Title:

Leaching of the plasticizer di (2-ethylhexyl) phthalate (DEHP) from plastic containers and the question of human exposure.

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Abstract

Di (2-ethylhexyl) phthalate (DEHP) is a widely used plasticizer to render poly(vinyl chloride) (PVC) soft and malleable. Plasticized PVC is used in hospital equipment, food wrapping, and numerous other commercial and industrial products. Unfortunately, plasticizers can migrate within the material and leach out of it over time, ending up in the environment and, frequently, the human body. DEHP has come under increased scrutiny as its breakdown products are believed to be endocrine disruptors and more toxic than DEHP itself. DEHP and its breakdown products have been identified as ubiquitous environmental contaminants and daily human exposure is estimated to be in the μg per kg level.

The objective of this review is to summarize and comment on published sources of DEHP exposure. Exposure through bottled water was examined specifically, as this concern is raised frequently, yet only little exposure to DEHP occurs through bottled water, and DEHP exposure is unlikely to stem from the packaging material itself. Packaged food was also examined and showed higher levels of DEHP contamination compared to bottled water. Exposure to DEHP also occurs in hospital environments, where DEHP leaches directly into liquids passed through PVC/DEHP tubing and equipment. The latter exposure is at considerably higher levels compared to food and bottled water, specifically putting patients with chronic illnesses at risk.

Overall, levels of DEHP in food and bottled water were below current tolerable daily intake (TDI) values. However, our understanding of the risks of DEHP exposure are still evolving. Given the prevalence of DEHP in our atmosphere and environment, and the uncertainty revolving around it, the precautionary principle would suggest its phase out and replacement. Increased efforts to develop viable replacement compounds, which necessarily includes rigorous leaching, toxicity and impact assessment studies, is needed before alternative plasticizers can be adopted as viable replacements.

Introduction

The use of polymers is increasing on a worldwide scale (PEMRG 2013) and along with it is the use of the many compounds added to these polymers to serve a broad variety of functions (Sears and Darby 1982). Such additives include plasticizers, coloring agents, flow aids, heat stabilizers and solvents. Due to their widespread and growing use, there is an increasing interest and concern as to the impacts that these additives have on humans, animals and the environment. This is especially important given that most of these additives are not chemically bound to the polymer chains, which means that they can migrate within the material and leach out over time. Furthermore, since these additives are usually small molecules compared to the large polymer chains, they are more prone to dissolve in the aqueous environment, thereby contributing to the risk of exposure of humans, animals and microbes.

Plasticizers account for a large fraction of additives to polymers (Murphy 2001) and can represent as much as 40 % by weight of such materials (ATSDR 2002). These molecules interact with the polymer chains to render the material more flexible and malleable. The most important class of plasticizers in widespread use are phthalate diesters and, within these phthalates, DEHP (Fig.1) is the most important (Murphy 2001). DEHP has also been the source of the greatest concern. For example, several studies have suggested that it is toxic and can disrupt normal endocrine function in humans (see section below "Varieties of phthalates: Phthalates and terephthalates"). Furthermore, the US Environmental Protection Agency (US EPA) has expressed its concern "about phthalates because of their toxicity and the evidence of pervasive human and environmental exposure to them" (US EPA 2012). Due to these concerns and

growing evidence of health impacts, certain phthalates have been banned for use in children's toys in the European Union (EU 2005), the United Stated (CPSIA 2008) and Canada (HPA 2010).

There have been recent reports of phthalates, specifically DEHP, in water bottles made from polyethylene terephthalate (PET) (see Table 1 and section below "Bottled Water"). PET is a hard material that is used to make containers, especially for use by the beverage industry. However, it does not contain, and is not blended, with DEHP. A plasticizers' role is to soften hard materials. PET also cannot release DEHP, as the terephthalate used to make the backbone of the polymer is a different chemical compound, while DEHP is a low molecular weight additive to the polymer resin. Contamination from DEHP in water bottles must stem from a processing step before bottling, or from the cap liner used.

There is growing public awareness regarding the human and environmental impacts associated with the use of DEHP owing to numerous recent scientific reports and increases in the regulation of this compound in various countries around the world. This awareness is also coupled with misconceptions about their use. Therefore, this article aims to clarify the risk of exposure to DEHP via liquids stored in plastic but also glass containers, including water, beverages and food, and in various types of hospital equipment. Measured and estimated levels of DEHP in a variety of samples stored in commercial and clinical containers reported in the literature are presented. The article also summarizes levels of phthalates present in the environment, resulting in a human "background exposure".

