIC PHORIGO	103 Pinewood DR.	65203	
Robert Dolt Potter	1508 Scarborough DR	6520.	
Amy Meier	906 Sandifer Ave Como	65201	
CULLY MEIER	906 SANDIFER AVE COMO	65201	
George Sarafianos	716 Lyon	65203	
William Palmieri	705 Ponnelly Ave	65203	

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and mandate a small charge of at least 10 cents for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Columbia, Missouri residents only.

Name	Address	Email (optional)
John Nichols	511 Hickman Ave. 65201	MrJohnRNichols@gmail.com
Sandy Gummersheimer	1005 Glenwood Ave	gummersheimers@missouri.edu
Jackson Hamlarik	408 N college Ave	jkh3rc@mail.missouri.edu
HUNTER WARET	821 E WALNUT St	" " @ME.COM
Andrew Beckerle	118 Tracy Dr	aybwdf@mail.missouri.edu
Dean Andersen	814 Timbers Ct. Co Mo	andersendc@health.missouri.edu
Hannah Peterson	1133 Ashland Rd Apt 901	hannahd35@yahoo.com
Tyana Washington	1517 Windsor St	
Esther Stroh	807 Leawood Terrace	
William C. Merrell	708 Fairview Ave	

10

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 Cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly Columbia Missouri Residents Only

Name	Address	Zip Code
Matt Hall	717 Bluff Dale Dr.	65201
Joe Kellenberger	260 Broadview Co MO	65201
Emily Tracy-Smith	407 W. Stewart Rd. Co, MO	65203
Matt Smith	407 W Stewart Rd. Co, MO	65203
Anthony Tinque	2401 W Broadway Apt 411	65202
Keith Thornburg	911 Broadhead St.	65203
Liza Babington	900 S. El Chaparral Ave	65201
Hannah Leap	3636 Aspen Nightc parkway	
Jeremy Smith	201 W Ash Columbia MO	65203

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 Cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Columbia Missouri Residents Only.

Name	Address	Zip Code
Allie Mislan	Shingen Mo	65284
Emily Harryman	1804 Sunrise Drive Apt A	65201
Melissa Williams	607 Westwood	65203
[illegible]	2700 Cimarron	65203
Margaret Waddell	6 Bearfield Subd.	65201
Clint Twellman	1804 Sunrise Dr Apt A	65201

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Columbia Missouri Residents Only

Name	Address	Zip Code	Email address (optional)
Elke Boyd	2004 N. Parkhurst Ct	65202	
Corey Dunn	1600 Cardinal Circle	65201	cg.d7n7@m.2300.edu
Brian Page	17 Aldeah Ave	65203	
Patty Turner	4412 Sullivan St	65202	patty@pisinspections.com
Ben Green	206 Anderson Dr.	65203	acombookstore@gmail
Andrew Linhares	1216 Isabella Dr	65203	andrew@renewma.org
Henry Collinger	511 Westwood	65203	
H. Knute Jacobson	711 E. Rockcreek	65203	kjacobson@columbiaonninth.org
G Arthur Pollard	1619 Windsor	65201	

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Columbia Missouri Residents Only

Name	Address	Zip Code	Email address (optional)
Jessica Hill	120 W. Ash St. Apt 4	65203	jahktb@mail.missouri.edu
Ellis Cole	1808 Sunrise Apt A	65201	
Susan Fourcade	627 Bluff Dale Drive	65201	573)449-2955
Kailey Hinehman	415 A N 8th Street	Columbia MO	65201

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Columbia Missouri Residents Only

Name	Address	Zip Code	Email address (optional)
Allie Brett	1301 S Cedar Grove	65201	Ø
~~Sarah Hooper~~			
Jeff Fischer	207 N. William	65201	
Dave Angle	2245 Bluff Blvd.	65201	
Jackie Caskey	1501 Stone St.	65201	
Kadie Crivello	Rte N	Columbia MO	65203
William Beever	Rte N	Columbia Mo	65203
Barbara Bassett	1709 Hinkson, Apt. 7	Columbia Mo	65201
Kim Carpenter	767 N. Ann St.	Columbia MO	65201
Dan Neiswanger	304 West Blvd. N	Columbia, Mo	65203
Rachel Brekhus	703 Hilltop Dr.	Columbia, MO	65201
Wayne Brekhus	703 Hilltop Dr.	Columbia, MO	65201
CORRI FLAKER	2211 Hominy BRANCH	Columbia, MO	65201

N

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Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Columbia Missouri Residents Only

Name	Address	Zip Code	Email address (optional)
Ruthie Moccia	608 S. Greenwood, Columbia, Mo.	65203	
[illegible]	410 Westridge, Columbia, Mo		
Alicia Burke	3602 Timber Run Dr	Columbia, MO 65203	aliciancwilliams@gmail.com
Sheila Greenberg	2501 Limerick	Columbia, Mo 65203	
Ashley Tulay	2006 Parks de Ville Pl	Columbia, MO 65203	tulayashley@yahoo.com

5

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Columbia Missouri Residents Only

Name	Address	Zip Code	Email address (optional)
Patrick Neelin	312 Anderson Ave	65203	patrick.neelin@gmail.com
Rachel Taylor	119 Clinton Drive	65203	rachel.haverstick@gmail.com
Melissa Sharp	4504 Oak View Dr.	65202	melicorstein@yahoo.com
Jason Hanson	207 Maplewood	65203	jason.hanson@pepsico.com
Tim Sparling	1112 Sunset Ln	65203	sparling.tim@gmail.com
Jonathan Asher	313 N. William St	65201	jonathan.c.asher@gmail.com
Rachel Penn	313 N. William St.	65201	rachel.penn@gmail.com
Sam Napier	400 Sondra Ave,	65202	sanapier3@gmail.com
Barbara Ramsbottom	717 Mikel,	65203	
Celena Schmolzi	805 Ridgeway Ave,	65203	
Meredith Truelove	4802 Marble Cedars Dr.	65203	
Sarah Goodnow	912 W. Walnut Ct.		
KAREN DWYER	2255 Bluff Blvd	65201	

26

over

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Columbia Missouri Residents Only

Name	Address	Zip Code	Email address (optional)
Catherine Thorn	213 Willow Way Apt A	65201	
Christina Mattson	807 Mt. Vernon Ave	65201	
Jenna Nelsen-McMiche	1606 Hinkson Ave	65201	jenv35@mail.missouri.edu
Elijah Fortel	1606 Hinkson Ave	65201	elitinkfort@gmail.com
Sara Wilks	1319 St. Christopher	65202	
LINDA FERRIS	2315 Bridgewater	65202	
Catherine L. Murphy	615 paris ct Apt. B	65201	
Caroline Kreps	2411 West Broadway	65203	carolinemkreps@gmail.com
Jim RONALD	1000 PROSPECT ST	65203	
Meg N Vogt	1507 Windsor Street	65201	megan7225@gmail.com
Trent Rash	104 Pershing Road	65203	
Leighton Newmann	419 W 8th St	65041	
Deborah Beste	3813 Dublin Ave	65203	

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Columbia Missouri Residents Only

Name	Address	Zip Code	Email address (optional)
Ellen Runge	2214 Ridgemont	65203	emrunge66@yahoo.com
Julie Macy	117 Aldeah	65203	
KAREN POTTER	1825 Scarborough	65201	
Gabrielle Reneau	3636 Aspen Heights	65201	
Tessa Bodine	5000 W. Driskel BD.		
RICH HOLLAND	2203 FAULKNER CT.	65202	
Maria D'Agostino	400 W. Worley	65202	mariadagostino@yahoo.com
Donna Strickland	416 N College #7	65201	
Mary Willard	237 E. Clemanew	65202	
DAN KUEBLER	16471 HAWKINS RD	65010	
Darcy Higgins	105 3rd Ave	65203	darcyohiggins@gmail
Sylvia Donnelly	p.o. box 22 hartsburg, MO	65039	out of town
Betty Acree	108 Miramar Ln Columbia MO	65203	

out of town (1) ~~25~~ Col

over

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Columbia Missouri Residents Only

Name	Address	Zip Code	Email address (optional)
[illegible]		65202	edBaschum@gmail.com
Sara Strand		65202	smayerca@stephens.edu
Nick Harvey		65203	
Katie Martor	28 Catherine	65203	
MACKENZIE MANZER		65201	
Nicole Pyatt		65201	
Kate Pulley		65203	
Samae Claspill	610 Adams Street Columbia, MO	65201	[illegible]
MARCHELO VERA		65201	
Curt Rohlfing	305 N 9th St Columbia, MO	65201	
Marilyn Parker		65201	
[illegible]	500 W. Walnut Como	65203	
Mark [illegible]	609 Hardin	65203	

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 Cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Name	Address	Zip Code
ANIE RUH	10. E. PHYLLIS AVE	65202
André Perez	23525 Hwy MM	65301
Dani Perez	↑	↑
Alex Stewart	1816B Sunrise Dr	65201
Ellie Cole	700 N William St	65201
MIKENZIE HART	1507 Rosemary ln #102	65201
Danielle Tevlin	1507 Rosemary Ln	65201
SHELBY FEISINER	1600 HINKSON	65201
Adam Saunders	214 Saint Joseph	65201
Anne Miller	215 Holly Ridge	65203
Gary Miller	215 Holly Ridge	65203
Allison Kellenberger	260 Broadview	65201

13

over

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 Cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Name Seth Linkerson **Address** 117 W Corley **Zip Code** 65201

Joe Meyer 615 Maplewood Dr 65203

Leslie Meyer 615 Maplewood Dr 65203

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 Cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Columbia Missouri Residents Only.

Name	Address	Zip Code
Shanae S.	1409 risen star	65202
Rod Leighton	1704 McAlester	65201
Erin Ry	1614 Hinkson Ave.	65201
Lara Landrum	2023 Vine	65201
Tom O'Connor	806 Leawood Ter	65203
Anna Ling	608 Spry Valley	65203
Shauna Marquardt	14 E Craig St	65202
Melody Nashan	608 Spring Valley Rd	65203
Kelsey Allen	3509 scottson way	65203
Sam Urkov	3509 Scottson Way	65203
Samuel Ott	1403 Hinkson Ave, Apt. 1	65201
Macthe Stevens	903 Agan St	65203

33

over

NAOPE

Name	Address	Zip
Steve Rosenberg	118 Tracy Dr	65203
Leigh Lockhart	28 S 9th	65201
JOHN EMERY	13 E LESLIE LN	65202
Joshua Stockwell	408 Melbourne St	65201
Tim Althether	2623 B Summit Rd	65203

Name	Address	Zip
Marina Champion-Garth	2101 W Broadway #103-190	65203
Darwin Hindman	223 Frances Dr	65203
Jill Lucht	100 Aldeck Ave	65203
Lori McCurdy	201 West Sexton	65203
PARKER Hall	103 E. Spring Ln	65201
Ben Eades	903 Again St	65203
Adam Saunders	214 St. Joseph	65201
Martha Hagan	1000 Prospect St	65203
Jim Ronald	1000 Prospect St	65203
Kim Hall & Parker	103 E. Spring Cmo	65201
~~Bobbi Davis~~		
Clare Cannon	PO Box 515 Columbia	65205
Billy Polansky	1009 Coats	65201
Liz Klug	408 Hirth	65203

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 Cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Columbia Missouri Residents Only.

Name	Address	Zip Code
Dusty Mitchell	1839 Timbercreek Dr	
Trent Kline	105 3rd Ave	65201
Curtis Pepper	6406 N Hwy VV	65202
Josh Huba	816 N William	65201
Leah Thomas	1300 E Rollins	65201
Stevie Myers	1100 E Rollins	65201
Ken Green	206 Anderson Ave	65203
Tom O'Connor	306 Leawood Ter	65203
Diana Moxon	"	"
Živile Raskauskaite	1016 South College Ave	65201
Dan Cullimore	715 Lyons St	65201
Paul Hemmel	2513 Oakmeadow Dr.	65202

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 Cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Name	Address	Zip Code
Desiree C Long	503 B Columbia Dr Col	65201
Thomas Andrews	1903 E. Walnut St. Apt 204	Columbia MO 65201
Barbara Smith	101 Anderson Av	Co Mo 65203
Greg Leonard	1306 Jean Rae Drive	Columbia Mo 65203
Jan Kon	1624 Kathy Dr.	Columbia Mo 65202
Lois Bennett	805 Ridgeway Ave	Columbia, mo 65203
Britta Simpson	805 Ridgeway Ave	Columbia, MO 65203
Lizzie Bryan	104 Clinkscales Rd #112	Columbia, MO 65203
Eric Pherigo	103 Pinewood	Columbia MO 65203
Emily Pherigo	103 Pinewood Dr	65203
Barbie Reid	104 W Lathrop	65203 65203
Nancy McIsaac	1807 Madison Park Drive	65203

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 Cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Name	Address	Zip Code
Holly Hendrickson	5600 Majestic Cir., Columbia MO	65203
Shae Collier	3900 citation on columbia, MO	65202
Marlene Lee	612 Huntridge Dr.	65201
Sarah Connor	3301 Parker St	65202
John Clayton	2504 Slocumb	65201
Vic Myers	4508 Mexico Gravel Rd.	65202
Allen Haker	3711 [illegible] Col.	65201
Lorrie Pfeffer	1702 Lakewood Pl Col. MO	65202
Tasha Hussla	1205 Agan St. Col. Mo	65203
Nathan Fox	317 W 6th St Fulton, MO	65251
[illegible]	34 W 6th St Fulton, MO	65251
Phil Berg	6131 Hyde Ln MO	65203

23

21 Col
2 out

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 Cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Columbia Citizens for Residential Comp.

Name	Address	Zip Code
Dick Parker	215 W Sexton	65203
	Do not add me to any mailing lists	
Gail Plemmons	17 Aldeah	65203
Katie Fisher	815 Park	65117
Christina Easley	2709 W Rollins Rd Apt 2	65203
Mary Froese	3606 Southland	65201
Mark Walter	2315 Calmendows Dr	65202
Caroline Kane	613 E Green Meadows Apt 307 Columbia, MO 65201	
Linda Brown	7780 Hwy 11H Hallsville MO	65255
Barbara Peck	2315 Bridgewater Columbia MO	65202
Neal Mitchell	916 Sandra Ave Columbia MO	65202
Ronda Mitchell	916 Sandra Ave Columbia MO	65202

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 Cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Columbia Missouri Residents Only.

Name	Address	Zip Code
Wang Yizhu	Apt 1508, 1133 Ashland Road	65201
Catherine Prieke	1316 Rolling Rock Dr	65201
Mark Stevenson	3212 Shoreside Dr	65203
Peggy Placier	209 S. Greenwood Ave	65203
Mary Lehmann	2601 S. Providence	65203
Marybeth Champion-Garthe	4903 Craydon Dr	65203
~~Aaron Stoker~~	~~704 McNeal Ln~~	~~65251~~ X
Kelsey Wingo	402 N Tenth St	65201
Sarah Horton	1133 Ashland Rd Apt 210	65201
Christopher Oldfather	1133 Ashland Rd Apt 210	65201
SungWoo Kahng	6506 Rippling Water Way	65201
Mary De la Guerra	3001 S providence	65203

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Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 Cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Columbia Missouri Residents Only.

Print + Sign

Name	Address	Zip Code
Nancy F Br		
Nancy F. Browning	2235 Bluff Blvd. Columbia	65201
Jason McConnell	302 Alexander ave Columbia, MO	65203
Kent Melton	1511 Stone Street Columbia MO	65201
Robert Schreiber	1511 Stone St. Columbia MO	65201
Claire Garden	1404 Gary Col	MO 65203
Bryant Holliday	8 N. Keene St Apt A8	MO 65201
Victoria Sights	2401 E Walnut St Columbia, MO	65201
Lisa Toole	6506 Rippling Water Way Como	65201
Mike McMillen	Youngenty Red Col	MO
Anna Hulbert	1700 Fonm Blvd 503 Colum	MO 65203
Sam Bennett	606 Independence St	65203
Alexis Dockinson	2024 S El Centro Ct Columbia	65201

Petition ~~to the~~ Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 Cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Columbia Missouri Residents Only.

Name	Address		Zip Code
Sami Tellatin	402 N. 10th Street, Apt B	Columbia, MO	65201
Ian Thomas	Millcreek drive	Columbia MO	65201
Patricia Snarski	801 HIRTH ST	COLUMBIA MO	65201
KYLE FUQUA	" " "	" "	"
Kevin Smith	Homeless		
Mallory Braun	324 W Brookside Lane	Columbia, MO	65201
CHRIS IHGER	1611 CRYSTAL PT	COLUMBIA MO	65203
Cynthia Koehler	210 Wheaton CT	"	"
Tristan Laughlin	466 Rehaven Dr	Columbia MO	65202
Emilee Ramseur	1224 E. Walnut Apt. 203	Columbia, MO	65201
Brendan Merz	710 S. College Avenue ~~5768 Kellridge Lane~~	Columbia, MO ~~St. Louis, MO~~	65201 ~~63108~~
Nick Yahl	2900 Old 63 S	Columbia, MO	65201

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 Cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Name	Address	Zip Code
Margaret Bergund	1512 Richardson St.	65201
[illegible]	1503 STONE ST	65201
Zang Silver	800 Hardin St	65203
Crystal Sollenberger	5305 Edenton Blvd. Apt 102	65203
Lynda Petillon	2613 Whitegate Dr. Columbia MO	65202
Mary Hussmann	210 Ridgeway Columbia MO	65203
Jake Rains	PO Box 7353 Columbia, MO	65205
Roxanne Richardson	1707 Hinkson Ave Apt 5 Columbia, MO	65201
George Clark	707 Timberhill Rd	Columbia, MO 65201
Grant Davis	3909 Oakland #202	Columbia MO 65202
[illegible]	1103 Jean Rae	Columbia MO
NICK [illegible]	1510	COL, MO

24

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 Cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Columbia Missouri Residents Only.

Name	Address	Zip Code	
CARLA WENNERMARK	615 MORNINGSIDE DR.	65201	
Brad Spudich	409 N. 8th St.	65201	Local MU student
Kaitlin Rounds	1402A Bass Ave	66226	
Heather Gillick	St Joseph St	65201	
Seth Lanning	"	"	
Jo. Kuszaj	2401 Broadway	65203	
Kurt Kraftmeyer	P.O. Box 7091	65205	
[illegible] Hardin	915 Broadway	65201	
[illegible]	1402 Richardson	65201	
Abigail Stokes	1316 Raleigh Dr	65202	
Sarah Leitwater	1405 University Ave	65201	
Elise Schmelzer	409 N. 8th	65201	

24

2 out 22 in

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 Cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Columbia Missouri Residents Only.

Name	Address	Zip Code
Freya Rowland	4618 W. Gillespie Bridge Rd, Columbia MO	65211
Anne de Velasco	1313 Overhill Rd Columbia MO	65203
Bell Clark	1709 Oakwood Ct. Col. MO	65203
Nancy Boon	8251 So. Rt. N, Columbia, MO	65203
Madison Rhyner	1102 E Locust St Columbia, MO	65201
Maria Parker	215 W. Sexton Rd Columbia, MO	65203
Warfield Teague	3101 West Rt K, Columbia	65203
Jordan Richards	807 Hinkson Ave, Apt 5 Columbia MO	65201
Paul Richardson	1405 Virginia Lane Auburn, IN	46706
Sasha Schmid	2008 Devonshire Dr. Columbia MO	65203
Richard Knapp	2812 Pine Dr. Columbia MO	65202
John Richardson	640 Devonshire Drive Belleville IL	62226

outst town

outst town

2

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 Cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Name	Address	Zip Code
Ruth Schaefer	2756 C.R. 338	65063
Stephen Rawlings	2308 Whitegate Drive	65202
Cindi Rawlings	2308 Whitegate Drive	65202
Bethany Rawlings	2309 W Broadway Apt 123,	65203
Carol Bocrady	810 Amy Dr	65201
J EDM	810 Amy Dr	65201
Phil Gibson	604 Rockhill Rd	65201
Marsha Maxwell	604 W 8th St	65201
Kassie Elliott	116 S College #2	65201
~~Sherry Gallen~~	~~[illegible]~~	
Erin Pearson	200 St Joseph St	65201
Chris Murukami	208 S Joseph St	65201

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 Cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Name	Address	Zip Code
Alex Clark	404 Kentucky Blvd	65201
Beatriz Machado	404 Kentucky Blvd	65201
Pam Bufford	2601 Ridgefield	65203
Glenna Stallee	608 William St	65201
Morgan Cooper	112 Antelope Dr.	65202
Clinton Cooper	112 Antelope Drive	65202
Amanda Lake	2212 Live Oak	65202
Alex Lake	2212 Live Oak	65202
Craig Cyr	8151 W HWY UU	65203
Alyce Turner	1204 Fieldcrest	65203
Jan Shye	2222 Bluff Blvd	65201
Elizabeth Scutti	41909 alpineridge dr.	65202

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 Cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Columbia Missouri Residents Only.

