HOW CAN WE MANAGE THE MOUNTAIN OF PBDEs PLUS “NEW” FLAME RETARDANTS? PBDE INVENTORY, FATE AND POLICY ANALYSIS

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Introduction

The United Nations Environmental Programme’s “Global Chemicals Outlook” has described that the global economy is undergoing chemical “intensification” as a result of the increasing use of synthetic substances as replacements for natural materials, the use of increasingly complex chemicals, and the use of chemicals in an increasing number of applications[1]. Flame retardant chemicals (FRs) provide an excellent and disturbing example of such chemical “intensification”. Ceresana Market Research (2011) estimated the global market for FRs at 1.9 million tonnes and valued at 5.1 billion USD in 2012, with expected growth by 2017 to 2.6 million tonnes valued at 6.2 billion USD[2].

According to thermodynamics and supported by many empirical studies, some fraction of the tonnage of semi-volatile FRs added to materials have been and will continued to be emitted to the environment from the production mass that has been added to products and materials. Csiszar et al. (2013) estimated that ~0.01% of the inventory of penta- and octaBDEs contained in in-use products was emitted in late 2000’s to early 2010’s to the air overlying Toronto Canada[3]. Emissions from products contravenes the intent of the Stockholm Convention of reducing and/or eliminating emissions and discharges of penta- and octaBDEs, which were classified as persistent organic pollutants (POPs) because of their risk to human and ecological health.

Our goal was to estimate the magnitude of the dynamic inventory of PBDEs in use and waste phases and a discrete dynamic system flow analysis of PBDEs in use and waste phases in US and Canada. We then discuss the implications of results in terms of emissions and chemical management policy.

Materials and methods

A dynamic inventory of penta-, octa- and decaBDE was compiled by assuming a geographic system boundary of US and Canada and a temporal boundary of 1970 to 2020. A “bottom up” approach was used for PBDEs in polyurethane foam (PUF) and other PUF products, electrical and electronic equipment and vehicles. This entailed assembling time-dependent consumption data and assuming that the lifespans of the products in the use phase followed a Weibull distribution that defines the fraction of each product leaving the use phase at a certain time[4]. The mass of electronic products (e.g., cathode ray tubes (CRTs) and flat screen monitors, computers, laptops, hard copy devices, TVs, wires and cables) entering the use phase was obtained from consumption patterns from 1970 to 1980 and sales data from 1980 to 2011. The same rate of change in product consumption from 2007 to 2011 was assumed after 2011. The mass of PUF was estimated from production data. For vehicles we used the annual number of registered vehicles including passenger cars and light duty trucks. A “top down” approach was used for textiles and fabrics since we did not have sales data. The products considered included curtains in public spaces, upholstery material and military tents. We used an estimate of the total amount of PBDEs that was used in the textile industry from 1973 to 2013 which was fitted to a Weibull distribution to obtain annual PBDE use in these products.

The mass of penta-, octa- and decaBDE was calculated from the mass of each product and the PBDE content of that product[5]. The discrete dynamic system flow analysis (SFA) was based on the material flow analysis developed by Morf et al. (2005)[6].

Results and discussion

We estimated that the peak inventories of penta-, octa- and decaBDE were 45 000, 20 000 and 250 000 tonnes in 2004, 2004 and 2013, respectively (Figure 1). The rate of increase of predominantly decaBDE during the 1990s and 2000s was 4-8% per year.