Varieties of phthalates: Phthalates and terephthalates

Because of a worldwide increase in the consumption of bottled water (Beverage Marketing Corporation 2011), the materials used to make these water bottles have come under increased scrutiny. Polymers such as polyethylene (PE; Recycling code #2) and polyethylene terephthalate (PET, also known as PETE; Recycling code #1; see Fig.1) are used in lieu of glass, mainly due to their low density and durability. The 2011 Beverage Market Report reported that "plastic packaging is preferred over glass in almost every country" and that "PET is the most dynamic and rapidly growing segment" (Beverage Marketing Corporation 2011). PET is a polyester that is produced either from the esterification reaction of

terephthalic acid with ethylene glycol (see Fig.1) and subsequent polymerization or, alternatively, by the transesterification of dimethyl terephthalate with ethylene glycol and subsequent polymerization. Once polymerized, the resulting polyester is a hard and rather brittle material. The only means by which some terephthalic acid, or an ester of it, could leach from it is by gradual depolymerization of the polymer chains forming the material. Fig.1 shows that the two carbonyl functions of terephthalic acid are in *para*-position to one another, on opposite ends of the aromatic ring. Terephthalic acid was originally reported to be non-toxic (Hoshi et al. 1968), but later studies involving higher levels of exposure to this compound suggested that it may impair testicular functions (Cui et al. 2004). An extensive review on terephthalic acid has been recently published (Ball et al. 2011). Although terephthalic acid has a certain resemblance in terms of molecular structure to DEHP, it cannot leach out of the final product and is chemically distinct from DEHP. Hence, it will not be dealt with further in this review article.

Phthalate plasticizers such as DEHP are synthesised from phthalic acid, in which the two carbonyl groups are in *ortho*-position to one another; i.e. on neighbouring carbon atoms in the aromatic ring (Fig.1). Phthalic acid is esterified with various alcohols, but most commonly with 2-ethyl hexanol, to form liquid DEHP (Fig.1). This is the final form and it is not further polymerized as PET, but it remains as a small molecule additive. Of all DEHP produced globally, 95% is used as a plasticizer to render hard and brittle polymers more flexible and malleable (ATSDR 2002). Poly(vinyl chloride) (PVC; Recycling code #3) is the most important of these brittle polymers, where an estimated 80% of all plasticizers produced are used to plasticize PVC (Murphy 2001; Stevens 1999). Of these plasticizers, the most common is DEHP, which accounts for approximately 50% of all plasticizers used in PVC (Murphy 2001). Its production was estimated to be about 2 million metric tons per year in 2004 (AgPU 2006). Some typical applications of plasticized PVC include medical equipment such as hospital tubing and blood bags, food wrapping, wire and cable insulation, and automobile parts (AgPU 2006; Rahman and Brazel 2004).

Plasticization (i.e. the functional effect of plasticizers) is caused by various interactions between the small plasticizer and the polymer (Wypych 2012). The interaction of the polar carbonyl functionalities

in DEHP with the polar carbon-chloride bonds in the polymer chains of PVC makes the two compounds compatible, but it is countered by the lack of interaction between the non-polar parts of the DEHP. As a result, in the presence of plasticizers, the long PVC chains interact less with each other and the material is rendered more flexible and malleable (Wypych 2012). To achieve this, a large quantity of DEHP is usually needed. For instance, plasticized PVC can contain up to 40% of plasticizer by weight (ATSDR 2002). It is important to recognize that the plasticizer is not chemically bound to the PVC, resulting in the possibility of DEHP migrating within the material and reaching the surface of the blend, where it can ultimately leach out of the material (Kastner et al. 2012). Ultimately, due to this behaviour, DEHP has been labelled a ubiquitous environmental contaminant as far back as 27 years ago (Wams 1987).

DEHP is particularly well studied because of concerns about its wide range of toxic effects (Akingbemi et al. 2004; Akingbemi et al. 2001; Foster et al. 2001; Gazouli et al. 2002; Horn et al. 2004). The US EPA has expressed its concern about phthalates "because of their toxicity and the evidence of pervasive human and environmental exposure to them" (US EPA 2012). The breakdown pattern of DEHP is very important, because stable metabolites are produced, such as 2-ethyl hexanol, 2-ethyl hexanoic acid, and its monoester, mono (2-ethylhexyl) phthalate (MEHP), each of which have been shown to be more toxic than DEHP itself (Horn et al. 2004; Nalli et al. 2006a; Nalli et al. 2006b; Nalli et al. 2006c). Special attention has been given to MEHP, as it is believed to be an endocrine disruptor and has been linked to antiandrogenic activities in humans (Fan et al. 2010; Pant et al. 2008; Piche et al. 2012; Richburg and Boekelheide 1996; Swan et al. 2005). Such findings eventually led to its ban in applications such as children's toys in much of the Western world (CPSIA 2008; EU 2005; HPA 2010). An extensive overview of DEHP has been provided in reports from the U.S. Department of Health and Human Services (ATSDR 2002) and the European Union (EU 2008).

Bottled Water

Although DEHP is not used to make water bottles, DEHP has been detected in many samples from water bottles regardless of its material, including PET, PE and glass (Table 1). This is unexpected

and suggests other potential routes of contamination. The quantities of DEHP found in bottled water have been reported in the range of 0 to 1 µg/L, with the exception of two studies: Bošnir et al. (2007) found an average of 8.78 µg/L and Keresztes et al. (2013) reported values of up to 1.7 µg/L. The variability in the concentrations of DEHP found in the different bottled waters shown in Table 1 could possibly be due to regional differences in the prevalence of contaminants such as DEHP.

Some of the studies shown in Table 1 looked directly at the content of DEHP of the water stored in PET, PE and glass containers. Given that DEHP is not used in the manufacturing of these bottles, the type of material of the bottle is obviously not as important as the source of the water. The water bottles were bought in local stores, so that a certain shelf life already had passed, and the DEHP content of the water was analyzed immediately thereafter (Bošnir et al. 2007; Cao 2008; Martine et al. 2013). Other studies involved the purchase of the bottles in local stores, but then monitored the concentration of DEHP in the water over time. Overall, the results have been very variable from study to study and no clear trend can be seen.