Name	Address	Zip Code
Mary Hulner	1200 Broadway	65201
Adrienne Stolwyk	214 St. Joseph St	65201
Badger Johnson	105 3rd Ave	65203
[illegible]	1100 Park Ave	65201
Out of town Ellis Froeschle	301 Second St Rocheport, MO	65279
Kara [illegible]	4400 West Broadway	65203
Anne Baumann	5515 S. Bethel Church Rd	65203
Out of town Jeffrey A. [illegible]	18571 S. Alma Ct., Hartsburg, MO	65039
[illegible]	902 S. Glenwood Ave.	65203
Olga Shelton	512 Noble Court, Columbia, MO	65203
Karen Dwyer	2255 Bluff Blvd	65201

Out of town 2

23

21 Col.

OVER

6

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 Cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Columbia Missouri Residents Only.

Name	Address	Zip Code
Punky Heppner	8040 Turtle Creek Columbia MO	65203
Mary Heppner	8040 Turtle Creek Columbia MO	65203
Tammy Phanichkul	1309 Pickford Place Apt A	65203
RICHARD STAHLHUT	194 W GREEN MEADOWS	65203
[illegible] Pollard	1619 WINDSOR COL. MO	65201
Dianna Kaino	117 Elk Dr, Como	65202
Margaret Vatterott	606 E Rollins Street	65202
Julia Guyton	1700 Forum Blvd	65203
Jackie Dearborn	1700 Forum Blvd	65203
Therese Folsom	111 N. Greenwood Ave	65203
Julie Donnelly	6901 W Charlene	65202
Cathy Harris	Columbia MO	65201

This is the description for the on-line petition.

Please ONLY Residents of Columbia, Missouri, USA should sign. We appreciate the good wishes of those outside the city limits but we need to keep this petition local to show the City Council of the broad-based support for this change. In the why this is important to me, you could also enter your Columbia, MO address for reference by your council person. Single-use plastic bags require petroleum oil to produce and energy to transport to store locations. 12 million barrels of oil are used to produce the 100 billion bags used each year in the United States. Most bags are used for less than 20 minutes and then thrown in the trash. Less than 12% are recycled. Plastic bags never biodegrade. Instead they break into very small pieces of plastic that pollute the soil and water and are eaten by fish and other wildlife. These bits then enter the human food supply.

In view of the need to reduce our usage of fossil fuels and the impact of this plastic pollution on wildlife and human health, we need to move to a more sustainable approach including greater usage of reusable bags.

The proposed ordinance would ban the store providing single-use plastic bags at check-out for stores selling groceries, pharmacies and convenience stores. Bags used for produce, wrapping meat or carrying bulk food items to check-out will still be allowed. Paper bags would be sold at 10 cents each with some exemptions for persons on food assistance programs.

On-line petition

#	Name	Country
1	[redacted]	United States of America
2	Claire Lea	United States of America
3	Lorri Auer	United States of America
4	Annie	United States of America
5	Jan	United States of America
6	Dean Andersen	United States of America
7	Susan Schabilion	United States of America
8	Peter Holmes	United States of America
9	David Finke	United States of America
10	Pippa Letsky	United States of America
11	Irene Serrano	United States of America
12	Cheryl Guthrie	United States of America
13	Megan E. Moore	United States of America
14	Mark Osborn	United States of America
15	Jennifer Davis	United States of America
16	Brian Powell	United States of America
17	Jackie Casteel	United States of America
18	Holly Froeschner	United States of America
19	Deb Jewett	United States of America
20	Rebecca Scott\	United States of America
21	Cully Meier	United States of America
22	Renee Maxwell	United States of America
23	Dustin Thewind	United States of America
24	Brock Williams	United States of America
25	Stephen Mudrick	United States of America
26	Howard Hutton	United States of America
27	tammy	United States of America
28	Tricia Price	United States of America
29	[redacted] Dup	United States of America
30	Heather Myers	United States of America
31	[redacted]	[redacted]
32	Anne Stanton	United States of America
33	[redacted]	[redacted]
34	david roberts	United States of America
35	Mary Protzman	United States of America
36	C. Boley	United States of America
37	Jaylynn Gentry	United States of America
38	Robert Sapp	United States of America
39	Debra Hawley	United States of America
40	Mark Haim	United States of America

#	Name	Country
41	Jeremiah Riely	United States of America
42	Jean Blackwood	United States of America
43	Frank Shulse	United States of America
44	Maroua Jawadi	United States of America
45	Mike Diel	United States of America
46	Tim sparling	United States of America
47	Kara	United States of America
48	mary denson	United States of America
49	Katherine	United States of America
50	[redacted]	[redacted]
51	[redacted]	Peru
52	Shannon Morris	United States of America
53	Linda	United States of America
54	[redacted]	[redacted]
55	Dustin Salter	United States of America
56	Samantha	United States of America
57	Phil Wood	United States of America
58	Marianne Branham	United States of America
59	Paul Blackwell	United States of America
60	Kim Jasmer	United States of America
61	Camellia Cosgray	United States of America
62	K	United States of America
63	Rebecca	United States of America
64	Margaret Waddell	United States of America
65	Marya Liberman	United States of America
66	Kathy Bennett	United States of America
67	Polita Sweitzer	United States of America
68	Randy Tindall	United States of America
[redacted]	[redacted]	[redacted]
	nmark	
[redacted]	[redacted]	Netherlands
[redacted]	[redacted]	[redacted]
72	Aly B	United States of America
73	Randy	United States of America
74	Naomi	United States of America
75	[redacted]	United States of America
76	Josh Crow	United States of America
77	Nancy Flores	United States of America
78	Clarise Keith	United States of America
79	Meg Rivers	United States of America
80	Steve Andsager	United States of America

#	Name	Country
81	Candace	United States of America
82	Melissa Silvey	United States of America
83	Zac Early	United States of America
84	Nicole Danielle	United States of America
85	Elizabeth Harding	United States of America
86	Elaine Myers	United States of America
87	Allison Starn	United States of America
88	Tara Barnett	United States of America
89	Natasha Hernandez	United States of America
90	Donna Reed	United States of America
91	Stephanie Wyatt	United States of America
92	Blanca Kelty	United States of America
93	Christy Henley	United States of America
94	Deb Nagel	United States of America
95	L LaVonne Patterson	United States of America
96	Carolyn Mathews	United States of America
97	Alexandra Prestia-Turner	United States of America
98	Megan Bloom	United States of America
99	Karen John	United States of America
100	Greg Leonard	United States of America
101	Allison Kellenberger	United States of America
102	jenifer mckenzie	United States of America
103	Les Paris	United States of America
104	Cara Christianson	United States of America
105	Kelli Quinata	United States of America
106	Layla Alazawy	United States of America
107	Veronica DeStefano	United States of America
108	Karon Rush	United States of America
109	Cleo Kottwitz	United States of America
110	Josie Sullivan	United States of America
111	Becky Erickson	United States of America
112	Monica Lee	United States of America
113	Kirsten	United States of America
114	[redacted] DNP	United States of America
115	Linda Green	United States of America
116	Laura Zimmerman	United States of America
117	Mary L. Lehmann	United States of America
118	Janice Albers	United States of America
119	Lois Bennett	United States of America
120	Zach Rubin	United States of America

#	Name	Country
121	Kelsey Dexter	United States of America
122	Meredith Morrow	United States of America
123	Melissa MacGowan	United States of America
124	Luna Langer	United States of America
125	Danielle Clements	United States of America
126	Amy Meier	United States of America
127	Lacy Hicks	United States of America
128	Sarah Linsenmeyer	United States of America
129	Katy Miller	United States of America
130	Alicia LaVaute	United States of America
131	Heather	United States of America
132	lori mccurdy	United States of America
133	Haley Schwarz	United States of America
134	samae claspill	United States of America
135	Matt Rahner	United States of America
136	Thomas Dillingham	United States of America
137	Cathie Brooks	United States of America
138	D	United States of America
139	Katie	United States of America
140	Heather Pyles	United States of America
141	Molly Fiegel	United States of America
142	Joy Amuedo	United States of America
143	Shannon Diaz	United States of America
144	Jolene Kington	United States of America
145	Enid Schatz	United States of America
146	Michael Marlo	United States of America
147	Auben Galloway	United States of America
148	cheri werner	United States of America
149	Richard King	United States of America
150	Rigel Oliveri	United States of America
151	Lisa Groshong	United States of America
152	Jim Collier	United States of America
153	John Stewart	United States of America
154	Kristopher	United States of America
155	Sherri Sachdev	United States of America
156	Alex D.	United States of America
157	Steven Thompson	United States of America
158	[redacted]	[redacted]
159	Ashlee	United States of America
160	Kim Hilden	United States of America

#	Name	Country
161	Hoyt DeVane	United States of America
162	Anna Sheals	United States of America
163	Endi	United States of America
164	Todd Berchek	United States of America
165	Laura Pintel	United States of America
166	MICHELLE MILLOT	United States of America
167	Sara Strand	United States of America
168	Becky Cowles Gibbs	United States of America
169	kirk utterback	United States of America
170	Jenna Rose	United States of America
171	Pablo Araujo	United States of America
172	Erica Eisenberg	United States of America
173	Megan	United States of America
174	SwK	United States of America
175	Chrissie Merideth	United States of America
176	Barbara Dawdy Wills	United States of America
177	Anh Braddock	United States of America
178	Catherine A Johnson	United States of America
179	Audrey McFadden	United States of America
180	Emily Thornton	United States of America
181	David Dittmer	United States of America
182	Dawn Brown	United States of America
183	Ellen Wilson	United States of America
184	Meg Gibson	United States of America
185	Claire	United States of America
186	Greg Flippin	United States of America
187	Jon Hagerman	United States of America
188	Jennifer Ann Wiggs	United States of America
189	Megan Stilley	United States of America
190	Stephanie buckridge	United States of America
191	Dan Hemmelgarn	United States of America
192	Amber Cox	United States of America
193	Stephen Bybee	United States of America
194	Richard Perkins	United States of America
195	Joy Castillo	United States of America
196	Michael lee Robins	United States of America
197	Toni Stark-Buckler	United States of America
198	Angela Speck	United States of America
199	brandi	United States of America
200	Sarah Smith	United States of America

#	Name	Country
201	LaTisha Rodriguez-Johns	United States of America
202	Elijah Fortel	United States of America
203	Amanda Carr	United States of America
204	Lisa Rohmiller	United States of America
205	Scott Wilson	United States of America
206	L. Coggeshall	United States of America
207	jawnie	United States of America
208	Katherine Beckner	United States of America
209	Mona Halaweh	United States of America
210	Kay Allen	United States of America
211	Gabriella McCord	United States of America
212	Stefanie Maurer	United States of America
213	Stephanie priesmeyer	United States of America
214	Chris Baker	United States of America
215	Barbara	United States of America
216	Claire	United States of America
217	Katie	United States of America
218	Michelle Windmoeller	United States of America
219	Stacey Kulik	United States of America
220	Jonas Weir	United States of America
221	Scott	United States of America
222	Sean Witzman	United States of America
223	Rachel Ruhlen	United States of America
224	Roger	United States of America
225	Lauralee Sparling	United States of America
226	Sherri B.	United States of America
227	Madison Russell	United States of America
228	Justin Gramarye	United States of America
229	Susan McCullough	United States of America
230	Shannon Slavit	United States of America
231	[redacted]	United States of America
232	Kara Brooks	United States of America
233	Terry Rice	United States of America
234	Abby St. George	United States of America
235	Bryan tanner	United States of America
236	Devin Frank	United States of America
237	Renee Cusick	United States of America
238	Meg Poché	United States of America
239	elissa masters	United States of America
240	Laura Kessel	United States of America

#	Name	Country
241	Cody Davis	United States of America
242	megan	United States of America
243	Elizabeth Bryan	United States of America
244	Jessica Barnett	United States of America
245	Surly Shawn	United States of America
246	Julie Baka	United States of America
247	Rebecca Maclin	United States of America
248	Katy McDonald	United States of America
249	Julie Boyle	United States of America
250	Ruth	United States of America
251	Courtney E. Mudd	United States of America
252	kelsey kertz	United States of America
253	Jennifer sieradzki	United States of America
254	Leslie Gillin	United States of America
255	Erin Blaise	United States of America
256	Sara Loveless Mueller	United States of America
257	Barbara Devine	United States of America
258	Liz	United States of America
259	aaron scott	United States of America
260	Julie N	United States of America
261	Mikayla	United States of America
262	Alfredo	United States of America
263	Sky Jimenez	United States of America
264	Kendall irwin	United States of America
265	Jackson Rothschild	United States of America
266	[redacted] PW	United States of America
267	Steven	United States of America
268	Melissa Click	United States of America
269	Jessica Blaine McBlainerson	United States of America
270	Shauna Marquardt	United States of America
271	Billie Stock	United States of America
272	Logan Epps	United States of America
273	Tom Leahy	United States of America
274	Sara Bartow-Fuller	United States of America
275	Laura	United States of America
276	Terry Johnson	United States of America
277	Lacy Clark	United States of America
278	Chloe Martinez	United States of America
279	Nita R	United States of America
280	Heidi Stallman	United States of America

#	Name	Country
281	katy hudler	United States of America
282	Marlene Stevens-Hanson	United States of America
283	Rachel Taylor	United States of America
284	Dee	United States of America
285	Rachael Hahn	United States of America
286	Maura Mudd	United States of America
287	Melissa maddox	United States of America
288	Britta Simpson	United States of America
289	Sabrina Marshall	United States of America
290	John Doerflinger	United States of America
291	ciera	United States of America
292	Katherine	United States of America
293	Sharon Curran	United States of America
294	Julie Evenson	United States of America
295	Alissa Smith	United States of America
296	donna	United States of America
297	Thomas Andes	United States of America
298	Eva Petakovic	United States of America
299	Linda	United States of America
300	Bernadette Dryden	United States of America
301	Shayna Buckridge	United States of America
302	Suzanne VanSickle	United States of America
303	Carole M Simon	United States of America
304	Marion Mace	United States of America
305	Pat Kay	United States of America
306	Adam Rolfes	United States of America
307	[redacted]	[redacted]
308	Brett Moore	United States of America
309	Carol Lane	United States of America
310	Elizabeth Behrens	United States of America
311	Kim Reardon	United States of America
312	Tia Marie Rogers	United States of America
313	Ashley	United States of America
314	Chelsea Olson	United States of America
315	Shashikanth	United States of America
316	Gajaraj	United States of America
317	Sherri Mankofsky	United States of America
318	Sam	United States of America
319	Alex Prentice	United States of America
320	Mark Bosky	United States of America

#	Name	Country
321	Sarah Gaston Carroll	United States of America
322	Sara	United States of America
323	Gretchen Maune	United States of America
324	Layla	United States of America
325	Alyce Turner	United States of America
326	Jeanette Shoemaker	United States of America
327	Linda Vandepopuliere	United States of America
328	sandra loar	United States of America
329	CJ Strawn	United States of America
330	Jill Edwards	United States of America
331	Anna Lindell	United States of America
332	Jill McReynolds	United States of America
333	Tom Lindell	United States of America
334	Pat Fowler	United States of America
335	Seymore Butts	United States of America
336	Mike Roch	United States of America
337	Ben Dover	United States of America
338	Hank E. Pankie	United States of America
339	Suzette Vos	United States of America
340	[redacted]	United States of America
341	[redacted]	United States of America
342	[redacted]	United States of America
343	[redacted]	United States of America
344	[redacted]	United States of America
345	Pat McDonald	United States of America
346	Dylan Gelbach	United States of America
347	Boyd ODell	United States of America
348	Charity Clark	United States of America
349	Tess Whitehurst	United States of America

349
−17 Fake or out of country
332 signatures
− 4 Duplicates of other sheets
328 signatures

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. Non-Columbia residents only

Name	Address	Zip Code	Email address (optional)
Heather Wilson	3738 Main St. Jamestown, MO	65046	heathershomegroundesigns@gmail.com
Donald Penny mayor Lopes Mo	3840 Adams Jamestown, MO	65046	
Sarina Wolfe	3840 Adams St. Jamestown, MO	65046	
Joseph Klinefelter	3834 Adams St Jamestown MO	65046	
Georgia Klinefelter	9201 E Davis Centralia, MO	65240	earlhandledpistol@gmail.com

5

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 Cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. For OUT OF TOWN RESIDENTS.

Name	Address	Zip Code	Do you shop in Columbia? Yes or No?
Melissa Hope	715 Rt M, JC, MO	65101	yes
Terica Bergin	9833 CR 435	65065	yes
Scott Pauley	5701 E. Claysville Rd.	65039	yes
Jon Read	1000 Riverva Dr, Boonville	65233	yes
Bonnie Yarnaguist	4435 Hwy PP	65202	yes
Darry Hallam	1719 Ridgemont	65203	yes
Sam Meyer	1000 Riverva Dr. Boonville MO	65233	
Karen Westin	11657 FLETCHER LN, HS, MO	65043	Some
Walter Baragan	PO Box 19, Ashland	65010	Yes

9

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 Cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. For OUT OF TOWN RESIDENTS.

Name	Address	Zip Code	Do you shop in Columbia? Yes or No?
Brett Dufur	204 Columbia Rochefort MO	65279	
Dejan Pursell	15001 Sworn School Rd	65039	yes
Brandon Butler	129 Meadow Lake Ln Fayette, MO	65248	Yes
Susan Barnes	2451 E. Nashville Church Rd Ashland, MO	65010	yes
Todd Schneider	111 Wagonwheel Dr.	65109	yes
Danielle Evans	111 WagonWheel Dr	65109	Yes
Angela Schneider	111 Wagonwheel Dr	65109	yes
D. Joseph Porzo	1700 E. Walnut Street Springfield, MO	65807	yes
Katherine Baumchter	821 E Loren Springfield MO	65807	yes

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 Cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. For OUT OF TOWN RESIDENTS.

Name	Address	Zip Code	Do you shop in Columbia? Yes or No?
John Brady	PO Box 139, Rocheport MO	65279	Yes
Galine Phillips	16303 Lipscomb Rd.		NO
8422 KIRSCHNER	812 N. MULBERRY	64658	YES
Pat Farrar	15555 Wren School Rd, Hartsburg	65040	yes
Beth Pearock	Rocheport MO		yes
Ron Frederick	6305 Oakmont Dr. Niya MO.	65714	yes
St. Louis	3214 magnolia S-W	63118	YES
Dave Huffman	9802 E Hwy WW Columbia MO	65201	yes
Matt Vitello	2131 Tanner Bridge Rd Jefferson City MO	65101	Yes

9

Petition to the Columbia City Council

I, the undersigned, would like to indicate I support a single-use plastic bag ban in Columbia, Missouri. I encourage the City Council to implement an ordinance to ban single-use plastic bags and implement a small charge (10 Cents) for recyclable paper bags that stores may choose to offer their customers as an alternative to single-use plastic bags.

Please write legibly. For OUT OF TOWN RESIDENTS

Name	Address	Zip Code	Do you shop in Columbia? Yes or No?
STEPH IRWIN	7065 CC RD 400	MOKAME MO 65#059	YES
RS YOUNGQUIST	4435 Hwy PP	Columbia 65202	YES
Diana Papantias	3951 County Rd 259	Fulton MO 65251	yes
John Holdmeier	110 E 5th street	Washington, MO 63090	yes
Barbara Elmore	8780 N Millsite Rd	Col 65201	yes
Kitty Peer	14701 S. Windmiller	Ashland 65010	yes
Judey Brown	11801 E Englewood Rd,	Ashland 65010	Yes!
Mark Walters	2515 Oakmeadows Dr.	65205	Yes
Judy Haney	1515 INVERNESS CT	65203	yes

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Please write legibly. For OUT OF TOWN RESIDENTS.

Name	Address	Zip Code	Do you shop in Columbia? Yes or No?
[illegible]	[illegible]	[illegible]	[illegible]
JOHN DELUREY	4134 CHOUTEAU AVE	63110	Yes
[illegible]	[illegible]	[illegible]	[illegible]
Tim Grace	~~P.O.~~ 381 Shady Ln	65062	Yes

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Please write legibly. Non-Columbia residents only

Name	Address	Zip Code	Email address (optional)
Stephanie ~~Stephanie~~ Yoakum	4122 W. Bethany	65203	stephanie.yoakum@gmail.com
Brian Powell	3575 N. Frederick Ct	65202	bp8292@gmail.com
Amanda Frevert	105 A. Sanders Ct	65202	amandafrevert@hotmail.com
Mary Lottes	10201 Burnett School	Ashland, 65010	
Sarah Anderson	9750 N Phillips Rd	65255	
EDWARD ARMOUR	100 17th St CROCKER	65452	
Larry Brown	5635 Loose Drive, Fulton	65251	
Dana Morris	185 C.R. 441, New Franklin	65274	
Maxine Mars	209 Ash Ashland	65010	
Ed Hamm	625 E. Glenwood SPFD	65807	
JONATHAN LAUTEN	2823 RACKERS FORD RD	J.C., MO 65101	
Debra Manning	1116 Anna Marie Dr.,	Jonesburg, MO 63351	
Michael Ugarte	1505 Windsor St.	Columbia MO 65201	

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Name	Address	Zip Code	Do you shop in Columbia? Yes or No?
Kory Kaufman	720 Hickory Grove School Rd	65279	Yes
Thomas Bothwell	~~[illegible]~~ 405 Claybrooke	63122	Yes
Ina Casagrand	1006 E. High St. Apt. #4 JCMO	65101	Yes!
David Elsberry	15,555 Wrenschool Rd Hartsburg	65039	Yes always
MATTHEW J BARNES	2451 E. NASHVILLE CHURCH RD ASHLAND, MO	65010	YES
Jacqueline Acton	529 E. High apt C Jefferson City, MO	65101	yes
Duncan Foss	412 Vetter Ln Jefferson City, MO	65101	Yes
Kristen Schulte	128 Visitlago Place, Cottleville, MO ~~109 Woodtown Drive Columbia, MO~~	63376	Yes
Melanie Cheney	10450 S. Route N, Columbia MO	65203	Yes!