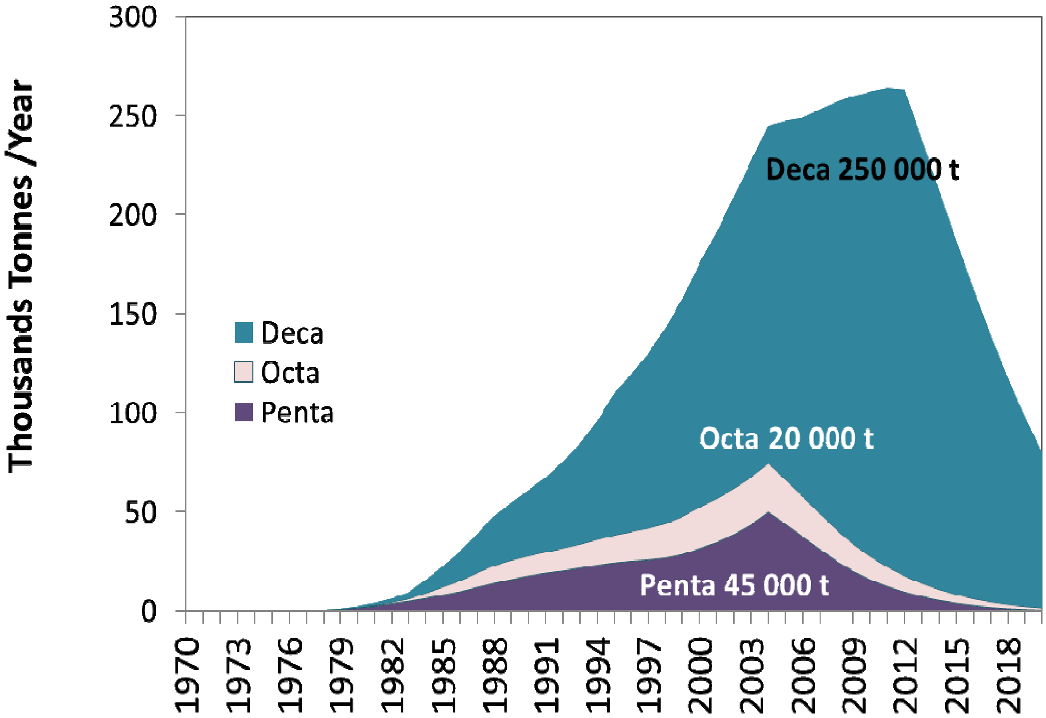

Figure 1. Inventory of PBDEs from 1970 to 2020 in Canada and the U.S. contained in polyurethane foam furniture and other products, electrical and electronic equipment, textiles and vehicles.

The greatest usage of penta was in foam furniture, of octa was in vehicles, and for decaBDE vehicles, textiles, and electronics.

Most penta- and octa-containing products will cease to enter the waste stream by 2020, but the mass of deca-containing products entering the waste stream will continue to increase after 2020 (Figure 2). These estimates represent an optimistic scenario since they are based on single sale life spans (except of vehicles) and hence neglect the life spans of resale or reuse as “re-used” products where re-use is common with, for example, furniture.

Over 90% of foam furniture (and hence most of penta) was estimated to be landfilled in North America by 2020. In comparison, 45% of deca in electrical and electronic equipment was expected to be exported off-shore as “used” equipment destined for dismantling and disposal (Fig. 2b). The remaining 30 and 17% of deca in electrical and electronic equipment, respectively, was estimated to be landfilled or processed at recycling facilities in North America. Incineration accounted for a negligible mass of end-of-life products.

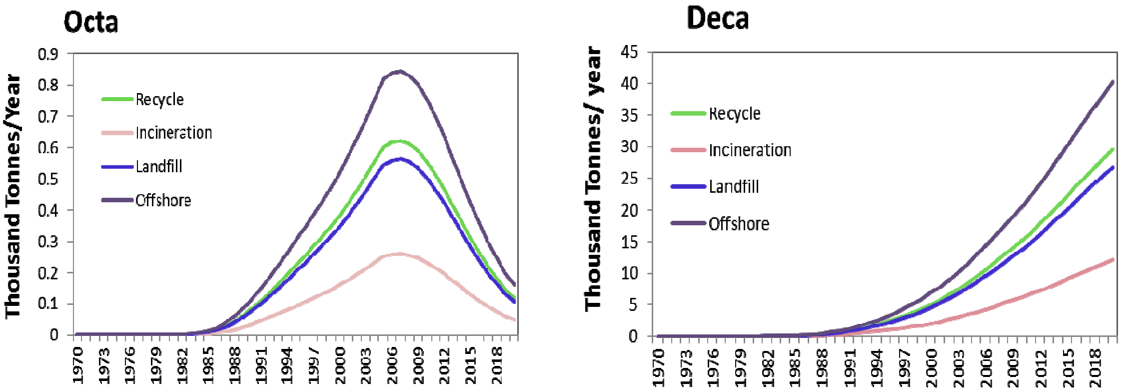

Figure 2. Estimated mass (tonnes/year) of (a) octa- and (b) decaBDE in products from Canada and the U.S. moving through the waste phase.

These results have important implications for human and environmental health and for future policy efforts. First, despite bans or voluntary phase outs of new production and new uses of penta and octaBDEs that have been promulgated over the past decade in Europe and North America, the substantial mass of PBDEs that remains in-use will continue to be a source of emissions and hence exposure for at least the next decade. The duration of emissions will depend on product life span which is short for electronics and lengthy for furniture and electrical wiring. Moreover, the long-term releases from the waste phase (e.g., due to landfill fires or leaks) are largely unknown.

Second, most legislation governing PBDEs does not contain provisions for the end-of-life of products containing these mixtures. As noted above, over 90% of PUF from furniture or 1700 tonnes/yr is landfilled in US and Canada with minimal flows to recycling or incineration. In contrast, large masses of PBDEs contained in waste electronic and electrical equipment continue to move from wealthy to developing countries as “used” equipment (e.g., Nnorom and Osibanjo 2008). This transboundary movement of these POPs circumvents the Stockholm and Basil Conventions and has the effect of contaminating receiving locations. Thus, PBDEs achieve global distribution by means of inadvertent long range atmospheric transport and intentional waste product transport.

Third, efforts to manage chemicals after they have penetrated the market are an expensive and very inefficient means of protecting human and ecological health. Chemical management schemes do not typically include provisions for product end-of-life and the cost of implementing such controls would be prohibitive. Despite this knowledge, flammability standards in the U.S., Canada and U.K. ensure that inventories of “new” flame retardants, some of which are also proving to be problematic, have and will continue to grow to take the place of PBDEs. The flammability standard setting process, at least in Canada, does not accommodate full disclosure and inclusion of stakeholders nor does it incorporate a benefit-risk analysis (e.g., benefits of fire prevention vs. risks associated with exposure to toxic chemicals). The result is that environmental scientists and health specialists tend to work at odds with fire scientists and standard setters, while those involved in chemical management are tasked with addressing chemical “safety” in a reactive rather than proactive fashion.

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