When examining the levels of DEHP in the bottled water over time, some studies report increasing concentrations (Casajuana and Lacorte 2003; Guart et al. 2014; Keresztes et al. 2013; Leivadara et al. 2008) and others report constant or even decreasing levels of DEHP (Al-Saleh et al. 2011; Diana and Dimitra 2011; Guart et al. 2014; Keresztes et al. 2013). A decrease of DEHP concentration would mean that some kind of breakdown had occurred, which seems very unlikely. Even within individual studies, both increases and decreases of DEHP concentration have been reported over time (Guart et al. 2014; Keresztes et al. 2013).

Some of the studies on bottled water also dealt with the effect of storage location and condition on DEHP concentrations in bottled water. These parameters compared storage inside or outside a building, with the main difference being storage temperature and exposure to direct sun light. No clear or consistent trends regarding the effects of storage condition on DEHP concentration have been produced, with contradictory observations of increasing concentrations (Al-Saleh et al. 2011; Diana and Dimitra

2011; Keresztes et al. 2013) and decreasing concentrations of DEHP (Leivadara et al. 2008) having been reported.

Finally, some studies reported higher concentrations of DEHP in distilled water compared to carbonated water (Keresztes et al. 2013), while others reported an opposite trend (Leivadara et al. 2008; Martine et al. 2013).

From these highly variable results, it can be seen that there was no overall trend of leaching of DEHP from the water bottles to the water contained therein. Furthermore, no standard protocol have been established for these kinds of measurements and the experimental setups were quite different throughout the presented studies, making it difficult to compare studies and draw substantiated conclusions. Some studies also lack supporting statistical analyses.

It is noteworthy to mention that three of the studies presented in Table 1 also included glass bottles. The study by Leivadara et al. (2008) reported that the contents of two of ten glass bottles contained DEHP above the limit of quantification (LOQ), with values between 0.1 µg/L and 1.5 µg/L, depending on storage conditions of the carbonated water. Similarly, Cao (2008) reported DEHP in the carbonated water contained in three tested glass bottles, at levels between 0.15 µg/L and 0.24 µg/L. The third study involving glass bottles is a recent and vast study in Spain by Guart et al. (2014), which looked at a large number of different bottle materials. It found that water bottled in glass with a metal cap (including a liner inside the cap) was most likely to contain detectable levels of DEHP, with 39% of all samples after one year of storage (Guart et al. 2014) yielding positive results. The liner within the metal cap is a soft material and would thus likely contain a plasticizer which could be the source of contamination. This is discussed in greater detail in the following section. Overall, these results demonstrate that the bottle material is not the important factor, but that the DEHP was more likely in the water prior to bottling, or in the liner of the cap.

Potential sources of contamination with DEHP

DEHP has been a contaminant of concern for almost three decades (Wams 1987) and contamination, if not from the liner of the cap, could stem from any step in the production and bottling process (Diana and Dimitra 2011; Keresztes et al. 2013; Leivadara et al. 2008). To start with, the source of water could be contaminated as suggested by studies that revealed the presence of DEHP in rain water, surface waters, and tap water. For instance, several studies summarized in Table 2 showed that rain water samples contained DEHP in levels up to 39 µg/L (Berge et al. 2013; Björklund et al. 2009; Cole et al. 1984; Ligocki et al. 1985; Teil et al. 2006; Thuren and Larsson 1990; Vethaak et al. 2005). Such rain water will end up in the watershed and carry with it the DEHP contamination. Treated wastewaters from municipal or industrial sources that is released into rivers and streams would also add to this contamination burden (Barnabe et al. 2008; Beauchesne et al. 2008). Such contamination by DEHP has been found in surface water (Vethaak et al. 2005), water from public fountains (Casajuana and Lacorte 2003), and river or creek water and melted snow (Horn et al. 2004). A portion of this contamination can end up in treated drinking water such as tap water (Horn et al. 2004; Martine et al. 2013) used as the source of bottled water. Such water may also be a source of contamination during the washing of water bottles prior to their being filled. Also, the use of PVC piping to bring water to the bottling plant and subsequently to fill the bottle could also be a source of contamination, thus exposing the water to DEHP – although PVC pipes are a hard material and would consequently only contain low concentrations of plasticizer. Further studies would be needed to examine this, where it would need to be established whether the mass transport of DEHP from the pipes into the water carried in them would be significant enough to account for such contamination. Later in the process, bottles are capped with a lid with a liner. The liner must be a softer material and, therefore, DEHP contamination could stem from it, as was found by Guart et al. (2014) and in a study on bottled beer in China by Ye et al. (2009). The German Federal Institute for Risk Assessment released two advisory opinions stating that a substantial amount of DEHP can migrate from such liners of containers to the food contained in them (BfR Stellungnahme 10/2005 ; BfR Stellungnahme 25/2007).