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Please write legibly. For OUT OF TOWN RESIDENTS.

Name	Address	Zip Code	Do you shop in Columbia? Yes or No?
Mary B Rose Mary B Rose	13940 Veach Rd Ashland	65010	yes
[illegible]	10931 N Calhoun	65202	yes
[illegible]	781 W Walter Dr	65203	yes
Ann Wilkins	943 Forest Ave	65201	yes
Bobbi Davis	781 County Rd 128	65243	yes

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Please write legibly. For OUT OF TOWN RESIDENTS.

Name	Address	Zip Code	Do you shop in Columbia? Yes or No?
Rita Morgan		65203	YES
Michael Frank	12901 S. Joy Rd Ashland MO	65010	YES
Shauna Brandwein	1077 Co Rd 1051 Martinsburg Mo	65264	yes
Dei [illegible]	1077 Co. Rd. 1051 Martinsburg Mo	65264	yes
John L. Baron	2104 Franklin Dr. Sedalia		yes
Dianna Stark	4440 W. Country Hill	65203	yes
Andja Abdurahmanovic	5210 W. Southeast Ct.	65202	Yes
Jamie Echols	4611 W. Knox Dr.	65203	yes

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Name	Address	Zip Code	Email address (optional)	
DIANE VITELLO	2131 Tanner Bridge	65101		
Diane Dorton	104 N Polk Jefferson City Mo	65101		out of town
Chris Felber	104 N Polk Jefferson City MO	65101		out of town
MIKE SMITH	701 E. 1st ST WASHINGTON, MO	63090		out of town
[illegible]	104 N polk Jefferson City MO	65101		out of town
Ward Miller	7115 Route M J.C. MO	65101		out of town
Katie Duel	202 Lark Hill Ln.	63021	Katie.p13@live.com	
Jacob Sprech	202 Lark Hill Ln	63021	KIsiecho@Hotmail.com	
Sarah Davis	2701 W Dripping Springs Rd	65202		
Heather Hoyle	9903 Hwy 87 Prairie Home	65068	greengeat@mac.com	
Jordan Olson	502 S Main St Warrensburg	64093	j.olson.amd@Hotmail.com	out of town
Keith Murray	502 S Main St Warrensburg	64093	Keith.l.Murray@gmail.com	
Peter Beiger	1411 PRATT	65201	[illegible]	

Petition to the Columbia City Council

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Please write legibly. For OUT OF TOWN RESIDENTS.

Name	Address	Zip Code	Do you shop in Columbia? Yes or No?
PEGGY ESCH	1116 E Rt M Hartsburg Mo	65039	yes
Linda Green	206 Anderson Ave. Columbia Mo	65203	Yes
Melanie Cheney	10450 S. Route N	Columbia 65203	yes
Terry Gainey	1515 Inverness Ct.	Columbia 65203	yes
Jody L. Frank	12901 S. Joy Rd	Ashland MO 65010	Yes
Crystal Essen	411 Billy Joe Supp Dr.	Ashland MO 65010	Yes
Cindy Quick	4250 Twin Oaks Dr	Fulton MO 65251	yes
Guy Rhoads	5830 Creasysprings Rd	Columbia MO 65202	yes
Pam Rhoads	5830 N. Creasysprings Rd.	Columbia MO 65202	yes

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Please write legibly. Non-Columbia residents only

Name	Address	Zip Code	Email address (optional)
Tricia Woolbright	1400W HighPoint Lane	65203	trish.woolbright@gmail.com

City of Columbia

701 East Broadway, Columbia, Missouri 65201

Agenda Item Number: B 51-15
Department Source: City Manager
To: City Council
From: City Manager & Staff
Council Meeting Date: 2/16/2015
Re: Ordinance Pertaining to the Regulation of Single-Use Plastic Bags

Documents Included With This Agenda Item

Council Memo, Resolution/Ordinance
Supporting documentation includes: Missouri Farmer's Association (MFA) Response to Proposed Ordinance, Hy-Vee Response to Proposed Ordinance, American Progressive Bag Alliance (APBA) Response to Proposed Ordinance

Executive Summary

As an accompaniment to the proposed ordinance pertaining to the regulation of single-use plastic bags, the following report provides estimates of the cost, staff and volunteer time spent on abating plastic bag waste; an approximate baseline of current city-wide plastic bag usage; reasons plastic bags cannot be recycled at the City's Material Recovery Facility (MRF); and a summary of retailer feedback to the proposed ordinance.

Discussion

City of Columbia Litter Abatement Costs Related to Plastic Bags

As a result of local litter contamination, the City of Columbia coordinates a number of cleanup programs, from individual park and stream clean-up days to Adopt-a-Spot Litter and the Hinkson Creek Clean Sweep, resulting in an estimated $177,448[1] worth of annual volunteer hours.[2] City staff estimates that plastic shopping bags rank third on the list of local litter pollutants, representing a good portion of the time and value spent cleaning up the city.

Moreover, in 2014, the city's landfill spent more than $45,000 specifically on litter control, while its MRF spent approximately $2,700 in litter policing, a total of $47,700. Considering solid waste staff estimates that 1/4 to 1/3 of this litter is plastic bag waste, the city spends anywhere from $11,925 to $15,900 on controlling plastic bag waste at our landfill and MRF annually.[3]

Estimated City of Columbia Annual Plastic Bag Use

A number of public agencies have estimated that the average American uses 350-500 plastic bags annually[4]. At Columbia's current population of 115,276[5], this amounts to the annual distribution of approximately 40 to 57 million plastic shopping bags within the city.

Challenges to Recycling Plastic Bags at Columbia's Material Recovery Facility

The City of Columbia does not accept single-use plastic grocery bags for recycling, because its

Material Recovery Facility does not have enough space to store enough plastic bags to cost-effectively recycle them. In addition, the plastic bags can get caught in the recycling sorting machinery and cause equipment jams. City Solid Waste staff suggests that customers recycle their plastic bags at grocery store receptacles, which can be found at Wal-Mart, Gerbes, Schnucks and Hy-Vee.

Local Retailer Feedback on Proposed Ordinance
Per council request, staff sought feedback on the proposed ordinance from Lucky's Market, Wal-Mart, Gerbes, Schnucks, Hy-Vee, Casey's General Store and Break Time stores (owned by MFA Oil). The manager of Lucky's Market stated that the business fully supported a full ban of single-use plastic bags but did not support any type of charge on paper bags, as such a charge is seen as a punishment to their customers. Lucky's prefers to incentivize customers to utilize reusable bags through the provision of refunds (accompanied by an opportunity to donate refunds to local causes).

MFA Oil, the owner of Break Time stores, provided a letter describing the reasons it does not support the ordinance. MFA views the ordinance as an unfair burden to its customers and employees that will add to the current challenges faced by the stores trying to adapt to a number of other new municipal ordinances, including Ban the Box and Tobacco 21. Hy-Vee also provided written feedback to the potential ordinance, comprised of its 2009 response to attempts to pass similar ordinances in Iowa. Hy-Vee suggests that in place of an ordinance local stores focus on efforts to reduce, reuse and recycle plastic bags.

The other stores contacted by staff did not provide a response to the proposed ordinance.

In addition, APBA, a trade organization representing plastic bag manufacturers, submitted a detailed report that challenges statements included in the City's Environment and Energy Commission report on the single-use plastic bag ordinance.

[1]Estimates based on FY14: Neighborhood Clean-ups:128.5 hours; Clean-up Columbia:2545; Litter-ONS:15: Park Clean-Ups:393.8; Eco-Clean-ups:515.5; Adopt-a-Spot:3192.3; Crawdads: 717; Hinkson Clean Sweep: 318; Stream Clean-up: 44. Total hours on city letter abatement efforts: 7869.1, resulting in an estimated annual value of $177,448.
[2]The hourly value of volunteer time in Missouri is $22.55, according to the Corporation for National & Community Service http://www.volunteeringinamerica.gov/
[3]The city pays temporary laborers through Job Finders, Labor Ready, etc. regularly on good-weather days and also usually has 1 Temp employee that is typically doing litter collection. Litter Collection hours are recorded in a daily log.
[4]Town of Truckee, Single Use Bag Ordinance, Consumer Information
http://www.townoftruckee.com/departments/solid-waste-recycling/single-use-bag-ordinance
[5]Current U.S. census estimates
http://factfinder2.census.gov/faces/tableservices/jsf/pages/productview.xhtml?src=CF

Fiscal Impact

Short-Term Impact: The city will not collect any of the bag fees associated with the ordinance.

However, there could be potential costs associated with the enforcement of the ordinance.
Long-Term Impact: Unknown

Vision, Strategic & Comprehensive Plan Impact

Vision Impact: Environment

Strategic Plan Impact: N/A

Comprehensive Plan Impact: Environmental Management

Suggested Council Action

Staff recommends that Council table the vote on this ordinance for one year, in order to allow staff time to collect more input from the general population and retailers, and to potentially engage in a community education outreach effort on this issue.

Legislative History

At the October 20, 2014 city council meeting, Jan Dye and Frederick Vom Saal brought the Single-use plastic bag ordinance to city council for consideration, and Ms. Hoppe requested that the Energy and Environment Commission develop a report relating to the potential ordinance. At the 1/5/15 council meeting, Mr. Skala requested that EEC's report on the plastic bag ordinance be sent to retailers in Columbia requesting a formal response to the ordinance, and input on how they would like to approach the challenge of plastic bag waste. At the 1/20/15 council meeting, Ms. Hoppe made a motion to have a first read of the ordinance at the 2-16-15 council meeting. Mr. Thomas seconded the motion.

Leah Christian
Department Approved

City Manager Approved

Introduced by _____

First Reading _____ Second Reading _____

Ordinance No. _____ Council Bill No. _____B 51-15_____

AN ORDINANCE

amending Chapter 11 of the City Code to add a new Article XII pertaining to the regulation of single-use plastic bags; and fixing the time when this ordinance shall become effective.

WHEREAS, the City of Columbia seeks to protect our waterways, local streams and the Missouri River; and

WHEREAS, single-use plastic carryout bags pollute our local streams and rivers and ultimately break down into smaller bits that contaminate soil and waterways and enter into the food supply that humans, animals and aquatic life ingest; and

WHEREAS, the City Council of the City of Columbia seeks to reduce the toxicity of waste materials in the solid waste stream that are directed to resource recovery and sanitary landfill facilities, and to maximize the removal of plastic carryout bags from the waste stream; and

WHEREAS, the City of Columbia signed Resolution 160-06A, endorsing the U.S. Mayors Climate Protection Agreement striving to meet or exceed Kyoto Protocol targets for reducing global warming pollution by taking actions in our community; and

WHEREAS, the production of plastic bags worldwide requires the use of more than twelve (12) million barrels of oil per year, which also has a significant environmental impact; and

WHEREAS, many chemicals in plastic products are now known to cause harm, but the chemicals present in plastic bags are not disclosed to the public and are not required to be tested for health effects, so chemicals in plastic bags cannot be assumed to be safe; and

WHEREAS, the City of Columbia has shown an interest in being a leader in initiatives with a positive effect on climate change through its curbside recycling program, methane gas bioenergy facility, solar energy array, LEED certified city buildings and increased use of renewable energy; and

WHEREAS, the City Council finds it in the best interest of the City to regulate the use of single-use plastic bags within the city limits.

NOW, THEREFORE, BE IT ORDAINED BY THE COUNCIL OF THE CITY OF COLUMBIA, MISSOURI, AS FOLLOWS:

SECTION 1. A new Article XII of Chapter 11 of the Code of Ordinances of the City of Columbia, Missouri, is hereby enacted as follows:

Material to be deleted in ~~strikeout~~; material to be added underlined.

ARTICLE XII. ~~RESERVED~~ SINGLE-USE PLASTIC BAGS REDUCTION

Sec. 11-346. Definitions.

The following definitions apply to this article:

Customer means any person purchasing or obtaining goods from a store.

Post-consumer recycled material means any material that would otherwise be destined for solid waste disposal, having completed its intended end use and product life cycle. This does not include materials and byproducts generated from, and commonly reused within, an original manufacturing and fabrication process.

Product bags means a bag without handles that is used to carry items to the point of sale, including:

(1) Bulk items, such as nuts, grain or candy;

(2) Meat, poultry, or fish, whether packaged or not;

(3) Flowers, potted plants, or other items where dampness needs to be controlled;

(4) Unwrapped prepared foods or bakery goods; or

(5) Fresh fruits or vegetables.

Recyclable paper bag means a bag that is made predominantly of paper and meets the following requirements:

(1) Contains no old growth fiber;

(2) Contains a minimum of forty percent (40%) post-consumer recycled material; and

(3) Displays the word “Recycle” or “Recyclable” or the universal recycling symbol on the outside of the bag.

Reusable bag means a bag with handles that is specifically designed and manufactured for reuse and meets the following requirements:

(1) Is machine washable or is made from a material that may be cleaned or disinfected;

(2) Does not contain lead, cadmium, or any other heavy metal in toxic amounts; and

(3) If made predominantly of plastic, is a minimum of at least 2.25 mils thick.

Single-use plastic bag means a bag that is made predominantly of any type of plastic, including degradable or biodegradable; provided, however, single-use plastic bag shall not include a reusable bag or a product bag.

Store means any self-service retail establishment that sells a line of dry grocery, canned goods, frozen food and perishable items, including, but not limited to, drug, pharmacy, supermarket, grocery and convenience stores.

Sec. 11-347. Single-use plastic bags prohibited.

(a) It shall be unlawful for any store to provide to any customer a single-use plastic bag for the purpose of enabling the customer to carry away goods from the point of sale, except as otherwise provided in this article.

(b) A store may provide recyclable paper bags as set forth in section 11-348, reusable bags or any combination thereof, to customers for the purpose of enabling the customer to carry away goods from the point of sale.

(c) This article does not apply to any type of bag that the customer brings to the store.

(d) This article does not apply to plastic or paper bags supplied by the store to enable a customer to carry hot prepared ready-to-eat food for consumption off the premises.

Sec. 11-348. Recyclable paper bag fees.

(a) When a store provides a recyclable paper bag to a customer for the purpose of enabling the customer to carry away goods from the point of sale, the store shall:

(1) Charge the customer a fee of not less than ten (10) cents per bag; and

(2) Indicate on the customer's transaction receipt(s) the count of recyclable paper bags provided as well as the total charge for the bags provided.

(b) The requirements set forth in subsection (a) of this section shall not apply to customers using food assistance programs such as Women, Infants, and Children (WIC) and Supplemental Nutrition Assistance Program at the point of sale.

SECTION 2. This ordinance shall be in full force and effect from and after October 1, 2015.

PASSED this _____ day of _____, 2015.

ATTEST:

_____ _____

City Clerk Mayor and Presiding Officer

APPROVED AS TO FORM:

_____

City Counselor

City of Columbia
701 East Broadway, Columbia, Missouri 65201

SUPPORTING DOCUMENTS INCLUDED WITH THIS AGENDA ITEM ARE AS FOLLOWS:

1) MFA's Response to Proposed Ordinance
2) Hy-Vee's Response to Proposed Ordinance
3) APBA's Response to Proposed Ordinance

February 2, 2015

Subject: SINGLE-USE PLASTIC BAG ORDINANCE

To Leah Christian

Leah,

MFA Oil is opposed to this ordinance. These types of ordinances, while seemingly small, generally cause a ripple effect that grows as more and more are added making it difficult for businesses to succeed. There are several reasons that we oppose this ordinance.

This ordinance forces us as a retailer to charge a fee to each person that makes a purchase and needs a bag. This fee is in reality a tax on our customer and the ordinance does not spell out exactly how the fee is to be used. If we are forced to meet the standard of a recyclable paper bag, why will there be a fee associated with the use of that bag? In reality there is a very small percentage of customers that take bags from convenience stores when making a purchase. Many of the purchases are small, with a minimum number of items being purchased. It would also be a safe assumption that a large number of the bags taken are used to dispose of waste items, often to keep a vehicle free from trash.

If this ordinance passes we now have to make another change to the operation of our business just for our Columbia locations. We will have to spend time sourcing paper bags, stock these in our grocery warehouse and retrain our employees in Columbia alone to charge someone for a bag. This process will slow down the checkout process, destroying a key component of our business model, speed and convenience.

It is also unfair to single out restaurants as an exception to the rule since many convenience stores offer hot food to our customer. Does that make us exempt from the ordinance at the locations that we prepare food, place in a container and then put in a plastic bag for the consumer to take with them? We also sell bakery items, based on the definition of product bags they meet this criteria, but bakery items are not necessarily mentioned.

The process of determining which consumer does not have to pay for the bag also becomes difficult. Since our locations accept SNAP, we would have to wait until the form of payment was presented before it could be determined whether or not to charge for a bag. Since these purchases are generally larger, at this point in the transaction the

items would have been placed in the bag, and we would have charged the consumer for the bag. We cannot ask for the form of payment prior to beginning the transaction.

The Break Time locations in Columbia do participate in the recycling program, placing the blue receptacles at our locations, both at the dispensers and near the entrance to our stores.

This ordinance would in reality be the third change since Thanksgiving that imposes unnecessary pressure on retailers, especially convenience stores. We have had to change our hiring process because of ban the box. While the intention of this ordinance was admirable, anyone selling lottery should have been exempt since it is a requirement by the state of Missouri that felons cannot sell lottery tickets. We have lost the ability to sell tobacco products to a group of individuals that are legally able to purchase cigarettes everywhere else in the state of Missouri. With the passage of both of these ordinances, we now have to operate our locations in Columbia different than other areas of the state, in essence any change that alters how we do business in Columbia is an added burden and generally added costs because we have to create new policies, procedures and training mechanisms for a portion of our locations. Adding the plastic bag ordinance to these two, in addition to requirements for all employees to pass the SMART exam to acquire an alcohol servers permit in order to just sell packaged alcohol, creates an onerous burden on our employees.

Thank you

Curtis Chaney
Sr. Vice President of Retail
MFA Oil
573-876-0313
cchaney@mfaoil.com

Hy-Vee's Response to 2009 Iowa Effort to Ban Plastic Bag

This document was developed in response to communities considering bans on plastic bags.

Many grocery retailers have combatted plastic bag litter (which we believe to be at the heart of the issue when it comes to plastic bags) with programs including the goals outlined below. The Iowa Grocery Industry Association's Build With Bags was developed with these goals in mind (www.itseasytorecycle.com).

REDUCE - Reduce the use plastic bags

- Train baggers to reduce bag usage and eliminate unnecessary double bagging
- Promote us of better quality plastic bags that increase items per bag counts
- Work with customers to identify ways reducing bag use and litter

REUSE - Promote use of reusable bags

- Market and promote increased use of reusable bags
- Offer an assortment of reusable bags
- Implement practices that encourage the use of reusable bags

RECYCLE - Increase recycling of plastic bags

- Offer plastic bag recycling at readily available and easily identified locations
- Promote plastic bag recycling with signage and other communication tools
- Publish information about bag recycling rates

Factors that should play a key role in deliberations over a plastic bag bans include:

Plastic Bag Ban Impacts

The impacts of plastic bag bans are often-times overlooked. Some of these impacts include:

- In small towns bans can cause customers to travel to other towns to shop. The potential financial impact of this can be great – particularly to small towns in close proximity to other towns that do not have bag bans.
- In normal economic conditions, towns may be able to generate revenues by choosing to collect plastic bags, as opposed to banning them.
- Plastic bags from other jurisdictions will continue to cause litter and to enter the town's waste stream from purchases made in other communities.
- While bans may limit the number of plastic shopping bags, they will only force people to purchase other plastic bags (for such things as waste disposal or picking up dog 'litter'). Thus bans only force people to buy plastic bags for these other purchases. Being mindful of economic pressures, this only adds to citizens economic burdens.
- Bans shift bag consumption to paper bags, which have a much larger environmental and carbon footprint than plastic bags.