Another important factor to consider is the possibility of DEHP contamination arising from the materials and solvents used in the laboratory conducting the analyses, especially during the preparation and analysis of the samples. Contamination from the use of plastic sampling bottles, caps with liners, filters, tubing *et cetera* must be accounted for during the sampling, preparation and analytical processes and eliminated, otherwise this would result in background levels of DEHP (Fankhauser-Noti and Grob 2007). The role that these factors play in sample contamination, especially for sensitive bio-assays, was demonstrated by Olivieri et al. (2012).

Food containers

Studies on DEHP found in food containers (Table 1) report a broad range of contamination, showing again the ubiquity of DEHP. DEHP concentrations measured in food are generally higher than in water. Again, glass bottles are not immune to DEHP contamination as revealed in studies where it has been reported in significant amounts in wine (Carrillo et al. 2007; Del Carlo et al. 2008; Russo et al. 2012). Overall, the results suggest contamination occurs during the preparation and filling of water, food-stuff or food at the source.

The concentrations of DEHP found in milk products that were stored in various containers (Sørensen 2006) were higher than those found in other liquids such as water, wine or beer placed in similar containers. This can be explained by the higher lipid content of the milk in which DEHP would have a higher solubility (ATSDR 2002). Part I of the study by Heinemeyer et al. (2012a) for the German Federal Environmental Agency (UBA) used literature data to show a correlation between increased DEHP content of milk and milk products with increasing fat content ($R^2 = 0.79$). Extensive reviews on phthalate esters in foods in Canada (Cao 2010) and Europe (Heinemeyer et al. 2012a; Heinemeyer et al. 2012b; Martine et al. 2013; Wormuth et al. 2006) are available and consistently report that the highest DEHP loads are found in the fattiest foods, such as citrus essential oils, fresh meat, fish terrine, chicken and mayonnaise (Cao 2010; Heinemeyer et al. 2012a; Heinemeyer et al. 2012b; Martine et al. 2013; Wormuth et al. 2006).

For the general population, the most important route of exposure to DEHP and phthalates is via foods, which has been reported to account for 80% (Heinemeyer et al. 2012a) or more than 90% (Wormuth et al. 2006) of the daily intake of DEHP in adults. The estimated levels reported, however, are below the tolerable daily intake (TDI) or reference dose (RfD), both based on non-cancer effects, used in various jurisdictions: i.e. 50 µg/kg of bodyweight (kg bw) in the EU (EFSA 2005), 44 µg/kg bw in Canada (Health Canada 1996), and 22 µg/kg bw in the USA (EPA 1997). An uncertainty factor, which is used to translate the TDI to a no-observed-adverse-effect level (ANOEL; i.e., the dose at which no adverse effect of DEHP is expected), is also provided by each governing body. The uncertainty factors is 100 in the case of the EU and 1000 for both Canada and the USA, and translates to ANOELs of 5 mg/kg bw, 44 mg/kg bw and 22 mg/kg bw for the EU, Canada and the USA, respectively. An overview of how these levels are established is provided by the US Food and Drug Administration (US FDA 2001). The USA also defines a specific maximum limit for DEHP for bottled water, which is set at 6 µg/L (US EPA 2013). When comparing exposure to DEHP through food and through bottled water specifically, levels found in food can be as much three orders of magnitude higher (Cao 2010; Martine et al. 2013).

Hospital equipment

Over the past 10 years, DEHP has been banned for use in children's toys throughout the Western world (CPSIA 2008; EU 2005; HPA 2010), yet it is still heavily used in hospital equipment due to its very low cost (Blass 2001). DEHP also seems to have a certain stabilizing effect on blood platelets stored in PVC/DEHP blood storage bags (Horowitz et al. 1985). This heavy use is an important consideration given the extensive amounts of one-time use materials that are consumed in hospitals. As such, Table 1 provides data on the leaching of DEHP from some commonly used hospital equipment, such as infusion sets (Bagel et al. 2011; Rose et al. 2012), tubing (Takehisa et al. 2005), PVC storage bags (Demore et al. 2002; Veiga et al. 2012), and PVC blood bags (Buchta et al. 2005; Inoue et al. 2005; Jaeger and Rubin 1972; Peck et al. 1979; Rock et al. 1978). Concentrations of DEHP in leachates from such equipment are significantly greater than what has been reported in water and foods, as was discussed above. These

differences range from three to six orders of magnitude higher than DEHP levels found in water and food, as shown in Table 1; for example, 50,000 – 70,000 µg/L in whole blood from blood bags (Inoue et al. 2005), 17,000 to 25,000 µg/L in aqueous solutions stored in PVC bags (Demore et al. 2002), and 1,700,000 to 3,100,000 µg/L in pure oils and 19,400 to 65,800 µg/L in lipid emulsions in infusion sets (Bagel et al. 2011). Consistent with what has been observed for food containers, the more lipophilic the solution used in the container, the higher the amount of DEHP observed in the content, mostly due to higher solubility of DEHP under such conditions (ATSDR 2002). Another study supporting this trend shows that in patients undergoing dialysis a higher total amount of leached DEHP was retained in the patients' blood with increasing plasma lipid (cholesterol and triglycerides) concentration of these patients (Faouzi et al. 1999).

Discussion

Based on the results presented above, it is clear that the major route of exposure of the general population to DEHP is via food. DEHP is present in bottled water, but bottled water does not pose a major threat of exposure to DEHP (Diana and Dimitra 2011; Fromme et al. 2007; Montuori et al. 2008; Schmid et al. 2008). Several studies show that the ingestion of DEHP from food is far greater than from any other source (Heinemeyer et al. 2012b) and, in the general population, can be as much as 1000 times higher than from water (Martine et al. 2013). Nonetheless, levels found even in food are below what is considered safe by the European Union and the US EPA (see section "Food containers" above). Given the omnipresent prevalence of DEHP in the environment, another way for human uptake is through a general environmental exposure (Bauer and Herrmann 1997; EU 2008; Guo et al. 2011; Martine et al. 2013; Staples et al. 1997; Wams 1987). Routes of exposure include, but are not limited to, house dust (Becker et al. 2004; Butte et al. 2001), indoor air (Butte and Heinzow 2002), soil (Cartwright et al. 2000), and watersheds (see Table 2). However, the study by (Heinemeyer et al. 2012b) suggests that these routes are minor compared to the uptake through food and only young infants are likely to be at increased risk through a combination of house dust and mouthing of toys.

The most intense degree of exposure of individuals to DEHP occurs in hospital patients. Table 1 shows a difference of three to six orders of magnitudes between the amounts of DEHP found in water and food samples compared to in liquids passed through hospital equipment. This is the only case where leaching can be observed directly from the container and linked to the specific hospital equipment. This is not surprising given that hospital equipment plastics such as tubing and IV bags are soft and frequently made using PVC and a plasticizer. However, the exposure experienced by an individual in a hospital environment is most likely to be of a short-term nature (i.e., acute exposure). Thus, it is difficult to compare to exposure through food and water, which would be more continuous and long-term over the life of the individual (i.e., chronic exposure). One also has to take into account the fact that DEHP is rapidly eliminated from the body, with between 65 and 70% of it being secreted in the first 24 hours after exposure (Koch et al. 2005). This means that the general population would likely not be at risk from infrequent hospital visits. However, people with certain medical conditions requiring regular treatment (i.e. neonates, dialysis patients, *et cetera*) would be at higher risk due to their more frequent exposure to high levels of DEHP.

From the data presented, it is clear that one cannot avoid contact with DEHP due to its omnipresence. However, it is be expected that regional differences in the amounts of DEHP present in the environment will result in different exposure risks. In fact, such a correlation has been observed for another contaminant, bisphenol A (BPA), which is a monomer used to make polyester and polycarbonate, hard plastics. Teuten et al. (2009) reported a correlation between the amount BPA in leachates from waste disposal sites with the Gross Domestic Product (GDP) of the country where this leachate was collected: i.e. the higher the GDP, the more BPA was found in the leachates (Teuten et al. 2009). BPA is a very different compound with different impacts altogether but, considering that production and use of plastics is expected to be different in developed countries producing more plastic waste, we can reasonably hypothesize that there is a higher risk of DEHP contamination in specific regions of the world. This is supported by a direct correlation between the industrial production of DEHP in Germany and the daily

intake of DEHP by German university students (Helm 2007). This also suggests that there are likely many more routes of exposure than we can account for at present.

Although most studies suggest that levels of DEHP due to chronic exposure are below what is considered safe by the several governmental agencies (see end of section "Food Containers"), our understanding of the toxic effects of DEHP are still evolving (Fan et al. 2010; Martinez-Arguelles et al. 2013; Piche et al. 2012). Yet, there are arguments for the replacement of DEHP. Most of the established levels for daily intake are based on estimations and weighted risk factors, rendering these estimation prone to error. On the other hand, the levels established by the governmental agencies are based mainly on extrapolation of data acquired in experiments with high concentrations of DEHP in animals. There are reports of non-monotonic dose responses of endocrine-disrupting compounds, meaning that an extrapolation of these results at high concentrations of DEHP is not necessarily valid (Vandenberg et al. 2012). This of course adds to the uncertainty of what safe levels might be. Furthermore, there is little work looking at synergistic effects DEHP might have with other contaminants or compounds found in the environment and the human body.

Despite this uncertainty, given the widespread use of DEHP in common materials and commercial products coupled with its established ubiquity in the environment, the accumulating evidence of negative impacts associated with this compounds – both on the environment and health – and especially given the multiple routes of exposure of humans to this compound through water, food, indoor air and other avenues, it is suggested that, as per the precautionary principle, the replacement of DEHP with less problematic compounds should strongly be considered. Under this principle, there is a duty to take anticipatory action to prevent harm and for government, industry and the general public to share in this responsibility. At present, there is a growing awareness of industry and government about the potential impacts of DEHP based on scientific findings. Furthermore, the general public is now expressing concerns over exposure to this compound and many are seeking ways to avoid such exposure in their day to day activities and purchases. Consumers also have a right to know about the potential impacts associated with the use of products and the burden to provide this information lies with the

producer. In the case of DEHP, exposure of the public comes via many routes, most of which are unknown to the average consumer and are involuntary. The knowledge is in the hands of the producer to act upon and limit these routes of exposure. It is also in their hands to demonstrate that their products are safe to use.