- In Ireland, where a shopping bag tax was levied, the sales of plastic shopping bags decreased by 90 percent, although the sales of other plastic bags, such as garbage bags, increased by 400 percent!
- There have also been several unintended consequences of the Irish Shopping Bag Tax. These include the loss of shopping baskets and carts and an increase in the instances of shoppers filling their carts and walking out of the store without paying.
- Bans, which aim to control litter, do little to do so. Litter control is better addressed by targeting littering and providing recycling and trash receptacles.
- Plastic bag bans negatively impact the plastic bag recycling industry
- Banning plastic bags will not reduce dependence on foreign oil because 80 percent of plastic bags are made from domestic natural gas (polyethylene is made from ethylene which is made from ethane, which is commonly extracted from natural gas) and the energy used to make plastic bags is embodied in the bag itself and is thus available for new products.
- After considering all the implications of bans, many communities have now chosen to focus their efforts on promoting plastic bag recycling.
- Additional information on the impacts of bag bans may be found in the ULS Report entitled, "A Qualitative Study of Grocery Bag Use in San Francisco"

Plastic Bags Facts

Often-times the facts about plastic bags are clouded in misinformation. Some facts about plastic bags are provided below:

- The environmental footprint of plastic bags is arguably less than that of paper. Reported statistics for plastic bags are as follows:
  - 70 percent less energy to produce than a paper bag and 50 percent less greenhouse gas emissions
  - 80 percent less waste and 90 percent less energy to recycle
  - 6 percent of the water needed to make paper bags
  - Considerably less energy to ship based on paper's greater weight and volume (2000 plastic bags weigh 30 pounds, whereas 2000 paper bags weigh 280 pounds). As an example, for every seven trucks needed to deliver paper bags – only one truck is needed for the same number of plastic bags
- For more information on plastic bags, see the "FMI Backgrounder: Plastic Grocery Bags – Challenges and Opportunities" or go to www.americanchemistry.com.
- Statistics indicate that 90 percent of today's consumers reuse plastic bags as liners for household waste baskets, lunch totes, and laundry or garment bags.
- Many argue about the biodegradability of plastic bags in landfills, while in actuality very little biodegradation takes place in landfills based on their design.
- Plastic bags are used by customers for a variety of reasons, including trash and animal refuse. If banned, the replacement for these bags is likely to be commercially available plastic bags which are both more expensive and have a higher mill rate (and thus degrade even less in landfills).

Plastic Bag Recycling

A number of organizations actively promote plastic bag recycling, with the Progressive Bag Affiliates (a division of the American Chemistry Council) now sponsoring a well developed and comprehensive bag recycling program which provides free education, marketing and support materials to help retailers recycle bags.

- There is a growing market for recycled plastic, with companies now being able to earn upwards of $.15 to $.20 per pound for recycled plastic bags.
- Recycled plastic resins are now less expensive than virgin resins
- Recycled plastic bag resins are now used to make: new plastic shopping bags, shopping carts and baskets, decking boards, pallets, patio pavers, laundry baskets, parking lot speed bumps, car stops, and many other products.
- Through the "The Full Circle Recycling Initiative," plastic bag manufacturers have set a goal of 40 percent recycled content (25 percent post-consumer) by the year 2015.
- For more information on plastic bag recycling, go to: www.plasticfilmrecycling.org

Reusable Bags

Reusable bags are a good alternative to plastic bags, although there are some issues related to their use. These trade-offs include:

- Many reusable bags are made of polyethylene imported from China
- The environmental footprint of these bags has not been proven through life cycle analysis to be better than the typical plastic bag (how many t-shirt bags does it take to make one pressed polyethylene reusable bag?)
- Reusable bags can present cross-contamination/food safety issues that do not exist for single-use bags

Degradable Bags

There are several types of "degradable bags", including biodegradable, photo degradable, and other forms of degradability depending on manufacturer claims. One should exercise caution when it comes to claims of degradability, particularly claims of "biodegradability" that are not backed up by sound scientific evidence.

- Degradable bags do nothing to address the litter problem – which is at the core of the plastic bag problem.
- Some degradable bags contaminate the plastic recycling stream
- Degradable bags can come at a cost premium and performance penalty
- Generally speaking, degradable bags do not biodegrade in landfills
- In order to degrade, biodegradable bags typically need to be composted in commercial compost facilities

Rebuttal to "Report on Plastic Bag Restrictions" by the City of Columbia & County of Boone's Environment & Energy Commission

February 9, 2015

WHEREAS, the City of Columbia seeks to protect our waterways, our local streams including Hinkson Creek, the Missouri River, and ultimately our oceans; and

Plastic bags are not a major source of ocean litter.

- Ocean Conservancy sponsors beach cleanup days throughout the U.S. each year. For the first time, plastic grocery bags were tallied separately in 2013. Based on data from 2,609 U.S. sites surveyed in 44 states, **plastic grocery bags comprised 2.1% of all U.S. beach litter**. **For 25 of the 44 states, plastic grocery bags comprised 1.9% or less**, including California (1.7 percent), Oregon (1.4 percent) and Washington (0.9 percent).[1]

- In 2014, scientists from the U.S. Proceedings of the National Academy of Sciences reported that the **overall amount of plastic in our oceans is "far less than expected."** [2]

- Marine experts at the 2011 International Marine Debris Conference stated that the **most pressing concerns for oceans are derelict fishing gear and general solid waste**—not specific products such as plastic bags.[3]

- According to an Oregon State University study, "if we were to filter the surface area of the ocean equivalent to a football field in waters having the highest concentration [of plastic] ever recorded…**the amount of plastic recovered would not even extend to the 1-inch line."**[4]

WHEREAS, single-use plastic carryout bags pollute our local streams and rivers; these bags ultimately break down into smaller bits that contaminate soil and waterways and enter into the food supply that humans, animals, and aquatic life ingest; and

Plastic bags are safe and non-toxic.

- Retail **plastic bags do not contain Phthalates, Bisphenol A (BPA) or other products often referred to as endocrine disruptors**, which are cited as such environmental contaminates.[5]

    - The American Progressive Bag Alliance (APBA) is committed to providing retailers and shoppers with safe, non-toxic plastic bags. The bags manufactured by APBA companies are made from **polyethylene** and **calcium** only – potentially with color added. Ink and color additives used by APBA companies contain **no heavy metals**, and all suppliers have submitted letters confirming their compliance with federal safety standards.[6]

- According to National Oceanic and Atmospheric Administration (NOAA), "To date **there are no published studies specifically researching how many marine mammals die each year directly due to marine debris**" but the **closest figure available "does not state marine**

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[1] ER Planning PR Bags Report Brief – 2014 Litter Survey Rankings (report available upon request)
[2] PNAS: Plastic debris in the open ocean; Andrés Cózar; 2014
[3] Technical Proceedings of the Fifth International Marine Debris Conference 2011
[4] Oregon State University: Oceanic "garbage patch" not nearly as big as portrayed in media
[5] Polyethylene Resins Product Declaration available upon request
[6] Polyethylene Resins Product Declaration available upon request

mammals are dying from plastic pieces, but rather that mortality is caused by entanglement from lost fishing gear and other unknown causes." [7]

- Regarding Surfrider's claims that plastics are responsible for the deaths of 1.5 million marine animals, Senior Staff Scientist Rick Wilson said: "I will admit **it's difficult to track down a definitive scientific study** source for it."[8]

- Dr. Chris Reddy—a top cited and published marine scientist said **a plastic bag ban may actually increase damage to marine life** since alternatives to plastic bags contribute greatly to "oceanic dead zones" caused by nitrogen in the water.[9]

WHEREAS, the City Council of the City of Columbia seeks to reduce the toxicity of waste materials in the solid waste stream that are directed to resource recovery and sanitary landfill facilities, and to maximize the removal of plastic carryout bags from the waste stream; and

Bag bans and taxes don't reduce waste or litter and don't save cities money.

- **Plastic bags make up just 0.4% of the U.S. municipal solid waste stream**, so banning or taxing them won't effectively reduce the amount of solid waste sent to the landfill.[10]

- In nearby Iowa, a statewide waste characterization study found plastic retail bags make up **only 0.3% of all waste** in the state.[11] (Note: identified Missouri waste studies do not detail the percentage of plastic bags among their waste stream data.)

- Plastic bags take up less space in a landfill than paper or reusable bags. Besides, a standard "reusable bag" is a **petroleum-based** product made from nonwoven polypropylene (NWPP) and is **not recyclable.**[12] Yet, we **import half-a-billion each year** from countries such as China and Vietnam.[13] As a result, **95.5% of NWPP bags are sent to landfills**[14] **after only about 15 uses**;[15] thus, the environmental community is worried about **surplus sacks adding up in our landfills.**[16]

- A 2013 study examined budgets for litter collection and waste disposal in cities that banned plastic grocery bags and found **"no evidence of a reduction in costs attributable to reduced use of plastic bags"** in San Francisco, San Jose, and the City and County of Los Angeles, CA; Washington, D.C.; and Brownsville and Austin, TX.[17]

- Denver, Colorado's Chief of Sustainability called p**lastic bag bans and taxes "misguided" if the policy's goal is to make a substantial dent in waste.**[18]

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[7] NOAA: What we actually know about common marine debris factoids
[8] San Francisco Chronicle: Garbage-patch tale as flimsy as a single-use plastic bag; Debra J. Saunders; July 2014
[9] Kirkland Reporter: The science and comedy of Kirkland's proposed bag ban; Jan. 22, 2015
[10] EPA Municipal Solid Waste Report: 2011
[11] Mid Atlantic Solid Waste Consultants, "2011 Iowa Statewide Waste Characterization Study;" September 2011.
[12] Gathered from tariff and trade data from the U.S. Department of Commerce and the U.S. International Trade Commission.
[13] Gathered from tariff and trade data from the U.S. Department of Commerce and the U.S. International Trade Commission.
[14] Joseph Greene, "Survey of Reusable and Single-use Grocery Bags in Northern California," California State University, Chico Research Foundation, August 31, 2010.
[15] Edelman Berland: "Reusable Bag Study;" May 2014.
[16] The Dec. 4, 2011 article in New York Magazine, "The Inconvenient Truth of Reusable Grocery Bags," also noted that 12% of the bags carry E. coli bacteria.
[17] NCPA: Do Bans on Plastic Grocery Bags Save Cities Money?, December 2013.
[18] Brief: Denver, CO Chief of Sustainability

WHEREAS, the City of Columbia signed Resolution 160-06A, endorsing the Mayors Climate Protection Agreement striving to meet or exceed Kyoto Protocol targets for reducing global warming pollution by taking actions in our community; and

Plastic bags are the most environmentally friendly option at the checkout.

- 100% recyclable, reusable and American-made **plastic retail bags are produced using high-density polyethylene, a byproduct of U.S. natural gas, not foreign oil**.[19] A standard "reusable bag" is a **petroleum-based** product and is **not recyclable.**[20]

- Plastic bags are more resource efficient and generate **fewer greenhouse gas emissions** than alternatives;[21] Reusable bags make a **greater contribution to global warming** than plastic bags.[22]

- A study by the UK government found a standard **cotton grocery bag must be used *131 times*** before its contribution to global climate change is lower than that of a **plastic bag used only once**.[23]

- In regard to choosing grocery bags, Dr. David Tyler of the University of Oregon said: "If the most important environmental impact you wanted **to alleviate was global warming, then you would go with plastic.**"[24]

- A standard **paper bag must be reused *3 times* before its contribution to global climate change is lower than that of a plastic bag** used only once.[25]

WHEREAS, the City of Columbia has shown an interest in being a leader in initiatives with a positive effect on climate change through its curbside recycling program, methane gas bioenergy facility, solar energy arrays, LEED certified city buildings, increased use of renewable energy; and

Plastic bags are 100% recyclable.

- The **plastic bag manufacturing and recycling industry has invested more than $1.1 million in a public education program** called "A Bag's Life," which **promotes reuse and recycling of plastic bags.**

- Through its Bag-2-Bag program, **APBA member NOVOLEX has been working with grocery stores and retailers across the U.S. to establish roughly 30,000 plastic bag recycling points** over the past four years.

    - **Consumers can bring their 100% recyclable plastic bags and wraps to participating stores** and drop them into plastic bag recycling bins. From there, the bags and wraps are

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[19] [19] Analysis by Chemical Market Associates, Inc.; February, 2011.
[20] Gathered from tariff and trade data from the U.S. Department of Commerce and the U.S. International Trade Commission.
[21] This figure was calculated by Boustead Consulting & Associates in their 2007 study entitled, "Life Cycle Assessment for Three Types of Grocery Bags—Recyclable Plastic; Compostable, Biodegradable Plastic; and Recycled, Recyclable Paper." The resource allocation for different types of bags can be found on page 4 and shows that polyethylene grocery bags use less oil, and less potable water. In addition, polyethylene plastic grocery bags emit fewer global warming gases, less acid rain emissions, and less solid wastes.
[22] The lifecycle impacts of each carrier bag on global warming potential can be found on page 33.
[23] U.K. Environmental Agency. "Life Cycle Assessment of Supermarket Carrier Bags." February 2011.
[24] "Paper or Plastic? The answer might surprise you;" University of Oregon, Cascade Magazine; Fall 2012
[25] U.K. Environmental Agency. "Life Cycle Assessment of Supermarket Carrier Bags." February 2011.

picked up for recycling. **Plastic bags are recycled into eco-friendly material** for playgrounds, construction equipment and new plastic bags.

- **Trex Co. recycles about 1.3 billion plastic retail bags each year** to produce splinter-free, mold-resistant decking material that doesn't need staining or painting. Trex products were **used to rebuild boardwalks at four East Coast beaches damaged as a result of Hurricane Sandy.**[26]

- **More than 90% of the U.S. population has access to plastic bag recycling**;[27] according to EPA data, 14.7% of polyethylene bags, sacks and wraps made in the U.S. are recycled.[28]

- Ordinances to ban or tax plastic retail bags hurt our country's recycling infrastructure and positive innovations such as retailer take-back programs.

- "Rather than bans and fees that take away jobs and increase costs to consumers, policy makers should take advantage of the great economic and environmental opportunities associated with responsibly recycling these bags." — Institute of Scrap Recycling Industries (ISRI)[29]

WHEREAS, the production of plastic bags worldwide requires the use of more than 12 million barrels of oil per year, which also has a significant environmental impact; and

Plastic bags are made from natural gas, not oil.

- 100% recyclable, reusable and American-made **plastic retail bags are produced using high-density polyethylene, a byproduct of U.S. natural gas, not foreign oil**.[30] **A standard "reusable bag" is a petroleum-based product** and is **not recyclable.**[31]

WHEREAS, many chemicals in plastic products are now known to cause harm, but the chemicals present in plastic bags are not disclosed to the public and are not required to be tested for health effects, so chemicals in plastic bags cannot be assumed to be safe.

Plastic bags are safe and non-toxic.

- The American Progressive Bag Alliance (APBA) is committed to providing retailers and shoppers with safe, non-toxic plastic bags. The bags manufactured by APBA companies are made from **polyethylene** and **calcium** only – potentially with color added. Ink and color additives used by APBA companies contain **no heavy metals**, and all suppliers have submitted letters confirming their compliance with federal safety standards.[32]

- Full disclosure is included in the "Composition and Compliance" materials of the Appendix.

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[26] Green Builder: "Deck Durability," 4/30/2013
[27] Moore Recycling Associates, "Plastic Film and Bag Recycling Collection: National Reach Study," 2012
[28] EPA Municipal Solid Waste Report: 2011
[29] ISRI statement on bag bans and taxes
[30] [30] Analysis by Chemical Market Associates, Inc.; February, 2011.
[31] Gathered from tariff and trade data from the U.S. Department of Commerce and the U.S. International Trade Commission.
[32] Polyethylene Resins Product Declaration available upon request

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Composition and Compliance

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PLASTIC BAG RECYCLING: A BETTER SOLUTION FOR COLUMBIA, MO

Communities across the U.S. are looking at ways to help the environment and at the same time support their local economy. Bans and taxes on 100% recyclable, America-made plastic bags—while well intentioned—actually weigh down the economy, increase costs and inconvenience consumers.

By targeting a single product, bans and taxes do not address the greater litter issue and will instead steer consumers to alternatives that are worse for the environment. Recycling is a common-sense policy that's good for the economy, the environment and consumers, and it's one that provides a positive direction for everyone.

Here are the facts…

Plastic bags are the most environmentally friendly option at the checkout…

- Plastic bags are **100% recyclable, reusable** and made from a byproduct of **natural gas, not oil**.[i]
- Plastic bags are more resource efficient, take up less landfill space and generate **fewer greenhouse gas emissions** than alternatives.[ii]
- A study by the UK government found a standard **cotton grocery bag must be used *131 times*** before its contribution to global climate change is lower than that of a **plastic bag used only once**.[iii]

So-called "reusable" bags are not the eco-friendly solution…

- Reusable bags make a **greater contribution to global warming** than plastic bags.[iv]
- A standard "reusable" bag is a **petroleum-based** product made from nonwoven polypropylene (NWPP).
- NWPP bags are **not recyclable**[v] and yet, we **import half-a-billion** each year from countries such as China.[vi]
- As a result, **95.5% of NWPP bags are sent to landfills**[vii] **after only about 15 uses**[viii]; and now the environmental community is worried about surplus sacks adding up in our landfills.[ix]

Bag bans and taxes don't reduce waste or litter and don't save cities money…

- Plastic bags typically make up less than **1% of roadside litter**,[x] and only a **tiny fraction (0.4%)** of the U.S. municipal solid waste stream, so a ban or tax would have very little effect on litter and waste overall.[xi]
- In nearby Iowa, a statewide waste characterization study found plastic retail bags make up **only 0.3% of all waste** in the state.[xii] (Note: identified Missouri waste studies do not detail the percentage of plastic bags among their waste stream data.)
- A 2013 study examined budgets for litter collection and waste disposal in cities that banned plastic grocery bags and found **"no evidence of a reduction in costs attributable to reduced use of plastic bags"** in San Francisco, San Jose, and the City and County of Los Angeles, CA; Washington, D.C.; and Brownsville and Austin, TX.[xiii]

Bans and taxes burden local businesses…

- Bans and taxes on plastic bags force local businesses to comply with **additional government regulations** that mandate measuring, counting, and maintaining records, or face fines and penalties.[xiv]
- Retailers across the U.S. are reporting an increase in **shoplifting after their cities banned plastic bags**.[xv]
- A recent study found bans on plastic bags negatively impact retail sales and employment inside the ban area by **shifting business just outside the bag ban region**.[xvi]

Bans and taxes burden consumers…

- Most people **(82%) believe the government shouldn't decide** what types of bags are OK to use.[xvii]
- Reusable **bag owners forget their reusable bags** on nearly half of their grocery trips;[xviii] those people would be forced to pay the tax for a paper bag under this proposed ordinance each time they forget their bags.
- Food prices have risen each year, and are expected to jump **another 3% in 2015**,[xix] making **an extra tax** on grocery bills particularly hurtful to a lot of families struggling to make ends meet.

"Reusable" bags are not a healthy alternative…

- A 2011 study found **bacteria in 99% of reusable bags** tested, coliform bacteria in over 50% of the bags tested, and 8% contained E. coli; plus, **97% of people surveyed never wash their reusable bags**.[xx]
- Another study found bacteria build-up on reusable bags to be **300% higher than what is considered safe**,[xxi] and storing these bags in a hot trunk causes the **bacteria to grow 10 times faster**.[xxii]

Recycling is the better alternative to bans and taxes.