Another critical element of the precautionary principle is the importance of examining alternatives to the compound of concern and the selection of alternatives that have the least potential impact on human health and the environment. In many instances, the application of the principle is confounded by a lack of alternatives. However, in the case of DEHP, alternative materials are available and can be engineered to be less harmful. When evaluating potential alternatives to DEHP, it is essential that consideration be given to all foreseeable costs and life cycle, including raw materials, the manufacturing, transportation, use, environmental remediation and eventual disposal of created products, as well as health costs, even when such costs are not reflected in the price of the material/product itself.

Therefore, replacement compounds for DEHP must be well designed and thoroughly studied prior to widespread commercialization to avoid the types of problems observed with DEHP, such as its persistence in the environment and accumulation of toxic metabolites. DEHP has especially been shown to be rather resistant to biodegradation by microbes, likely due to the positioning of the two ester groups to one another, as well as the branching on the side chains (Erythropel et al. 2013; Erythropel et al. 2012; Gartshore et al. 2003; Nalli et al. 2002; Sauvageau et al. 2009). Similarly, its most problematic metabolite MEHP is also rather recalcitrant due to the same reasons as its parent compound (Amir et al. 2005; Erythropel et al. 2012; Jonsson et al. 2003).

To tackle this, recent research aims at producing equally effective plasticizers that are more biodegradable and less toxic than DEHP and would thus be eliminated more quickly in the environment, an idea that can also be extended to the metabolites of this plasticizer (Erythropel et al. 2013; Erythropel et al. 2012; Firlotte et al. 2009; Pour et al. 2009a; Pour et al. 2009b; Shi et al. 2011; Stuart et al. 2010). There also exist many commercial efforts to create plasticizers that are less toxic and recalcitrant, however formulations of these compounds are often unknown. Among the proposed alternatives are

vegetable-oil based plasticizers, citrate and succinate diesters, as well as hydrogenated DEHP-resembling compounds (DINCH®) (Lanxess AG 2011; Markarian 2010; Rahman and Brazel 2004). Much more testing is needed to validate these replacement compounds to not only ensure their utility as plasticizers but also to provide sufficient evidence that will either drive industry to adopt them or regulatory agencies to ban the use of DEHP to favour of these replacements.

Conclusion

Due to its widespread use and ubiquitous presence in the environment, DEHP ultimately finds its way into humans. It has been shown in this review that there is a broad consensus that this exposure is mainly due to ingestion through food and that bottled water in PET bottles only plays a very minor role in this exposure. In the latter case, it is highly unlikely that the contamination of the water stems from the container material itself, but it was more likely contaminated in an earlier production step or in the water source. However, there is one area of application in which the leaching of DEHP from a container to a liquid in contact with it does occur to a significant extent and that is in hospital equipment, which can result in a high acute exposure. This is especially problematic for those that need regular medical attention.

Although is it difficult to accurately estimate the total chronic exposure of the general population to DEHP, the general consensus seems to be that this chronic exposure is below what government agencies judge as safe. However, given the prevalence of DEHP as contaminant worldwide, the uncertainty concerning what is considered a safe level of exposure, the possibility of synergistic effects with other compounds, and the possibility to engineer more biodegradable plasticizers, we suggest, following the precautionary principle, that DEHP be replaced with more suitable compounds. This would result in reduced human exposure to DEHP. However, before this can be achieved, a replacement plasticizer must be designed very carefully and tested thoroughly to ensure that its implementation avoids effects such as those observed with DEHP.