- Plastic bags are **recycled into new items** such as backyard decking, park benches and playground and construction equipment.
- Leading plastic bag recycler NOVOLEX has worked with retailers to establish roughly **30,000 plastic bag recycling drop-off points across the U.S.** over the past four years.
- In 2013 alone, NOVOLEX's "Bag-2-Bag" recycling program **processed more than 35 million pounds** of post-consumer plastic bags, sacks and wraps.
- In nearby Iowa, responsible recycling is promoted through the successful "**Build with Bags Grant Program**"—a cooperative effort among the Iowa Grocery Industry Association, Keep Iowa Beautiful, the Iowa Department of Natural Resources and *The Des Moines Register*, among others—that provides incentive for communities to develop their plastic bag recycling efforts and funds for parks and schools to purchase playground equipment made from recycled plastic bags. As their website states, this program "provides an effective alternative to plastic bag bans and the unintended consequences that often result from bans and other related regulatory efforts."[xxiii]

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[i] An analysis by Chemical Market Associates, Inc. in February 2011, debunked several common myths about plastic bags.
[ii] This figure was calculated by Boustead Consulting & Associates in their 2007 study entitled, "Life Cycle Assessment for Three Types of Grocery Bags—Recyclable Plastic; Compostable, Biodegradable Plastic; and Recycled, Recyclable Paper." The resource allocation for different types of bags can be found on page 4 and shows that polyethylene grocery bags use less oil, and less potable water. In addition, polyethylene plastic grocery bags emit fewer global warming gases, less acid rain emissions, and less solid wastes.
[iii] U.K. Environmental Agency. "Life Cycle Assessment of Supermarket Carrier Bags." February 2011.
[iv] The lifecycle impacts of each carrier bag on global warming potential can be found on page 33.
[v] Gathered from tariff and trade data from the U.S. Department of Commerce and the U.S. International Trade Commission.
[vi] Gathered from tariff and trade data from the U.S. Department of Commerce and the U.S. International Trade Commission.
[vii] Joseph Greene, "Survey of Reusable and Single-use Grocery Bags in Northern California," California State University, Chico Research Foundation, August 31, 2010.
[viii] Edelman Berland: "Reusable Bag Study;" May 2014.
[ix] The Dec. 4, 2011 article in New York Magazine, "The Inconvenient Truth of Reusable Grocery Bags," also noted that 12% of the bags carry E. coli bacteria.
[x] The number was derived by Environmental Resources Planning LLC in their 2012 ER Planning Report Brief: Plastic Retail Bags in Litter, which randomly surveyed landfills across the U.S. and Canada from 1994-2012.
[xi] EPA Municipal Solid Waste Report: 2011; U.S. Environmental Protection Agency
[xii] Mid Atlantic Solid Waste Consultants, "2011 Iowa Statewide Waste Characterization Study;" September 2011.
[xiii] NCPA: Do Bans on Plastic Grocery Bags Save Cities Money?, December 2013.
[xiv] San Francisco Environment Code: Ch. 17 – Plastic Bag Reduction Ordinance, LA County Department of Public Works: About the Bag Ban FAQ, City of Portland City Code and Charter: Ordinance No. 18573, and District of Columbia Official Code Title 8 Subtitle A Chaper 1 Subchapter 1-A
[xv] Washington City Paper: Safeway: Bag Tax Causes Theft!, SeattlePI: Store Owners Say Plastic Bag Ban Causes More Shoplifting, and San Leandro Times: Plastic Bag Ban Spurs Shoplifting
[xvi] NCPA: A Survey on the Economic Effects of Los Angeles County's Plastic Bag Ban
[xvii] Reason-Rupe May 2013 Public Opinion Survey
[xviii] Edelman Berland: "Reusable Bag Study;" May 2014.
[xix] "Changes in food price indexes, 2010 through 2013;" USDA; January 2013.
[xx] "Assessment of the Potential for Cross Contamination of Food Products by Reusable Shopping Bags;" Charles Gerba; University of Arizona; August 2011.
[xxi] "A Microbiological Study of Reusable Bags and 'First or single-use' Plastic Bags," Environment and Plastics Industry Council; May 20, 2009.
[xxii] "Assessment of the Potential for Cross Contamination of Food Products by Reusable Shopping Bags," Charles Gerba; University of Arizona; August 2011.
[xxiii] Iowa Grocery Industry Association, "Build with Bags Program Overview;" 2014.

bag the ban.

the truth About Plastic Bags

Despite the facts, plastic bags have gotten a bad rap. Here's the truth about your plastic bags and why they are the right choice for the environment, your wallet and your community.

Litter and Composition

Plastic shopping bags made in the United States are made from natural gas.

0.5%

EPA data shows that plastic shopping bags make up only 0.5 percent of the U.S. municipal waste stream.

RECYCLE PLASTIC BAGS

Plastic bags are 100% reusable and recyclable. Recycle your plastic bags in the bins outside your local grocery store.

Thus, ban plastic bags won't reduce overall litter.

the truth... About Plastic Bags

Reusing and Recycling

100%

Plastic bags are 100 percent recyclable and reusable.

Nine Of TEN

Plastic bags are not "single-use". More than 9 out of 10 of Americans reuse their plastic bags at least once, for everything from storage to waste disposal to packing material.

1 BILLION

In 2011, an estimated 1 billion pounds of post-consumer plastic bags and films were collected for recycling in the United States. The same report showed that plastic bag and film recovery has increased by 55 percent since 2005.

30,000

There are over 30,000 locations for plastic bag recycling across the country!

Recycled plastic bags are used to make new plastic bags and products, such as backyard decks, playground equipment, plumbing pipes and fencing.

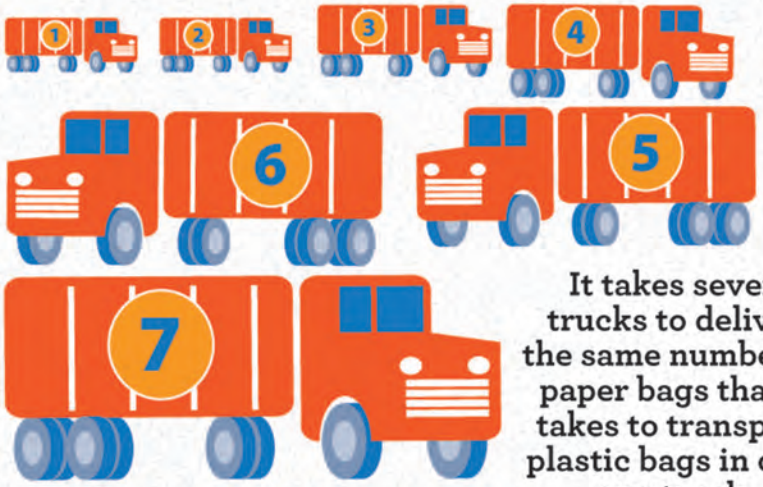
91-93%

91 to 93 percent of the U.S. population has access to nearby plastic bag recycling.

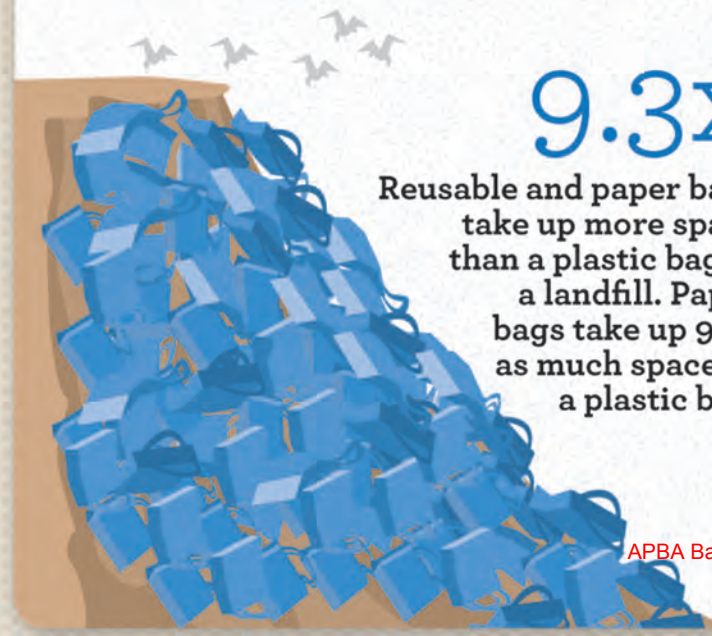
the truth... About Plastic Bags

The Real Green Bag

Plastic grocery bags require 70% less energy to manufacture than paper bags. The production of plastic bags consumes less than 4% of the water needed to make paper bags.



It takes seven trucks to deliver the same number of paper bags that it takes to transport plastic bags in only one truck.



9.3x

Reusable and paper bags take up more space than a plastic bag in a landfill. Paper bags take up 9.3x as much space as a plastic bag.

Plastic bags produce FEWER greenhouse gasses per use than paper bags or cotton bags.

the truth... About Plastic Bags

The Not So Green Reusable Bags

131x

A reusable cotton bag must be used no less than 131 times before having a "greener" environmental impact than a common plastic grocery bag!

MORE

Many reusable bags are made from heavier and thicker plastic or cotton which takes up more energy to produce. Reusable bags aren't recyclable.

The U.S. ITC reports that hundreds of millions of reusable bags are imported to the United States each year. It takes more energy to transport a reusable bag from overseas than it does to manufacture and transport an American-made plastic bag.

TRASHED

Reusable bag giveaways are environmentally costly when unwanted bags end up inthe dumpster, often after only one use.

the truth... About Plastic Bags

Bag Bans Hurt Businesses

-25%

Some stores have seen declines in business. One Solana Beach business saw a 25 percent decline in business following the implementation of the ban.

-$10,000

A Grocery Outlet Store told a Portland newspaper that they have lost over $10,000 to shoplifters using reusable bags.

300

One store found that 300 plastic grocery baskets have been stolen by customers since the bag ban was announced.

40%

Following Seattle's ban on plastic bags, nearly 40 percent of storeowners surveyed post-ban reported seeing their costs for carryout bags increase between 40 and 200 percent.

Sources:

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http://www.rannv.org/documents/8/April%202008.pdf

Boustead Consulting & Associates: "Life Cycle Assessment for Three Types of Grocery Bags—Recyclable Plastic; Compostable, Biodegradable Plastic; and Recycled, Recyclable Paper," 2007.

Tariff and trade data from the U.S. Department of Commerce and the U.S. International Trade Commission.

Plastic Retail Bags & Recycling

Presentation to City of Columbia, MO Energy & Environment Commission

January 27, 2015

Overview

- About the Plastic Retail Bag Industry
- About Plastic Retail Bags
- About Reuse and Recycling
- About “Reusable” Bags
- Facts and Avoiding Misguided Policies
- Economic Consequences of Bag Regulation
- Environmental Consequences of Bag Regulation
- Viable Alternatives to Bag Legislation
- Conclusion

About the Plastic Retail Bag Industry

380 Facilities

30,900 Jobs in U.S.
500 in Missouri

$1.3 Billion in
Annual Payroll

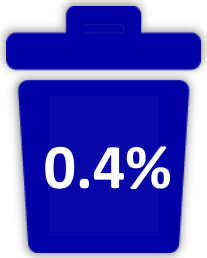
$268 Million in
Capital Expenditures

Provides **Manufacturing and Recycling Jobs**

Millions Invested in
Recycling Education

About Plastic Retail Bags

Most plastic retail bags are made from a **by-product of natural gas.**[1]



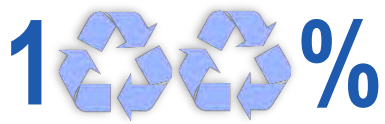
EPA data shows that plastic shopping bags make up just **0.4% of the U.S. municipal waste stream.**[2]

Plastic retail bags are 100% recyclable and can be recycled through retail take-back bins. [3]

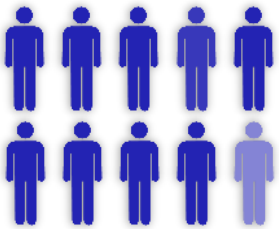
The Better Alternative: Plastic Retail Bags are

- ✓ **More resource-efficient**
- ✓ **Take up less landfill space**
- ✓ **Generate fewer greenhouse gas emissions**

About Reuse and Recycling



100% Recyclable and Reusable [4]



9 out of 10 People Reuse Plastic Retail Bags [5]

NOVOLEX has issued **30,000** Plastic Retail Bag Recycling Bins Across U.S. [6]

More than 90% of People Have Access to Plastic Retail Bag Recycling [7]

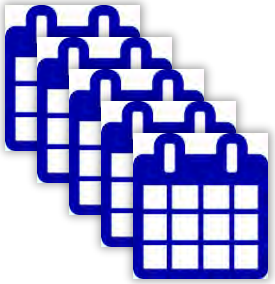
Recycled Plastic Retail Bags Are Used to Create **New Products** [8]

More than **1 Billion lbs. Collected** Annually [9]

About "Reusable" Bags

Standard "reusable" grocery bags are

- **Made from oil**
- **Non-recyclable**
- **500 million imported annually into the U.S. from overseas, primarily China**



We support consumer choice, but certain facts about alternatives are undeniable:

- You would have to use the same cotton bag for **5 years** before it is a better option for the environment than a plastic retail bag *used twice*. [10]

- Reusable bags **spread bacteria and disease** because they are rarely washed. In one study, coliform bacteria were found in 51% of the bags tested, with generic *E. coli* in 8%. [11]

Facts and Avoiding Misguided Policies

"Garbage-patch tale as flimsy as a single-use plastic bag,"
San Francisco Chronicle, July 2, 2014

Economic Consequences of Bag Regulation

Plastic Retail Bag Bans and Taxes:

- **Threaten thousands** of U.S. manufacturing and recycling jobs
- Incentivize consumers to **shop outside of the ban region**—a particular issue for shops located near border regions
- Increase **shop-lifting** [12]

- Create **bureaucratic red tape** for small business owners and more **administrative challenges**
- Present storage **challenges for small store owners** with limited space for bulkier bagging options
- **Impose a regressive tax** on low-income families.
- Reveal **no evidence of a reduction in costs** attributable to reduced use of plastic retail bags [13]

Environmental Consequences of Bag Regulation

Plastic Retail Bag Bans and Taxes:

- Force customers to alternatives that generate **more greenhouse gases**
- Can introduce **more plastic into landfills** after reusable bags and thicker plastic retail bags are discarded
- **Do not reduce litter or waste**—or the cost of litter and waste collection
- **Waste more water**, which is needed to manufacture alternatives to plastic retail bags
- **Distract local officials** from tackling serious environmental problems

Evaluating the Facts

"It's very unlikely that many animals are killed by plastic bags. The evidence show just the opposite. We are not going to solve the problem of waste by focusing on plastic bags… With larger mammals it's fishing gear that's the big problem. On a global basis plastic bags aren't an issue."

- David Santillo, marine biologist Greenpeace [14]

Evaluating the Facts

"Concluding that a bag fee will make a substantial dent in waste going to landfills is misguided."

- Chief of Sustainability, Denver, Colorado [15]

"There are much greater threats to the ocean than plastic bags and a bag ban in Rhode Island would not be meaningful."

- Woods Hole Oceanographic Institute [16]

"…this problem is solved not by bans on products but by proper disposal, recycling and education."

- Timothy Wright, Retired Captain, National Oceanic and Atmospheric Administration [17]

"Reusable shopping bags have proliferated so greatly that eco-activists are worried about surplus sacks winding up in landfills."

- New York Magazine [18]

Avoiding Misguided Policies

- **January 2015** – **Huntington Beach, CA,** City Council voted to repeal with their bag ban / fee ordinance
- **December 2014** – More than 800,000 citizens in **California** signed petitions to place a statewide bag ban law (approved by the Governor in Sept. 2014) on hold and the measure will now go to a public referendum in November 2016.
- **December 2014** – The mayor of **Baltimore, MD,** vetoed an ordinance
- **November 2014** – Citizens of **Mercer County, NJ,** reject referendum to place 5-cent fee on bags
- **October 2014** – **Fort Collins, CO,** voted 6-1 in favor of repealing a fee on bags
- **August 2014** – Texas Attorney General Greg Abbott issued a 5-page opinion article concluding that municipal plastic retail bag relegation is illegal under **Texas** state law
- **April 2014** – Local policymakers in **Johnson County, IA,** decided to pursue recycling strategies in lieu of ban
- **November 2013**– **Durango, CO,** voters overturn ban on plastic bags
- **March 2013** – **Eau Claire, WI,** adopts plan for sustainable bag use

Viable Alternatives to Bag Legislation

Nearly 90% of the country recycles plastic retail bags through initiatives such as **"A Bags Life"** which aims to:

- **Reduce** – Encourage reuse of plastic retail bags at the grocery store
- **Recycle** – Collect plastic retail bags from consumers after use
- **Reuse** – Promote reuse of plastic retail bags in a variety of ways such as a trash can liner, lunch bag and for their pets

www.ABagsLife.com

Bag-2-Bag® Recycling Program

Education Works

- With help, we can implement educational programs such as:
  - Retailer take-back programs and drop-off sites
  - School system initiatives (e.g., collection contests)
  - Online marketing and consumer education
  - Advertising campaigns

The Result

Smart Sustainability	Consumer Education	Increased Recycled Content
Litter Prevention	Landfill Diversion	Robust Recycling Infrastructure

Thank you.

Questions?

References

1 Analysis by Chemical Market Associates, Inc.; February, 2011
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4 Analysis by Chemical Market Associates, Inc.; February, 2011
5 Reason-Rupe May 2013 Public Opinion Survey
6 Bag the Ban website
7 Bag the Ban website, "NYC Myth vs Facts"
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10 U.K. Environmental Agency. "Life Cycle Assessment of Supermarket Carrier Bags." February 2011
11 International Association for Food Protection, "Assessment of the Potential for Cross-contamination of Food Products by Reusable Shopping Bags," 2011
12 Washington City Paper: Safeway: Bag Tax Causes Theft!, Seattle PI: Store Owners Say Plastic Bag Ban Causes More Shoplifting, and San Leandro Times: Plastic Bag Ban Spurs Shoplifting
13 NCPA: Do Bans on Plastic Grocery Bags Save Cities Money?, Dec. 2013
14 Las Cruces Sun-News, Julian Morris: Bans on plastic shopping bags don't help environment; Oct. 5, 2014
15 Denver, CO Office of Sustainability, September 2009
16 Testimony to Rhode Island Senate's Committee on Environment and Agriculture
17 *Providence Journal*, Timothy B. Wright: The oceans face much bigger problems than plastic bags; Feb. 2014
18 *New York Magazine*: Noelia De La Cruz, Tote Overload; Dec. 4, 2011

Appendix

NOVOLEX's Environmental Mission

Recycle if Clean & Dry
Store Drop-off
PLASTIC
BAG
how2recycle.info

To be a leading provider of environmentally-preferred, high-quality packaging solutions that create value and provide superior service for our customers by…

1. **Embracing Innovation That Reduces Waste**
   Product quality / source reduction / material innovation

2. **Promoting Recycling**
   Diversion from waste stream / increased use of recycled content

3. **Advocating Waste Reduction**
   Consumer education / supporting recycling programs

The Oil Myth

There is a claim repeated over and over again on the Internet that plastic bags are made out of oil and that 12 million barrels of oil are used annually in the United States to make the plastic bags that Americans use.

It is not true.

About 72.5% of plastic bags used in the United States are made in the United States. Plastic bags are made out of polyethylene. In the United States, ethylene is made of ethane which is a waste by-product obtained from natural gas refining. **Domestically produced plastic bags are not made out of oil.**

The ethane must be removed from the natural gas anyway to lower the BTU value of the natural gas to an acceptable level. Ethane burns too hot to be allowed to remain in high levels in natural gas that is delivered to homes and businesses for fuel. There is nothing else that the ethane can be used for except to make ethylene. If ethane is not used to make plastic, it will have to be burned off, resulting in greenhouse gas emissions.

Using the ethane to make plastic does not in any way reduce the amount of fuel available for transportation or power generation or increase our energy imports.

If we were to abolish plastic bags, it would have zero impact on our dependence on foreign oil.

The United States is an exporter of polyethylene. The United States imports virtually no polyethylene.

ER Planning Report Brief: Plastic Retail Bags in Litter

Environmental Resources Planning, LLC is the only U.S. firm focusing exclusively on litter-related field surveys and research studies. Our firm analyzes select components of the litter stream to better understand the dynamics underlying littering rates. Our staff led the design and project management of Keep America Beautiful's *2009 National Litter Survey*. That study found that plastic bags of all types comprise only 0.6 percent of litter. Percentages for categories such as plastic bags constituted such a minute portion of roadside litter that they were not specifically addressed in the *2009 National Litter Survey*.

National, state and city-wide litter surveys conducted with statistically-based scientific methodologies have established that plastic retail bags continue to comprise a small percentage of litter and the waste stream. Our staff have planned and conducted a number of recent litter surveys. These statistically-based studies were conducted with scientific rigor using trained professionals. Data and methodologies were explained in detail to allow review by interested parties and affected stakeholders.

Litter surveys showing unusually high rates of items such as plastic bags were typically conducted by volunteers rather than professional staff. These surveys tended to lack random sampling and statistical methodologies. At times, material categories were not consistent. While such studies have helped create the awareness of litter's impacts, their limitations have, in some cases, resulted in erroneous depictions of plastic retail bags as a component of the overall litter stream.

Retail Plastic Bags in Recent Litter Surveys

#	Survey	Year	Percent	#	Survey	Year	Percent
1	Toronto	2012	0.8%	11	Durham	2003	0.3%
2	Edmonton	2011	1.1%	12	Peel	2003	0.1%
3	Alberta	2009	0.0%	13	York	2003	0.4%
4	San Francisco	2008	0.6%	14	Toronto	2002	0.6%
5	San Jose	2008	0.4%	15	Florida	2002	0.5%
6	KAB	2008	0.6%	16	Florida	2001	0.7%
7	Alberta	2007	2.0%	17	Florida	1997	0.6%
8	San Francisco	2007	0.6%	18	Florida	1996	1.0%
9	Toronto	2006	0.1%	19	Florida	1995	0.7%
10	Toronto	2004	0.2%	20	Florida	1994	0.6%

As shown in the table above, recent science-based litter surveys using random sampling methodologies consistently found that retail plastic bags comprise a minor portion of litter, usually less than one percent.

Steven R. Stein

Steven R. Stein, Principal
Environmental Resources Planning, LLC

ER PLANNING

© Environmental Resources Planning, LLC

Executive Summary

As shown in the statistically based studies outlined in this report, plastic retail bags (*PR Bags)* comprise a very small portion of litter. National, statewide and citywide litter surveys that characterize litter using statistically based sampling methodologies have established and consistently show that *PR Bags* typically constitute less than 1.0 percent of litter.

Of that small portion, a significant number are unbranded and originate from sources such as small independent restaurants, retailers and convenience stores - the very stores often exempted from bag ordinances. A summary of *PR Bags* found in recent visible litter surveys is provided below:

- Texas: 1.95 percent in 2013 (ERP 2013)[1],
- Toronto (Canada): 0.80 percent in 2012 (ERP 2012),
- Northeast Litter Survey in 2010 (Maine, New Hampshire and Vermont) showed that plastic film, bags and wraps of all types constituted between 2.1 percent and 3.0 percent of all litter (ERP 2010),
- San Francisco: 0.64 percent in 2008 (HDR 2008),
- San Francisco: 0.59 percent in 2007 (HDR 2007),
- Florida: <1.0 percent in five litter surveys conducted 1994-2002 (FL 2002), and
- Nationally, all plastic bags (e.g., trash bags, retail bags, take-out food bags, bulk food bags, sandwich bags, etc.) comprise 0.60 percent of all litter based on Keep America Beautiful's 2009 National Litter Survey (KAB 2009).

With cities and counties focusing on litter issues, this study will help government officials understand the types of *PR Bags* found along city streets.

To accurately determine the types of plastic and paper bags found in litter, ER Planning conducted three separate citywide litter surveys between December 2011 and January 2012 in two California cities (Oakland and San Francisco) and in Washington, D.C. Each of these cities has taken a different approach to managing bag litter.

Field crews physically surveyed 180 sites (60 in each city), covering a total of 6.48 million square feet. In each city, field crews collected data for all types of plastic and paper bags including the source (e.g., convenience store) and brand label on each bag found in litter.

PR Bags from grocery stores, pharmacies, convenience stores and take-out food outlets were each categorized separately. *PR Bags* from all other retail stores such as Dollar Tree and Home Depot were categorized as *Other Retail Bags*.

Table ES-1 below shows the percentage of paper and plastic bags that were unbranded. San Francisco yielded the highest percentage of unbranded bags (82 percent).

---

[1] *Unbranded or generically labeled PR Bags comprised 1.3 percent of this total, while branded bags (with store names) comprised 0.7 percent (ERP 2013).*

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Oakland (66 percent) and Washington, D.C. (67 percent) yielded a similar percentage of unbranded bags. Unbranded bags are used by independent stores and small vendors, but are not typically used by corporate stores, which tend to emboss their logos on the bags they use.

Table ES-1 – Unbranded Bags

City	All Paper Bags	All Plastic Bags	All Paper and All Plastic Bags
Oakland	49%	72%	66%
San Francisco	56%	96%	82%
Washington, D.C.	29%	73%	67%
All Cities	**48%**	**75%**	**68%**

PR Bags from all types of stores (i.e., convenience stores, take-out food establishments, grocery, pharmacy and other retail stores) comprised less than half of all bags and slightly more than half of plastic bags littered in Washington D.C. and Oakland, and much less in San Francisco.

Table ES-2 – *PR Bags* (Branded and Unbranded)

City	#	% of All Plastic Bags	% of All Paper & Plastic Bags
Oakland	149	57%	42%
San Francisco	9	18%	12%
Washington, D.C.	49	52%	44%
All Cities	**207**	**51%**	**38%**

Table ES-3 shows the percentage of *PR Bags* in each city that were unbranded. The highest percentage of unbranded *PR Bags* was observed in San Francisco (78 percent). Approximately half of the *PR Bags* littered in Oakland (50 percent) and Washington D.C. (49 percent) were unbranded.

The high percentage of unbranded *PR Bags* observed in all three cities suggests smaller, independent stores as the likely source. Cities that exempt independent stores from bag ordinances do so at their own peril, since more than half of all *PR Bags* surveyed in these three cities represented bags used by independent stores (unbranded).

Table ES-3 – Unbranded *PR Bags* in Litter

City	Unbranded *PR Bags*	All *PR Bags*	Percent Unbranded
Oakland	75	149	50%
San Francisco	7	9	78%
Washington, D.C.	24	49	49%
All Cities	**106**	**207**	**51%**

Similar to *PR Bags*, almost half of all paper bags observed in the three cities (48 percent) were unbranded as shown in Figure ES-4. This was particularly true in Oakland (49 percent) and San Francisco (56%), while slightly less than one-third of all littered paper bags in D.C. (29 percent) were unbranded.

Table ES-4 – Unbranded Paper Bags in Litter

City	Unbranded Paper Bags	All Paper Bags	Percent Unbranded
Oakland	44	90	49%
San Francisco	15	27	56%
Washington, D.C.	5	17	29%
All Cities	**64**	**134**	**48%**

Key Findings

- Most littered paper and plastic bags were unbranded in San Francisco (82 percent), D.C. (67 percent) and Oakland (66 percent) as shown in Table ES-1.
- A significant portion of *PR Bags* littered were unbranded in San Francisco (78 percent). About half of the *PR Bags* littered in Oakland (50 percent) and D.C. (49 percent) were unbranded (Table ES-3).
- *Other Retail Bags* (Home Depot, Dollar Tree, etc.) made up the largest portion of plastic bag litter (29 percent), followed by Bulk Food bags (21 percent) and Trash bags (19 percent).
- Field crews observed a number of unsecured trash setouts in all three cities. Such setouts have the potential to create litter of many types.
- Independent store sites had almost twice as many littered plastic bags as corporate store sites.
- San Francisco had the highest percentage of littered paper grocery bags (44 percent) compared to Oakland (2 percent) and D.C. (none).

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Table 7 (continued)

PLASTICS IN PRODUCTS IN MSW, 2011

(In thousands of tons, and percent of generation by resin)

Product Category	Generation (Thousand tons)	Recovery (Thousand tons)	Recovery (Percent of Gen.)	Discards (Thousand tons)
Plastic Containers & Packaging, cont.				
Other plastic containers				
HDPE	1,480	270	18.2%	1,210
PVC	30	Neg.		30
LDPE/LLDPE	30	Neg.		30
PP	240	20	8.3%	220
PS	90	Neg.		90
Subtotal Other Containers	**1,870**	**290**	**15.5%**	**1,580**
Bags, sacks, & wraps				
HDPE	700	60	8.6%	640
PVC	50			50
LDPE/LLDPE	2,350	370	15.7%	1,980
PP	660			660
PS	120			120
Subtotal Bags, Sacks, & Wraps	**3,880**	**430**	**11.1%**	**3,450**
Other Plastics Packaging‡				
PET	790	30	3.8%	760
HDPE	650	Neg.		650
PVC	320	Neg.		320
LDPE/LLDPE	1,140	Neg.		1,140
PLA	10	Neg.		10
PP	1,060	10	0.9%	1,050
PS	290	20	6.9%	270
Other resins	380	Neg.		380
Subtotal Other Packaging	**4,640**	**60**	**1.3%**	**4,580**
Total Plastics in Containers & Packaging, by resin				
PET	3,530	830	23.5%	2,700
HDPE	3,600	550	15.3%	3,050
PVC	400	Neg.		400
LDPE/LLDPE	3,520	370	10.5%	3,150
PLA	10	Neg.		10
PP	1,960	30	1.5%	1,930
PS	500	20	4.0%	480
Other resins	380	Neg.		380
Total Plastics in Cont. & Packaging	**13,900**	**1,800**	**12.9%**	**12,100**
Total Plastics in MSW, by resin				
PET	4,280	830	19.4%	3,450
HDPE	5,590	550	9.8%	5,040
PVC	900			900
LDPE/LLDPE	7,520	370	4.9%	7,150
PLA	50			50
PP	7,180	30	0.4%	7,150
PS	2,170	20	0.9%	2,150
Other resins	4,150	850	20.5%	3,300
Total Plastics in MSW	**31,840**	**2,650**	**8.3%**	**29,190**

HDPE = High density polyethylene
LDPE = Low density polyethylene
LLDPE = Linear low density polyethylene
PET = Polyethylene terephthalate
PLA = Polylactide
PP = Polypropylene
PS = Polystyrene
PVC = Polyvinyl chloride

‡ Other plastic packaging includes coatings, closures, lids, PET cups, caps, clamshells, egg cartons, produce baskets, trays, shapes, loose fill, etc.
PP caps and lids recovered with PET bottles and jars are included in the recovery estimate for PET bottles and jars.
Other resins include commingled/undefined plastic packaging recovery.
Some detail of recovery by resin omitted due to lack of data.

Table 7 (continued)

PLASTICS IN PRODUCTS IN MSW, 2012

(In thousands of tons, and percent of generation by resin)

Product Category	Generation (Thousand tons)	Recovery (Thousand tons)	Recovery (Percent of Gen.)	Discards (Thousand tons)
Plastic Containers & Packaging, cont.				
Other plastic containers				
HDPE	1,410	290	20.6%	1,120
PVC	40	Neg.		40
LDPE/LLDPE	40	Neg.		40
PP	280	20	7.1%	260
PS	80	Neg.		80
Subtotal Other Containers	**1,850**	**310**	**16.8%**	**1,540**
Bags, sacks, & wraps				
HDPE	700	50	7.1%	650
PVC	50			50
LDPE/LLDPE	2,280	390	17.1%	1,890
PP	640			640
PS	140			140
Subtotal Bags, Sacks, & Wraps	**3,810**	**440**	**11.5%**	**3,370**
Other Plastics Packaging‡				
PET	840	20	2.4%	820
HDPE	670	10	1.5%	660
PVC	330	Neg.		330
LDPE/LLDPE	1,070	Neg.		1,070
PLA	10	Neg.		10
PP	960	20	2.1%	940
PS	300	20	6.7%	280
Other resins	370	Neg.		370
Subtotal Other Packaging	**4,550**	**70**	**1.5%**	**4,480**
Total Plastics in Containers & Packaging, by resin				
PET	3,630	880	24.2%	2,750
HDPE	3,560	570	16.0%	2,990
PVC	420	Neg.		420
LDPE/LLDPE	3,390	390	11.5%	3,000
PLA	10	Neg.		10
PP	1,880	40	2.1%	1,840
PS	520	20	3.8%	500
Other resins	370	Neg.		370
Total Plastics in Cont. & Packaging	**13,780**	**1,900**	**13.8%**	**11,880**
Total Plastics in MSW, by resin				
PET	4,520	880	19.5%	3,640
HDPE	5,530	570	10.3%	4,960
PVC	870	Neg.		870
LDPE/LLDPE	7,350	390	5.3%	6,960
PLA	50	Neg.		50
PP	7,190	40	0.6%	7,150
PS	2,240	20	0.9%	2,220
Other resins	4,000	900	22.5%	3,100
Total Plastics in MSW	**31,750**	**2,800**	**8.8%**	**28,950**

HDPE = High density polyethylene
LDPE = Low density polyethylene
LLDPE = Linear low density polyethylene
PET = Polyethylene terephthalate
PP = Polypropylene
PLA = Polylactide
PS = Polystyrene
PVC = Polyvinyl chloride

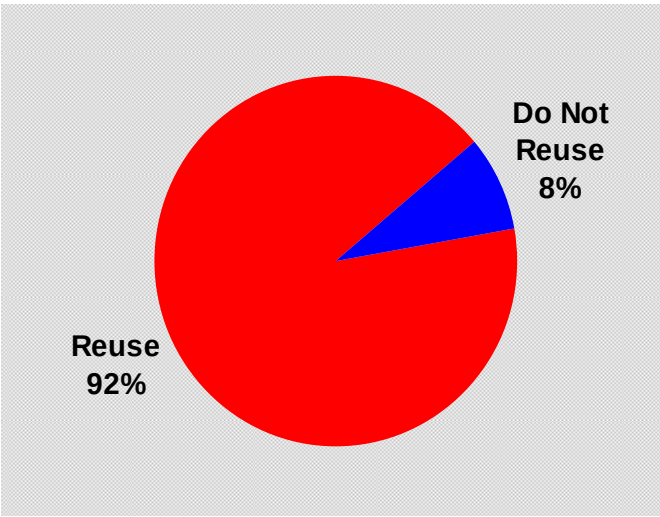
‡ Other plastic packaging includes coatings, closures, lids, PET cups, caps, clamshells, egg cartons, produce baskets, trays, shapes, loose fill, etc.
PP caps and lids recovered with PET bottles and jars are included in the recovery estimate for PET bottles and jars.
Other resins include commingled/undefined plastic packaging recovery.
Some detail of recovery by resin omitted due to lack of data.

According to a nationally representative survey conducted by APCO, a third-party research firm, over 92 percent of American consumers reuse their plastic bags.

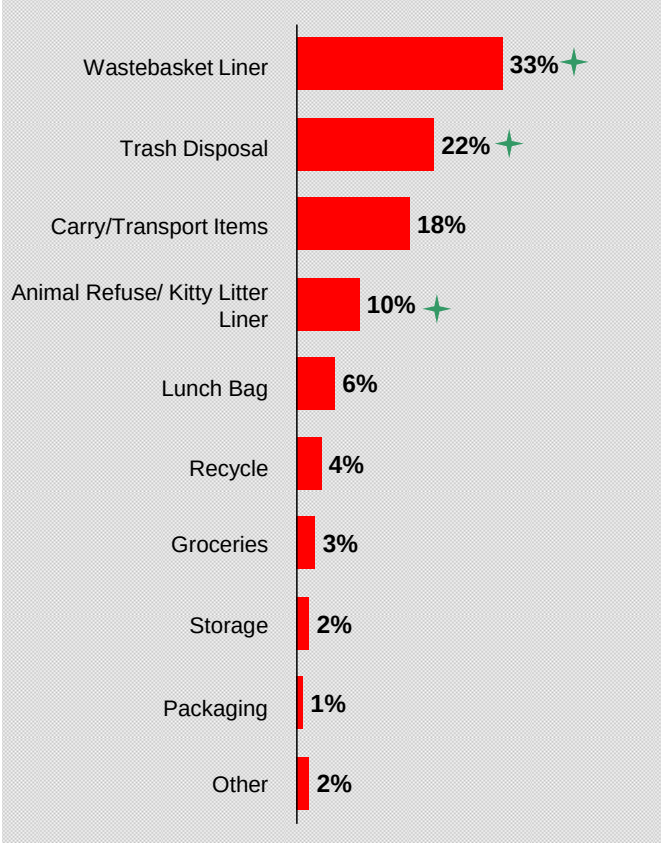
Near Universal Reuse of Plastic Shopping Bags…

- The reuse of plastic shopping bags is nearly universal, with about two thirds (65%) of respondents using them to contain trash

Reuse of Plastic Shopping Bags



Uses for Plastic Shopping Bags



Q5. Do you or does anyone in your household ever reuse plastic shopping bags?
Q6. [IF Q5 = YES]: What is the primary purpose you reuse plastic bags for? (n=462)

APCO insight

9

Methodology

- This APCO Insight study is an assessments of attitudes and awareness associated with the recycling of plastic shopping bags as well as a quantitative evaluation of two creative executions of plastic shopping bag recycling logos/posters. It was conducted among 502 randomly selected consumers who are responsible for household grocery shopping at least "some of the time".

Survey Population:	General Adult Public
Sample Design:	Screened Random Sample
Eligibility Criteria:	Responsible for some household grocery shopping. Marketing, PR, opinion research or media exclusion.
Sample Size:	n = 502
Margin of Error:	± 4.5 % (at 95% confidence level)
Data Collection Methodology:	Interactive TV panel
Field Dates:	03/06/07 – 03/15/06

2

APCO insight

December 10, 2013

Re: Product Declaration - Formolene® Polyethylene Resins

Dear Valued Customer**:**

Formolene® polyethylene resins are manufactured in Formosa's Point Comfort, Texas facility. Greater than 99 percent of all materials used in the manufacture of these resins are sourced from domestic suppliers. NAFTA Certificates of Origin are available upon request.

Please be advised that, Formosa Plastics Corporation does not intentionally add any of the materials listed below during the manufacture of *Formolene*® polyethylene resins. Furthermore, based upon our knowledge of the manufacturing process and information provided by our raw material suppliers, we would not expect these substances to be present in our final product. Analysis for these chemicals is not routinely performed. Please note that, as the supplier of the raw material, Formosa cannot make any claim with regard to extruded or molded products or components made from these resins.

Food Allergens
Formolene® polyethylene resins do not contain allergenic ingredients such as tree nuts, peanut products, soybean products, egg products, milk products, fish, shell fish, wheat products, sunflower seeds, poppy seeds, sesame seeds, or sulfites.

Genetically Modified Organisms
Genetically modified organisms are not used in the formulation or manufacture of *Formolene*® polyethylene resins.

Latex
To the best of our knowledge, the materials used, manufactured, and processed for *Formolene*® polyethylene resins do not contain natural latex rubber or dry natural rubber.

Ozone Depleting Substances
Materials listed in the Clean Air Act Amendments of 1990 (Class I CFC's, Class II HCFC's and the solvents, carbon tetrachloride and 1,1,1-trichloroethane) are not used in the manufacture of *Formolene*® polyethylene resins.

Heavy Metals
These resins comply with the package requirements for heavy metals as set forth by the Coalition of Northeast Governors (CONEG), the California Toxics in Packaging Act and Article 11 of EU Directive 94/62/EC. Lead, cadmium, mercury, and hexavalent chromium are not used in the formulation or manufacture of *Formolene*® polyethylene resins. The incidental (non-intentionally added) concentrations of these heavy metals does not exceed 100 parts per million by weight.

EU Restriction of Hazardous Substances (RoHS) Directive 2011/65/EU
These resins meet the safety and regulatory requirements for certification under this standard. Formosa Plastics Corporation does not intentionally add lead, cadmium, mercury, hexavalent chromium, polybrominated biphenyls (PBB), or polybrominated diphenyl ethers (PBDE) during the manufacture of *Formolene*® polyethylene resins.

California Proposition 65
To the best of our knowledge, *Formolene*® polyethylene resins do not contain any of the chemical substances listed by the State of California in Proposition 65.

Pentabromodiphenyl Ether & Octabromodiphenyl Ether
Pentabromodiphenyl Ether & Octabromodiphenyl Ether are not used in the formulation or manufacture of *Formolene*® polyethylene resins. To the best of our knowledge, this product is in compliance with EU Directives 2003/11/EC and 76/769/EEC.

Phthalates and Bisphenol A (BPA)
Phthalates (DEHP, DBP, BBP, DINP, DIDP, DNOP) or Bisphenol A are not used in the formulation or manufacture of *Formolene*® polyethylene resins.

Glycidyl Ethers (BADGE, BFDGE, NOGE)
Bisphenol A Diglycicyl ether (BADGE), Bisphenol F diglycidyl ether (BFDGE) and Novolac glycidyl ether (NOGE) are not used in the formulation or manufacture of *Formolene*® polyethylene resins.

Butylated Hydroxytoluene (BHT) & Butylated Hyroxyanisole (BHA)
Butylated Hydroxytoluene (BHT) & Butylated Hyroxyanisole (BHA) are not used in the formulation or manufacture of *Formolene*® polyethylene resins.

Organotin Compounds
Organotin compounds are not used in the formulation or manufacture of *Formolene*® polyethylene resins.

PFOA, PFOS, DMF
Perfluorooctanoic Acid (PFOA), Perfluorooctanesulfonic Acid (PFOS) and Dimethyl Fumarate (DMF) are not used in the formulation or manufacture of *Formolene*® polyethylene resins.

Some *Formolene*® polyethylene resins may contain the following ingredients:

Animal Derived Materials
Glycerol Monostearate and Calcium Stearate are used to manufacture some resin grades and these are known to contain substances derived from animals. However, these resin grades meet both European Union and U.S. Food & Drug Administration standards for being free from contamination with Transmissible Spongiform Encephalopathy (TSE) agents. Also, to the best of our knowledge, all materials used in cleaning and validation

are similarly compliant. If you require a resin grade to be completely free of animal derived materials, please contact your Sales Representative. Many grades can be made with alternate materials upon request.

As always, *You expect more. And Formosa delivers*®. If you have questions, please contact your Sales Representative.

Kind Regards,

Fred W. Neske, CIH, CSP, CFPS
Manager – Corporate Safety & Industrial Hygiene

IMPORTANT NOTICE:
The information and statements herein are believed to be reliable but are not to be construed as a warranty or representation for which we assume legal responsibility. Users should undertake sufficient verification and testing to determine the suitability for their own particular purpose of any information or products referred to herein. NO WARRANTY OF FITNESS FOR A PARTICULAR PURPOSE IS MADE.

Omya Inc.