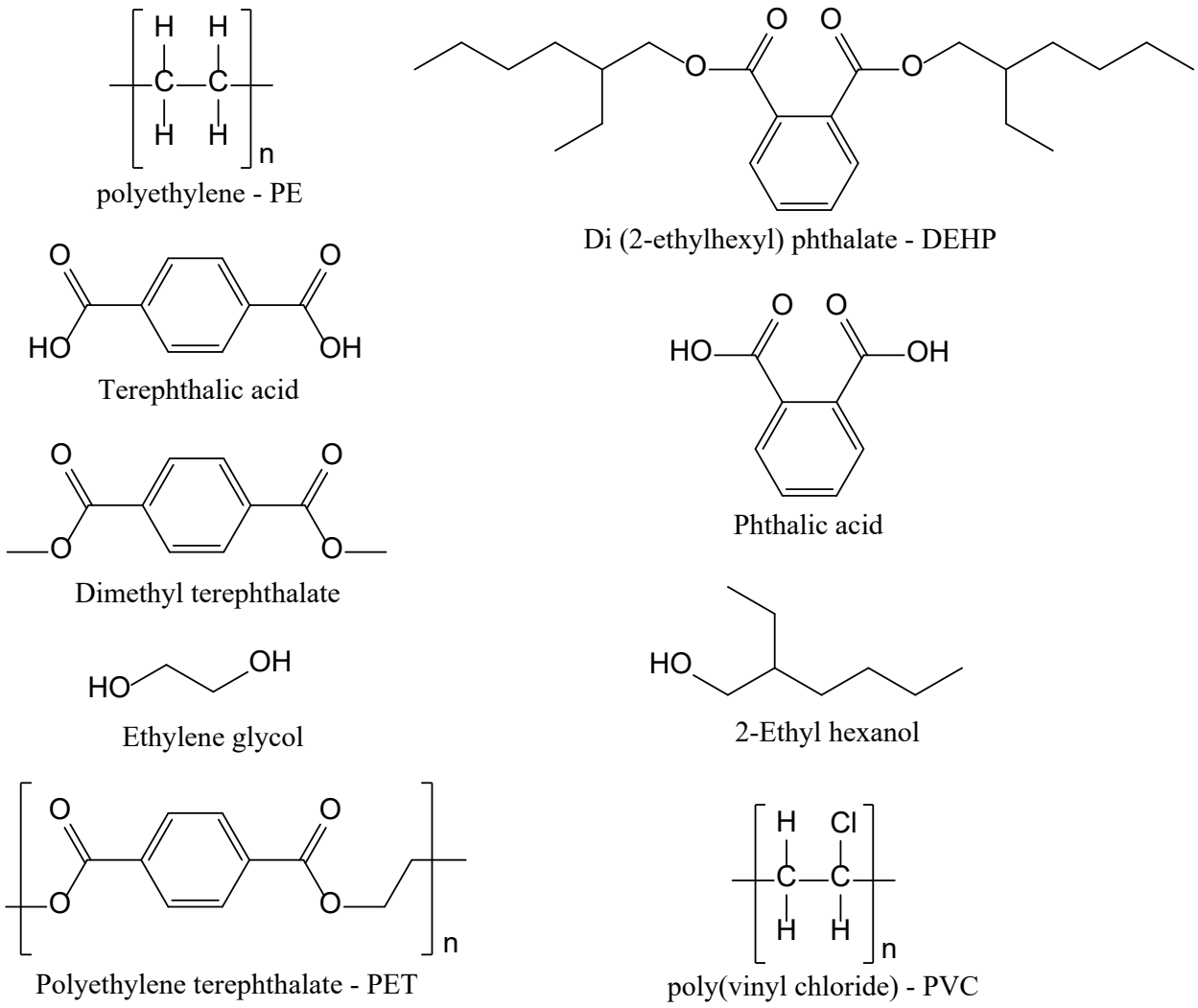

Fig.1 – Chemical structures of chemical compounds discussed.

Table 1: Quantities of DEHP found in various liquids stored in plastic and glass containers (LOQ = limit of quantification).

Type of samples	Concentration of DEHP (µg/L or µg/kg)	Number of replicates (n)	Country of origin	Source
Bottled water				
Water in PET bottle, non-carbonated	0.02 – 1.7	3	Hungary	(Keresztes et al. 2013)
Water in PET bottles, carbonated	< 0.02	3	Hungary	(Keresztes et al. 2013)
Water in recycled PET bottles	0.1 – 0.7	15	Honduras, Nepal, Switzerland	(Schmid et al. 2008)
Water in PET bottles	0.12 ± 0.1	7	Canada	(Cao 2008)
Water in polycarbonate bottle	0.22	1	Canada	(Cao 2008)
Water in glass bottle	0.17 ± 0.05	3	Canada	(Cao 2008)
Water in PET bottles, non-carbonated	0.125 ± 0.09	11	Paris, France	(Martine et al. 2013)
Water in PET bottles, carbonated	0.15 ± 0.07	4	Paris, France	(Martine et al. 2013)

Water in PET bottles	0.35 – 0.48	30	Thessaloniki, Greece	(Diana and Dimitra 2011)
Water in PET bottles	< 0.7 – 1.07	150	Riyadh, S-A	(Al-Saleh et al. 2011)
Water PET and glass bottles, carbonated	< 0.02 – 6.8	12	Mytilene, Greece	(Leivadara et al. 2008)
Water PET and glass bottles, non-carbonated	< 0.02 – 0.2	14	Mytilene, Greece	(Leivadara et al. 2008)
RO water stored in PET and glass bottles	< 0.02 – 0.06	5	Mytilene, Greece	(Leivadara et al. 2008)
Water in PET bottles	0 – 0.19	4	Catalunya, Spain	(Casajuana and Lacorte 2003)
Water in PE bottles	0.15 – 0.33	3	Catalunya, Spain	(Casajuana and Lacorte 2003)
Still and carbonated water in glass bottles with metallic crown	< LOQ – 11.9	170	Spain	(Guart et al. 2014)
Still and carbonated water in PET bottles	< LOQ – 13	448	Spain	(Guart et al. 2014)
Water in PET bottles	8.78	9	Croatia	(Bošnir et al. 2007)
Food-containers				
Wine in glass bottle	2.4 – 16	6	Italy	(Russo et al. 2012)

Wine in glass bottle	< 24 – 242	52	Italy	(Del Carlo et al. 2008)
Wine in PE box	25 - 276	10	Italy	(Del Carlo et al. 2008)
Wine in glass bottle	< 2	3	La Rioja, Spain	(Carrillo et al. 2007)
Beer in glass bottle	4.7	3	China	(Ye et al. 2009)
Soft drinks	18.9	36	Croatia	(Bošnir et al. 2007)
RO water in vegetable cans (121°C)	0 – 0.66	24	Mexico	(Gonzalez-Castro et al. 2011)
RO water in plastic containers (121°C)	0 – 0.23	22	Mexico	(Gonzalez-Castro et al. 2011)
Raw milk	Ca. 7-30	18	Denmark	(Sørensen 2006)
Pasteurized and homogenised milk	Ca. 13-27	4	Denmark	(Sørensen 2006)
Leaching from PVC tubing (Terufusion® IV set, 1m)				
Into a Tween80 solution (2mg/ml) – dynamic testing 66-1125 ml/h	100 – 1,000	3	Japan	(Takehisa et al. 2005)