61 Main Street
Proctor, Vermont 05765
Tel: (802) 459-3311 Fax: (802) 459-3428
www.omya-na.com

Product Data Sheet

OMYACARB® FT - FL

Production Plant	Florence, VT/USA	
Short Description of Product	A high purity, fine, wet ground, surface-treated natural calcium carbonate.	
Chemical Analysis of Raw Material	Calcium Carbonate, percent*	98
	Magnesium Carbonate, percent*	1
	Treatment, percent	1.1
Typical Properties	Y Brightness	95
	Retained on 325 mesh, ppm*	7
	Moisture Loss at 110°C, percent	0.04
	Hegman	5.5
	Specific Gravity	2.7
	Mean Refractive Index	1.57
	Pounds Pigment/Solid Gallon	22.6
General Product Data	Median Diameter, microns*	1.4
	Percent Finer than 2 Microns*	60
	Percent Finer than 1 Micron*	40
	Specific Surface Area, m^2/g	5.5

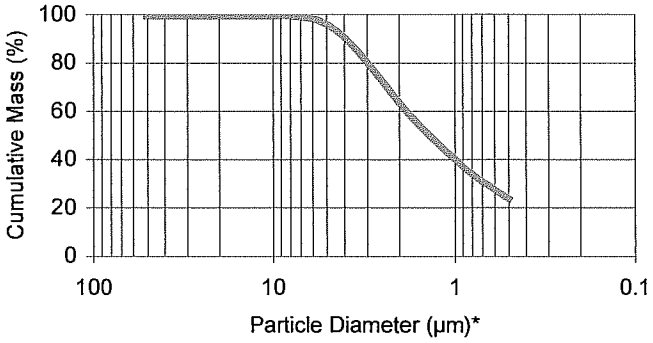
* Measured on untreated pigment

Applications

Plastics applications in rigid PVC, flexible PVC, and polyolefins. Paint applications in solvent based and powder coatings.

Regulatory Approvals

This product may be used as an indirect food additive in food packaging applications under 21 CFR (FDA) Sections 174.5, 175.300, and 178.3297. It does not quality as a substance permitted for direct addition to human food or animal feed.



Other than a representation that the products sold by Omya Inc. will, on the average, meet the criteria set forth above, which is and shall be subject to confirmation by the purchaser prior to the use of the products by purchaser, Omya Inc. makes no warranty, guarantee, or representation of any kind, express or implied, and specifically EXCLUDES without limitation any and all WARRANTIES OF MERCHANTABILITY AND FITNESS FOR A PARTICULAR PURPOSE AND MAKES NO WARRANTIES BEYOND THOSE CONTAINED HEREIN. If any of the products in any shipment do not conform to the representation contained herein, purchaser's sole remedy will be to provide written notice to Omya Inc. of such non-conforming product. Such notice shall be given within fourteen (14) days of product delivery to purchaser and Omya Inc., at its option if it determines the product does not conform, either promptly will replace the non-conforming product or will refund the purchase price paid for the non-conforming product. In no event shall Omya Inc. be liable for special, indirect or consequential damages nor shall Omya Inc. be liable for damages of any kind arising from the presence or use of products delivered, whether used singly or in combination with other substances. Omya Inc. disclaims any liability arising from the use of the products, which may infringe upon patents applied for, pending, or existing. No claim of any kind shall be greater than nor shall Omya Inc. in any event be liable for an amount in excess of the purchase price paid for the products in respect of which such claim is made.

valid: 8/16/2004
6-prodinfo>: USFL_PI_40210
version : 5

AMERICAN INKS AND COATINGS

(503) 284-1822 phone
(503) 284-1734 fax

Perseco Ink Compliance Certification

March 24, 2014

We certify that all materials sold in the United States to **Hilex Poly – All locations** for the manufacturing of packages and packaging components, comply with the following requirements for Perseco-McDonalds:

- 16 CFR, Section 1303 (Ban of Lead-Containing Paint)
- ASTM F963-96a:

Antimony	< 60 ppm
Arsenic	< 25 ppm
Barium	< 250 ppm
Cadmium	< 50 ppm
Chromium	< 25 ppm
Lead	< 90 ppm
Mercury	< 25 ppm
Selenium	< 100 ppm

- Inks will resist rub off under normal usage in the presence of moisture, carbonated beverage, grease, ketchup, vinegar, and mayonnaise following ASTM D5264.

Please let me know if you have any questions or if I can be of further assistance with this or any other regulatory issue you may have. Thank you.

Tony Ellis
EHS/Regulatory Manager

AMERICAN INKS AND COATINGS

(503) 284-1822 phone
(503) 284-1734 fax

CONEG Certification
(Reduction of Toxins in Packaging)

March 24, 2014

We certify that all materials sold to **Hilex Poly – All locations** for the manufacturing of packages and packaging components comply in all respects to the package requirements for heavy metals of the CONEG Model legislation; namely, that the sum of the concentration levels of lead, cadmium, mercury, and hexavalent chromium present in any package or package component shall not exceed 100 parts per million by weight as incidental contaminants (effective January 1, 1994).

American Inks does not use these items in our manufacturing process nor are they intentionally added to any of our products. We will maintain adequate documentation of this certification, including that of any exemptions permitted by legislation. Documentation will be made available for inspection.

Tony Ellis
EHS/Regulatory Manager

colortech
A PPM COMPANY

Mr.Glenn Strickland, Area Quality/Material Compliance Manager
Hilex Poly Co. LLC
2800 Sprouse Drive
Richmond Virginia 23231

CONEG CERTIFICATION AND 94/62/EEC COMPLIANCE REGARDING HEAVY METALS

Colortech Incorporated certifies that color and additive concentrate products, supplied to **Hilex Poly Co. LLC** comply in all respects to the package requirements for heavy metals as defined by the Coalition of Northeastern Governors Model Toxics Legislation (CONEG) and the European Parliament and Council Directive 94/62/EEC of 20 December 1994 on Packaging and Packaging Waste regarding heavy metal content.

Concentrate products supplied to **Hilex Poly Co. LLC** by Colortech are formulated with components, which do not incorporate lead, mercury, cadmium or hexavalent chromium as a key chemical constituent. To the best of our knowledge, the summation of the trace concentration levels of lead, mercury, cadmium and hexavalent chromium in these products do not exceed 100 parts per million (ppm) by weight. These assurances are based on typical trace heavy metal analyses provided by our raw material suppliers.

This certification applies to the following concentrate products currently supplied to **Hilex Poly Co. LLC:**

11932-18 Cover White
12557-18 Yellow
13271-18 HMF Orange
16976-18 Blue
18690-18 Kraft
19516-00 Gray
19554-73 Gray
20005-00 Black
57232-18 Green
58165-18 Kraft

In addition, Colortech agrees to maintain adequate documentation in support of this certification and to make it available for inspection on request.

This certification is subject to our normal terms and conditions of sale.

Signature: June 9, 2014

Beth McGhee - Folsom

Certified By: Beth McGhee-Folsom
Title: Manager of Quality Systems & Regulatory Affairs

Colortech Inc.,5712 Commerce Blvd, Morristown, TN 37814 Tel: (800) 248-2029 Fax: (423) 587-0841

January 17, 2014

Re: Formolene® Linear Low Density Polyethylene Products Approved for Food Contact Applications by the U.S. Food & Drug Administration

Dear Valued Customer:

The following Formolene® products are approved by the U.S. Food & Drug Administration (FDA) for use in food contact applications. Please note that some grades have limitations on the Conditions of Use as described in Table 2 of 21 CFR 176.170(c). These Conditions of Use are for all food types unless otherwise noted.

Linear Low Density Polyethylene - Hexene Copolymer

Resin	FDA Clearance 21 CFR 177.1520	Specifications 21 CFR 177.1520	Conditions of Use 21 CFR 176.170(c)	Comments
L62009A	(a)(3)(i)(a)(2)	(c)(3.1a)	C-H	
L62009H	(a)(3)(i)(a)(2)	(c)(3.1a)	C-H	
L62009E2	(a)(3)(i)(a)(2)	(c)(3.1a)	C-H	
L62009X	(a)(3)(i)(a)(2)	(c)(3.1a)	C-H	
L62022B	(a)(3)(i)(a)(2)	(c)(3.1a)	C-H	
L62030B	(a)(3)(i)(a)(2)	(c)(3.1a)	C-H	
L62608PA	(a)(3)(i)(a)(2)	(c)(3.2a)	B-H	
L62608PE	(a)(3)(i)(a)(2)	(c)(3.2a)	B-H	
L63550U/4	(a)(3)(i)(a)(1)	(c)(3.2a)	C-G	The finished food article must have a volume of at least 18.9 liters (5 gallons). Maximum thickness limitation of 0.003 inch in contact with food types V and VII-A.
L63565	(a)(3)(i)(a)(2)	(c)(3.2a)	B-H	
L63565U4	(a)(3)(i)(a)(2)	(c)(3.2a)	C-G	The finished food article must have a volume of at least 18.9 liters (5 gallons). Maximum thickness limitation of 0.003 inch in contact with food types V and VII-A.
L63568	(a)(3)(i)(a)(2)	(c)(3.2a)	B-H	
L63568U	(a)(3)(i)(a)(2)	(c)(3.2a)	C-G	The finished food article must have a volume of at least 18.9 liters (5 gallons). Maximum thickness limitation of 0.003 inch in contact with food types V and VII-A.
L63935U/4	(a)(3)(i)(a)(1)	(c)(3.2a)	C-G	The finished food article must have a volume of at least 18.9 liters (5 gallons). Maximum thickness limitation of 0.003 inch in contact with food types V and VII-A.
L64220U/4	(a)(3)(i)(a)(1)	(c)(3.2a)	C-G	The finished food article must have a volume of at least 18.9 liters (5 gallons)

Linear Low Density Polyethylene - Butene Copolymer

Resin	FDA Clearance 21 CFR 177.1520	Specifications 21 CFR 177.1520	Conditions of Use 21 CFR 176.170(c)	Comments
L42009A	(a)(3)(i)(c)(1)	(c)(3.2a)	A-H	
L42009B	(a)(3)(i)(c)(1)	(c)(3.2a)	A-H	
L42009E2	(a)(3)(i)(c)(1)	(c)(3.2a)	A-H	
L42009F	(a)(3)(i)(c)(1)	(c)(3.2a)	A-H	
L42009H	(a)(3)(i)(c)(1)	(c)(3.2a)	A-H	
L42009M	(a)(3)(i)(c)(1)	(c)(3.2a)	A-H	
L42009PE	(a)(3)(i)(c)(1)	(c)(3.2a)	A-H	
L42022B	(a)(3)(i)(c)(1)	(c)(3.2a)	A-H	
L42022E2	(a)(3)(i)(c)(1)	(c)(3.2a)	A-H	

Linear Low Density Polyethylene Injection Molding Grades

Resin	FDA Clearance 21 CFR 177.1520	Specifications 21 CFR 177.1520	Conditions of Use 21 CFR 176.170(c)	Comments
LH5204	(a)(3)(i)(c)(1)	(c)(3.2a)	A-H	Butene Copolymer
LH5206	(a)(3)(i)(c)(1)	(c)(3.2a)	A-H	Butene Copolymer
LH5314	(a)(3)(i)(c)(1)	(c)(3.2a)	A-H	Butene Copolymer
LH5320	(a)(3)(i)(c)(1)	(c)(3.2a)	A-H	Butene Copolymer
LH6008	(a)(3)(i)(c)(1)	(c)(3.2a)	A-H	Homopolymer
LH6008U	(a)(3)(i)(c)(1)	(c)(3.2a)	B-H	When used with food types III, IV-A, V, VII-A, and IX, finished article must have a volume of at least 18.9 liters (5 gallons)

FORMAX™ High Strength LLDPE – Hexene Copolymer

Resin	FDA Clearance 21 CFR 177.1520	Specifications 21 CFR 177.1520	Conditions of Use 21 CFR 176.170(c)	Comments
L71709A	(a)(3)(i)(a)(2)	(c)(3.2a)	B-H	
L71709E	(a)(3)(i)(a)(2)	(c)(3.2a)	B-H	
L71709H	(a)(3)(i)(a)(2)	(c)(3.2a)	B-H	
L17109S	(a)(3)(i)(a)(2)	(c)(3.2a)	B-H	

FORMAX™ High Performance LLDPE – Hexene Copolymer

Resin	FDA Clearance 21 CFR 177.1520	Specifications 21 CFR 177.1520	Conditions of Use 21 CFR 176.170(c)	Comments
L91507A	(a)(3)(i)(a)(2)	(c)(3.2a)	B-H	
L91507E2	(a)(3)(i)(a)(2)	(c)(3.2a)	B-H	
L91507E3	(a)(3)(i)(a)(2)	(c)(3.2a)	B-H	
L91507H	(a)(3)(i)(a)(2)	(c)(3.2a)	B-H	

In all food applications, we recommend that the packager or manufacturer of the final product conduct appropriate tests to evaluate the possible contribution of the container to the aroma, taste and color of the food product.

As always, *You expect more. And Formosa delivers®*. If you have questions regarding FDA compliance for any Formosa Plastics Corporation product, please contact your Sales Representative.

Kind Regards,

Fred W. Neske, CIH, CSP, CFPS
Manager – Corporate Safety & Industrial Hygiene

IMPORTANT NOTICE:
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January 17, 2014

Re: Formolene® High Density Polyethylene Products Approved for Food Contact Applications by the U.S. Food & Drug Administration

Dear Valued Customer:

The following Formolene® products are approved by the U.S. Food & Drug Administration (FDA) for use in food contact applications. Please note that some grades have limitations on the Conditions of Use as described in Table 2 of 21 CFR 176.170(c). These Conditions of Use are for all food types unless otherwise noted.

Resin	FDA Clearance 21 CFR 177.1520	Specifications 21 CFR 177.1520	Conditions of Use 21 CFR 176.170(c)	Comments
HB6007	(a)(2)(i)	(c)(2.2)	A-H	Homopolymer
HB4903	(a)(3)(i)(a)(1)	(c)(3.2a)	B-H	Hexene Copolymer
HB5202B	(a)(3)(i)(a)(1)	(c)(3.2a)	A-H	Hexene Copolymer
HB5202B2	(a)(3)(i)(a)(1)	(c)(3.2a)	A-H	Hexene Copolymer
HB5502B	(a)(3)(i)(a)(1)	(c)(3.2a)	A-H	Hexene Copolymer
HB5502B4	(a)(3)(i)(a)(1)	(c)(3.2a)	A-H	Hexene Copolymer
HB5502F	(a)(3)(i)(a)(1)	(c)(3.2a)	A-H	Hexene Copolymer
HB5502S1	(a)(3)(i)(a)(1)	(c)(3.2a)	B-H	Hexene Copolymer
HB5502Z	(a)(3)(i)(a)(1)	(c)(3.2a)	A-H	Hexene Copolymer
HF3728	(a)(3)(i)(a)(1)	(c)(3.2a)	A-H	Hexene Copolymer
HL3812	(a)(3)(i)(a)(1)	(c)(3.2a)	C-G	Finished articles must have a volume of at least 18.9 liters (5 gallons).
HF4728	(a)(3)(i)(a)(1)	(c)(3.2a)	A-H	Hexene Copolymer
HL5010	(a)(3)(i)(a)(1)	(c)(3.2a)	B-H	Hexene Copolymer
E790T1	(a)(3)(i)(c)(1)	(c)(3.2a)	A-H	Butene Copolymer
E791T1	(a)(3)(i)(c)(1)	(c)(3.2a)	A-H	Butene Copolymer
E922	(a)(3)(i)(c)(1)	(c)(3.2a)	A-H	Butene Copolymer
E924	(a)(3)(i)(c)(1)	(c)(3.2a)	A-H	Butene Copolymer
E924F	(a)(3)(i)(c)(1)	(c)(3.2a)	A-H	Butene Copolymer
E924H	(a)(3)(i)(c)(1)	(c)(3.2a)	A-H	Butene Copolymer
E924ND	(a)(3)(i)(c)(1)	(c)(3.2a)	A-H	Butene Copolymer
E925	(a)(3)(i)(c)(1)	(c)(3.2a)	A-H	Butene Copolymer
E927	(a)(3)(i)(c)(1)	(c)(3.2a)	A-H	Butene Copolymer
E927ND	(a)(3)(i)(c)(1)	(c)(3.2a)	A-H	Butene Copolymer

In all food applications, we recommend that the packager or manufacturer of the final product conduct appropriate tests to evaluate the possible contribution of the container to the aroma, taste and color of the food product.

As always, *You expect more. And Formosa delivers*®. If you have questions regarding FDA compliance for any Formosa Plastics Corporation product, please contact your Sales Representative.

Kind Regards,

Fred W. Neske, CIH, CSP, CFPS
Manager – Corporate Safety & Industrial Hygiene
Formosa Plastics Corporation, U.S.A.

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2011 IOWA STATEWIDE WASTE CHARACTERIZATION STUDY

Prepared for:

IOWA DEPARTMENT OF NATURAL RESOURCES

Final Report

September 14, 2011

MSWCONSULTANTS

MID ATLANTIC SOLID WASTE CONSULTANTS
6225 Sawyer Road, New Market, MD 21774 301/607-6428
842 Spring Island Way, Orlando, FL 32828 407/380-8951
3407 Chestnut Street, Camp Hill, PA 17011 717/731-9708
www.mswconsultants.us

in conjunction with

CASCADIA CONSULTING GROUP

Foth

Foth Infrastructure & Environment

ACKNOWLEDGEMENTS

The Project Team of MidAtlantic Solid Waste Consultants (MSW Consultants), Cascadia Consulting Group, and Foth Infrastructure & Environment, LLC (Foth) would like to thank the management and operations staff of the following landfills and transfer stations that participated in this project by supplying detailed data about their incoming waste streams, and subsequently hosting sampling and sorting activities at their disposal facilities:

- Des Moines County Regional Solid Waste Commission,
- Ottumwa-Wapello Solid Waste Commission,
- South Central Iowa Solid Waste Agency,
- Poweshiek Transfer Station,
- Metro Waste Authority,
- Carroll County Solid Waste Management Commission,
- Crawford County Area Solid Waste Agency,
- Shelby County Solid Waste Agency,
- Northwest Iowa Area Solid Waste Agency,
- Dubuque Metropolitan Area Solid Waste Agency,
- Iowa City Landfill & Recycling Center, and
- Waste Commission of Scott County.

so-called Mixed Waste results were provided as if Mixed Waste was a separate generator sector. A number of steps were taken in this study update to reduce reliance on loads of Mixed Waste, and also to obtain grab samples from Mixed loads that could be identified as Residential or ICI. Details are provided in the body of the report.

ES 2. OVERVIEW OF RESULTS

Figure ES-1 shows the breakdown of major material groups for the aggregate Iowa statewide MSW stream (top pie chart); the Residential and ICI waste stream pie charts are directly underneath so that readers can quickly compare the contribution of various material groups. Results are shown in estimated percent composition disposed.

Figure ES-1 2011 Iowa Statewide MSW Composition

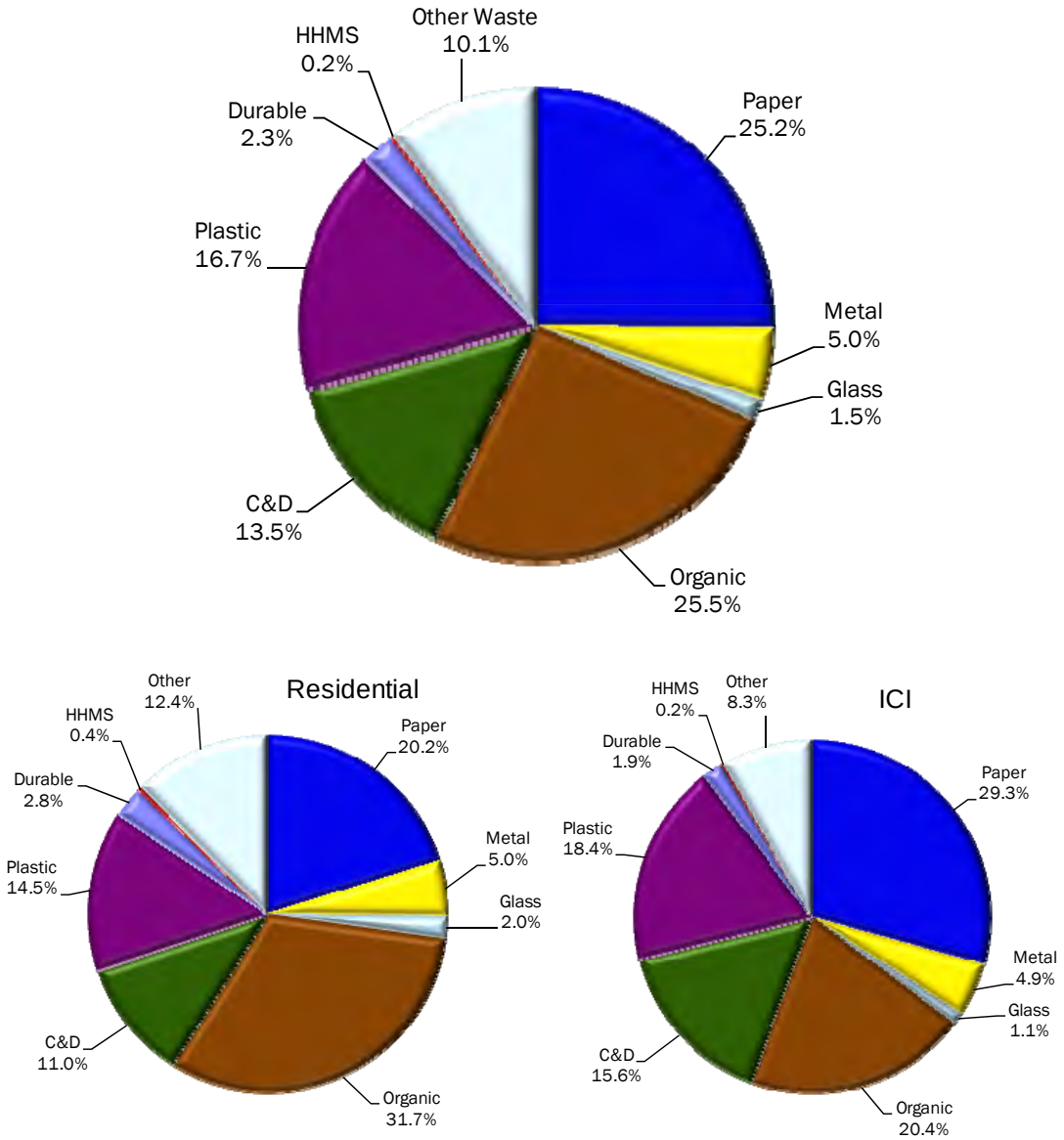


Table ES-1 shows the five most prevalent individual material categories disposed by Statewide Aggregate, Residential, and ICI generating sectors. The percent composition is shown in the table.