Leaching from infusion sets (Mediplus Dual TIVA infusion sets, PVC/DEHP 2m)				
Into Diprivan© 1% solution – dynamic testing 12 ml/h	4,200	3	UK	(Rose et al. 2012)
Into Propoven© 1% solution – dynamic testing 12 ml/h	4,900	3	UK	(Rose et al. 2012)
Into Intralipid© 10% solution – dynamic testing 12 ml/h	7,600	3	UK	(Rose et al. 2012)
Leaching from infusion sets (Extension Set, Laboratoire Cair, PVC/DEHP/PE 1.5m)				
Into six lipid emulsions, all 20% Intralipid©, Medialipid©, Structolipid©, Lipidem©, Clinoleic©, Omegaven© - static	19,400 – 65,800	6	France	(Bagel et al. 2011)
Into pure oils (olive, soybean, coconut, cod liver) - static	1,700,000 – 3,100,000	6	France	(Bagel et al. 2011)
Leaching from 5 different brands of PVC bags for injections (France)				

Into 0.9% NaCl + 0.4mg/ml etoposide sol'n	17,000 – 25,000	8	France	(Demore et al. 2002)
Into 5% dextrose + 0.4mg/ml etoposide sol'n	17,000 – 25,000	8	France	(Demore et al. 2002)
Leaching from PVC bag (500ml) with aq. Solutions (Brazil, Baxter bags)				
Into 10% Glucose sol'n	1900	2	Brazil	(Veiga et al. 2012)
Into 0.5 % (m/v) amino acid sol'n (Leu, His, Thr)	800 – 900	2	Brazil	(Veiga et al. 2012)
Into 0.9% NaCl sol'n	300	2	Brazil	(Veiga et al. 2012)
Into purified water	100	2	Brazil	(Veiga et al. 2012)
Leaching from PVC/DEHP blood bags, containing CPD (citrate-phosphate-dextrose)				
Into human blood 4°C (Fenwal)	50,000 – 65,000	3	Baltimore, US	(Jaeger and Rubin 1972)
Into whole blood 4°C (Fenwal)	80,000 ± 10,000	3	Ottawa, Canada	(Rock et al. 1978)
Into whole blood 4°C (Fenwal, adenine-enriched CPD)	152,000 ± 4,500	4	US	(Peck et al. 1979)

Into whole blood 4°C (McGaw adenine-enriched CPD)	123,000 ± 22,400	4	US	(Peck et al. 1979)
Into whole blood	50,000 – 70,000	18	Japan	(Inoue et al. 2005)
Into pure plateletpheresis concentrate	2,090-10,670	5	Austria	(Buchta et al. 2005)
Into pure 35% plateletpheresis concentrate + 65% T-Sol (Baxter Healthcare)	500 – 3,250	4	Austria	(Buchta et al. 2005)
PVC/29 wt-% DEHP				
Into RO water	8 – 20	3	Canada	(Kastner et al. 2012)

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Table 2: Environmental concentrations of DEHP reported in various countries in different environmental compartments (LOQ = limit of quantification).

Type of samples	Concentration of DEHP (µg/L or µg/kg)	Number of replicates (n)	Country of origin	Source
In public fountains	0 – 0.33	7	Catalunya, Spain	(Casajuana and Lacorte 2003)
Tap water	0.06	3	Paris, France	(Martine et al. 2013)
Tap water	4.6	1	Montréal, Canada	(Horn et al. 2004)
Surface water	< 0.9 – 5	81	Netherlands	(Vethaak et al. 2005)
Rainwater	0.02 – 0.1	7	Portland, OR, USA	(Ligocki et al. 1985)
Rainwater	0 – 0.43	56	Sweden	(Thuren and Larsson 1990)
Rainwater	0.69 – 1.7	3	Amsterdam, Netherlands	(Vethaak et al. 2005)
Rain water	0.42	72	Paris, France	(Teil et al. 2006)
Rain water	< LOQ – 3.25	92	France	(Dargnat 2008)
Storm water	< LOQ - 39	86	USA	(Cole et al. 1984)

Storm water	< 1 – 5	39	Stockholm, Sweden Göteborg, Sweden	(Björklund et al. 2009)
River water	180	1	Montréal, Canada	(Horn et al. 2004)
Melted snow 200 ml samples transferred to 10L bottle. All same area though… 1 or 50 samples?	130	1	Montréal, Canada	(Horn et al. 2004)
Creek water	47	1	Montréal, Canada	(Horn et al. 2004)
Landfill leachate	62	1	Montréal, Canada	(Horn et al. 2004)
Wastewater treatment plant influent 3.5L collected in 1h, 2 different influents	70	2	Montréal, Canada	(Barnabe et al. 2008)
Wastewater treatment plant effluent 3.5L collected in 1h, 2 different influents	54	1	Montréal, Canada	(Barnabe et al. 2008)

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