Table ES-1 Top 5 Most Prevalent Material Categories

Rank	Statewide MSW	Residential Waste	ICI Waste
1	Food Waste - 13.3%	Food Waste - 13.6%	OCC and Kraft Paper - 13.2%
2	OCC and Kraft Paper - 9.0%	Yard Waste - 7.8%	Food Waste - 13.1%
3	Other Film Plastic - 6.6%	Textiles and Leather - 5.9%	Other Plastic Products - 8.0%
4	Compostable Paper - 6.1%	Other Film Plastic - 5.8%	Wood – Untreated - 7.9%
5	Untreated Wood - 5.4%	Other Plastic Products - 5.2%	Other Film Plastic - 7.3%
Total	40.4%	38.3%	49.5%

Table ES-2 shown on the following page provides a detailed snapshot of the statewide MSW stream. Full results for statewide aggregate MSW, as well as for individual generator sectors and for Solid Waste, are contained in the full report.

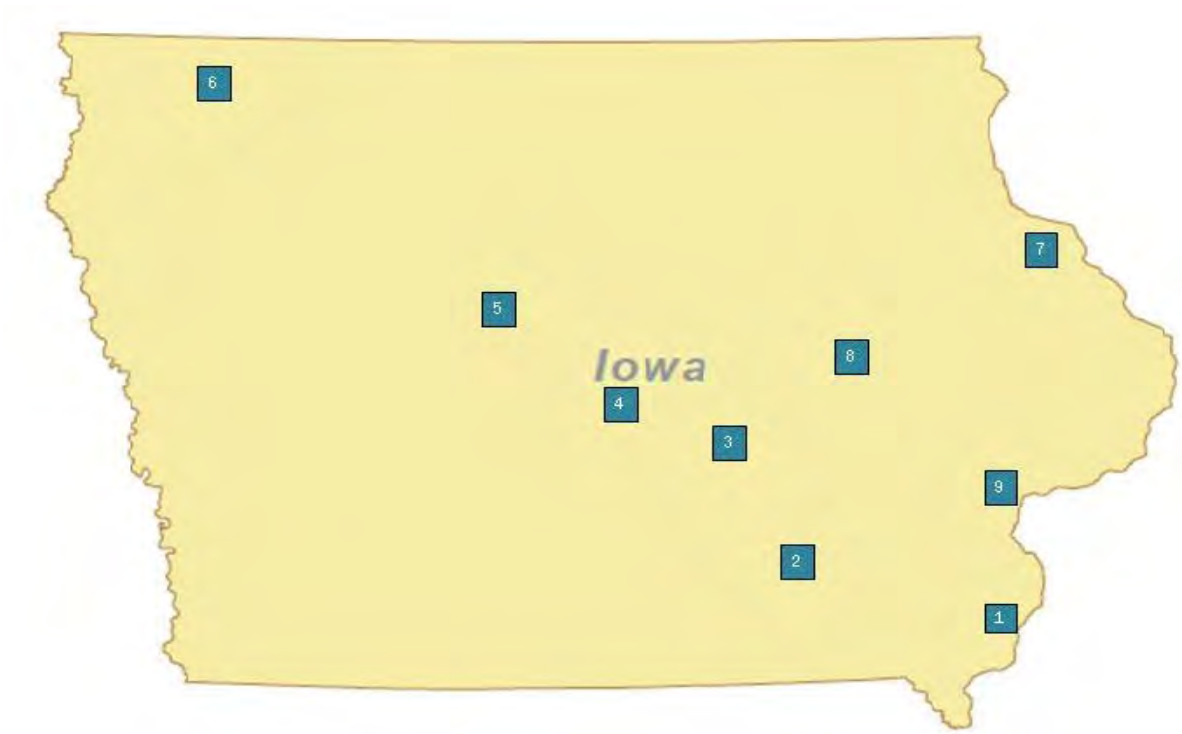
Table ES-2 Statewide MSW Detailed Composition Results

Material	Estimated Percent	Lower Bound	Upper Bound	Material	Estimated Percent	Lower Bound	Upper Bound
Paper	**25.2%**	**23.5% -**	**26.9%**	**Plastic**	**16.7%**	**15.0% -**	**18.3%**
Compostable Paper	6.1%	5.4% -	6.8%	#1 PET IA Deposit Beverage Containers	0.1%	0.1% -	0.2%
High Grade Office Paper	0.9%	0.6% -	1.2%	#1 PET Beverage Containers	0.5%	0.4% -	0.5%
Magazines/Catalogs	1.2%	1.0% -	1.4%	#2 HDPE Containers Natural	0.3%	0.2% -	0.3%
Mixed Recyclable Paper	3.7%	3.4% -	4.0%	#2 HDPE Containers Colored	0.4%	0.3% -	0.4%
Newsprint	1.6%	1.4% -	1.9%	Retail Shopping Bags	0.3%	0.2% -	0.3%
Non-Recyclable Paper	2.4%	1.8% -	2.9%	Other Film Plastic	6.4%	5.6% -	7.1%
OCC and Kraft Paper	9.0%	7.5% -	10.5%	Other #1 PET Containers	0.3%	0.2% -	0.3%
Aseptic/Gable Top Containers	0.2%	0.2% -	0.3%	Plastic Containers #3-#7	0.7%	0.6% -	0.7%
				Other plastic Containers	1.1%	0.1% -	2.0%
Metal	**5.0%**	**3.7% -**	**6.2%**	Expanded Polystyrene	1.5%	0.3% -	2.7%
Aluminum Beverage Containers	0.0%	0.0% -	0.1%	Other Plastic Products	5.3%	4.3% -	6.3%
Aluminum IA Deposit Beverage Containers	0.2%	0.2% -	0.2%				
Ferrous Food and Beverage Containers	0.7%	0.5% -	0.9%	**Durable**	**2.3%**	**1.5% -**	**3.1%**
Other Aluminum Containers	0.3%	0.2% -	0.4%	Cell Phones and Chargers	0.0%	0.0% -	0.0%
Other Ferrous Scrap Metals	2.2%	1.7% -	2.6%	Central Processing Units/Peripherals	0.4%	0.1% -	0.6%
Other Non-Ferrous Scrap Metals	1.6%	0.4% -	2.7%	Computer Monitors/T.V.s	0.3%	0.1% -	0.5%
				Electrical and Household Appliances	1.6%	0.9% -	2.3%
Glass	**1.5%**	**1.3% -**	**1.7%**				
Blue Glass	0.0%	0.0% -	0.0%	**HHMS**	**0.2%**	**0.2% -**	**0.3%**
Brown Glass	0.1%	0.1% -	0.2%	Automotive Products	0.1%	0.0% -	0.2%
Clear Glass	0.5%	0.5% -	0.6%	Household Cleaners	0.0%	0.0% -	0.0%
Glass Deposit Containers	0.2%	0.2% -	0.3%	Lead Acid Batteries	0.0%	0.0% -	0.0%
Green Glass	0.1%	0.0% -	0.1%	Mercury Container Products	0.0%	0.0% -	0.0%
Other Mixed Cullet	0.6%	0.4% -	0.7%	Other Batteries	0.1%	0.0% -	0.1%
				Paints and Solvents	0.0%	0.0% -	0.1%
Organic	**25.5%**	**23.8% -**	**27.1%**	Pesticides, Herbicides, Fungicides	0.0%	0.0% -	0.0%
Yard Waste	4.6%	3.9% -	5.2%	Sharps	0.0%	0.0% -	0.0%
Food Waste	13.3%	11.9% -	14.8%	Prescription Medications	0.0%	0.0% -	0.0%
Textiles and Leather	4.1%	3.5% -	4.7%				
Diapers	2.5%	2.1% -	2.8%	**Other**	**10.1%**	**9.2% -**	**11.1%**
Rubber	1.0%	0.6% -	1.5%	Other Organics	3.2%	2.7% -	3.7%
				Other Inorganics	0.3%	0.1% -	0.4%
C&D	**13.5%**	**11.3% -**	**15.7%**	Other C&D	1.1%	0.8% -	1.4%
Wood – Untreated	5.4%	3.7% -	7.1%	Other Durables	2.1%	1.4% -	2.8%
Wood – Treated	3.8%	3.1% -	4.5%	Other HHM	0.0%	0.0% -	0.1%
Asphalt Pavement, Brick, Rock, and Concrete	0.7%	0.5% -	0.9%	Fines	3.1%	2.6% -	3.5%
Asphalt Roofing	0.8%	0.0% -	1.6%	Other	0.5%	0.1% -	0.8%
Drywall/Gypsum Board	1.0%	0.5% -	1.5%				
Carpet and Carpet Padding	1.8%	1.2% -	2.5%	**Totals**	**100.0%**		
				Sample Count	**460**	**Conf.**	**90%**

2.3. HOST FACILITIES AND DISPOSAL QUANTITIES

The 1998 Study included five landfills and the 2005 Study included six landfills for hosting waste characterization analysis. The 2011 update has expanded the total to nine host solid waste commissions/agencies. Figure 2-1 below shows relative locations of the nine organizations that hosted sampling and sorting activities.

Figure 2-1 Location of Host Landfills, 2011 Study



1	Des Moines County Regional Solid Waste Commission	2	Ottumwa-Wapello County Solid Waste Commission	3	South Central Iowa Solid Waste Agency
4	Metro Waste Authority	5	Carroll County Solid Waste Management Commission	6	Northwest Iowa Area Solid Waste Agency
7	Dubuque Metropolitan Area Solid Waste Agency	8	Iowa City Landfill	9	Waste Commission of Scott County

Table 2-2 shows the tonnage and percentage of Municipal Solid Waste (MSW) disposed in the targeted landfills and total state-wide for the 2010 fiscal year. This table also shows which of the host organizations in this year's study hosted sampling and sorting events in prior studies. As shown, the nine targeted organizations dispose a little over 42 percent of the State's disposed MSW.

Table 4-1 (Continued) Comparison with Prior Studies – MSW

Comparison of Statewide Overall MSW Composition										
Materials	2011 Results			2005 Results			1998 Results			Likely Statistically Significant
	Mean	Lower Bound	Upper Bound	Mean	Lower Bound	Upper Bound	Mean	Lower Bound	Upper Bound	
Organic	**25.5%**	**23.8%**	**27.1%**	**20.1%**			**19.7%**			
Pumpkins	NA	NA	NA	0.3%	0.2%	0.4%	0.4%	0.3%	0.5%	
Yard Waste	4.6%	3.9%	5.2%	1.4%	1.0%	1.9%	1.3%	1.1%	1.5%	x
Food Waste	13.3%	11.9%	14.8%	10.6%	9.3%	12.2%	10.7%	9.6%	11.8%	
Textiles and Leather	4.1%	3.5%	4.7%	4.9%	4.0%	6.1%	4.2%	3.7%	4.7%	
Diapers	2.5%	2.1%	2.8%	2.4%	2.0%	2.8%	2.3%	2.0%	2.7%	
Rubber	1.0%	0.6%	1.5%	0.5%	0.3%	0.6%	0.8%	0.7%	0.9%	x
C&D	**13.5%**	**11.3%**	**15.7%**	**13.5%**			**11.2%**			
Wood – Untreated	5.4%	3.7%	7.1%	3.4%	2.7%	4.4%	2.8%	2.3%	3.2%	
Wood – Treated	3.8%	3.1%	4.5%	4.6%	3.6%	6.0%	3.6%	3.0%	4.2%	
Asphalt Pavement, Brick, Rock, and Concrete [1]	0.7%	0.5%	0.9%	5.5%	4.1%	7.6%	4.8%	4.0%	5.7%	
Asphalt Roofing [1]	0.8%	0.0%	1.6%							
Drywall/Gypsum Board [1]	1.0%	0.5%	1.5%							
Carpet and Carpet Padding [1]	1.8%	1.2%	2.5%							
Plastic	**16.7%**	**15.0%**	**18.3%**	**14.9%**	**13.4%**	**16.6%**	**14.4%**	**13.3%**	**15.6%**	
#1 PET IA Deposit Beverage Containers	0.1%	0.1%	0.2%	0.2%	0.2%	0.2%	0.1%	0.1%	0.2%	x
#1 PET Beverage Containers	0.5%	0.4%	0.5%	0.4%	0.4%	0.5%	0.2%	0.2%	0.3%	
#2 HDPE Containers Natural [2]	0.3%	0.2%	0.3%	1.0%	0.9%	1.2%	1.0%	0.9%	1.1%	
#2 HDPE Containers Colored [2]	0.4%	0.3%	0.4%							
Plastic Retail Shopping Bags [3]	0.3%	0.2%	0.3%	6.6%	5.7%	7.5%	4.8%	4.3%	5.2%	
Other Film Plastic [3]	6.4%	5.6%	7.1%							
Other #1 PET Containers	0.3%	0.2%	0.3%	0.3%	0.2%	0.3%	NA	NA	NA	
#3-#7 Plastic Containers	0.7%	0.6%	0.7%	NA	NA	NA	NA	NA	NA	
Other Plastic Containers	1.1%	0.1%	2.0%	0.4%	0.3%	0.5%	0.8%	0.7%	0.9%	
Expanded Polystyrene	1.5%	0.3%	2.7%	NA	NA	NA	NA	NA	NA	
Other Plastic Products	5.3%	4.3%	6.3%	6.0%	5.3%	6.9%	7.5%	6.7%	8.4%	

Citizens Group Pushes City Council to Repeal Disposable Shopping Bag Fee

October 23, 2014

On 21 October 2014, the Fort Collins, Colorado City Council repealed the disposable bag ordinance which would have imposed a 5-cent fee for each disposable paper or plastic bag distributed by retailers in the city beginning on April 1, 2015. (Gordon, 2014)

The ordinance was originally passed by the council on a 5-2 vote on August, 19, 2014. The ordinance would have required all retailers to impose a fee of 5-cents for each disposable paper or plastic shopping bag issued at the point of sale to customers. Exempt from the fee are plastic or paper bags used to package bulk food items and bags used to contain frozen food or meat and prevent contamination of reusable bags. Also exempt from the fee are newspaper bags, dry-cleaning bags, and bags sold in bulk packages to consumers such as trash bags. The purpose of the fee was to discourage use of disposable bags and encourage the use of reusable bags. (City of Fort Collins, 2014)

The ordinance met with swift opposition and the group Citizens For Recycling Choices filed a protest with the City Clerk's Office on August 26, 2014. (Udell, Fighters of plastic bag fee out gathering signatures, 2014) The group needed to collect 2,604 signatures and managed to collect more than 4,000 signatures. (Udell, Bag fee opponents collect more than 4K signatures, 2014) The group used social media and word of mouth to publicize the signature gathering effort. (Udell, Fighters of plastic bag fee out gathering signatures, 2014)

Mike Pruznick, an opponent of the bag fee and who helped to start the 200-memberCitizens for Recycling Choices group, said that the bag fee does not protect the environment, because shoppers will avoid the fee and stop getting plastic bags at grocery stores and instead of having these plastic bags on hand to reuse, such as for taking out the trash, will start using larger bags that contain more plastic. He stated that there is a use for both bags in our society. (CBS Denver, 2014)

The group had great support from residents to overturn the disposable bag fee. Some shoppers indicated that they would shop in neighboring communities if the bag fee was not overturned.

After the council repealed the ordinance, they also directed the City Manager not to pursue bag fees/bans as part of the Zero Waste program.

The City Council reiterated the City's commitment to reuse and recycling by acceleratingzero-waste goals in a new Zero Waste Resolution.

"We heard our citizens and responded – the disposable bags ordinance is not the way to go," said Bruce Hendee, Chief Sustainability Officer. "However, our community has old us that they want us to continue to find innovative ways to increase recycling and

environmental sustainability, especially with the Larimer County Landfill filling up. Tonight, we accelerated that road to zero waste by suggesting staff explore some key techniques." (Gordon, 2014)

The resolution adopted Tuesday directs staff to take a number of specific, short-term steps:

- Expand community-wide awareness of measures to prevent waste from being created
- Construct a new Community Recycling Center in 2015, subject to funding approval, for a one-stop recycling facility that also accepts a variety of hard-to-recycle materials such as electronic waste and certain household hazardous wastes
- Explore a universal recycling ordinance that would improve access to curbside recycling and organics collection for residents and businesses

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REUSABLE BAG BAN REPORT (DRAFT)

Implementation Progress

BACKGROUND

The City of Santa Fe passed Ordinance 2013-29 eliminating the single-use plastic bags from most retail stores and requiring a ten cent fee for single-use paper bags over a specific size. Subsequently it was discovered that the fee, as written in the ordinance, was beyond the City's authority to require and was eliminated through Ordinance 2014-08.

The ordinance, as amended, went into effect on February 27, 2014. The Environmental Services Division was charged with collecting data regarding the financial impact to stores and the Sustainable Santa Fe Commission was charged with reporting on the progress and effectiveness of the ordinance one year from its passage on August 27, 2013. This is that report.

INTENT

Reducing the impacts of solid waste involves reducing the volumes of waste, reusing waste materials several times, and recycling the remainder, in that order. The greatest reduction in impacts comes from the hierarchy of these actions. Reusing shopping bags therefore has a greater reduction in impacts than recycling single-use bags. This ordinance is intended to promote the use of reusable bags as much as possible.

GOAL:

Reduce single-use bag usage in Santa Fe by eliminating single-use plastic bags and encouraging people to avoid using paper bags

SUMMARY OF FINDINGS

Implementation of the ordinance has not resulted in significant public complaint, however, for the most part, people have exchanged using single-use plastic bags for single-use paper bags with little increase in reusable bags. Therefore, in order to meet the goal of reducing single-use bag usage, additional steps would need to be taken to discourage use of paper bags.

RECOMMENDATIONS

The Sustainable Santa Fe Commission recommends that the City adopt a disincentive for paper bag use that also generates revenue for the City and the stores to help cover administrative costs and facilitate the implementation of additional green initiatives. The rules governing cities authority differs from state to state. Here in New Mexico Santa Fe does not have the authority to require a third party (retail stores) to collect a fee for bags. The City Attorney is exploring all available options within the authority of Santa Fe as a Home Rule NM City. To that end she has submitted a request for interpretation to the New Mexico Attorney General as well as information from the New Mexico Municipal League.

METHODOLOGY

There is no existing data, and no easy way to collect data, regarding the number of single-use bags used in Santa Fe. The Environmental Services Division developed the methodology described below for collecting qualitative data from retail stores to assess the impacts of the ordinance. All stores contacting provided data during two separate interview times.

Data Sampling

The City was divided into three geographical areas: downtown and surrounding area, mid-town, and southside. 10 to 11 stores were selected to be surveyed in each of these three areas at two times. Surveys were conducted about 2 months after implementation and again about 4 months after implementation. Additionally, 12 retail stores that are large users of single-use bags were surveyed from across the City, some with multiple locations. These stores were surveyed about 4 months after implementation. The survey questions from the first survey focused on the financial impacts and perceptions of the public's feelings towards the bill. The same questions were asked during the second survey plus some additional questions were added to understand how the bill was being implemented internally at the business and to get any data available regarding actual reductions in single bag use.