LIVSMEDELSVERKETS SAMARBETSRAPPORT

S 2020 nr 01

Contaminants in blood and urine from adolescents in Sweden

Results from the national dietary survey *Riksmaten Adolescents 2016–17*

NATURVÅRDSVERKET

Livsmedelsverket

Denna titel kan laddas ner från: Livsmedelsverkets webbsida för att beställa eller ladda ner material.

Författare:
Livsmedelsverket, Naturvårdsverket.

Rekommenderad citering:
Livsmedelsverket, Naturvårdsverket. S 2020 nr 01: Contaminants in blood and urine from adolescents in Sweden. Livsmedelsverkets samarbetsrapport. Uppsala.

S 2020 nr 01
ISSN 1104-7089

Omslag: Livsmedelsverket

Preface

The present report summarises the results from analysis of contaminants in blood and urine samples from participants in the dietary survey *Riksmaten Adolescents 2016–17*. These biomonitoring data provide unique information on total exposure to contaminants from all sources, including food, in Swedish adolescents. The results will be used further in risk assessments of contaminants in food by the Swedish Food Agency (Livsmedelsverket). Data from the project is also part of the national health-related environmental monitoring at the Swedish Environmental Protection Agency (Naturvårdsverket). The aim of this monitoring is to estimate human exposure to hazardous substances, follow temporal trends in human exposure, and to link environmental exposure to effects on health. The results from this report may also be useful for experts working with risk assessment and risk management in other organizations at the national or regional level.

The dietary survey *Riksmaten Adolescents 2016–17* was carried out by the Swedish Food Agency. The analysis of contaminants and the writing of this report were mainly financed by the Swedish Environmental Protection Agency (agreement numbers 2215-17-007, 2215-17-017 and 2215-18-010) and by the Swedish Food Agency. Financial support was also provided by the Swedish Civil Contingencies Agency (MSB) and Sahlgrenska University Hospital (Occupational and Environmental Medicine), Gothenburg.

Galia Zamaratskaia, Sanna Lignell and Helena Bjermo were responsible for the statistical analyses and the writing of this report. A number of experts at the Swedish Food Agency have contributed with expert advice and critical review of the report: Anna Karin Lindroos, Barbro Kollander, Irina Gyllenhammar, Emma Halldin Ankarberg, Tatiana Cantillana, Anneli Widenfalk, Lotta Moraeus and Cecilia Nälsén. A special thanks to all colleagues who were involved in the planning and implementation of *Riksmaten Adolescents 2016–17*, including handling of samples and chemical analyses. Thanks also to the regional divisions of Occupational and Environmental Medicine (AMM) who were responsible for the collection of human samples. Responsible for the chemical analyses were Barbro Kollander and Tatiana Cantillana (the Swedish Food Agency), Jon Benskin (Department of Environmental Science and Analytical Chemistry, Stockholm University), Christian H Lindh and Thomas Lundh (Division of Occupational and Environmental Medicine, Lund University) and Hannu Kiviranta (National Institute for Health and Welfare, Finland).

Livsmedelsverket / The Swedish Food Agency

Per Bergman
Avdelningschef, Risk- och nyttovärderingsavdelningen / Head of the Risk and Benefit Assessment Department

March 2020

Content

Abbreviations

AI	adequate intake
AMM	Occupational and Environmental Medicine (Arbets- och miljömedicin)
ANOVA	analysis of variance
BFR	brominated flame retardant
BMDL	benchmark dose lower bound
bw	body weight
CI	confidence interval
CV	coefficient of variations
EFSA	European Food Safety Authority
IARC	International Agency for Research on Cancer
IOTF	International Obesity Task Force
LOD	limit of detection
LOQ	limit of quantification
NHANES	National Health and Nutrition Examination Survey (U.S.)
NOAEL	no observable adverse effect level
PAH	polycyclic aromatic hydrocarbon
PFAS	per- and polyfluoroalkyl substance
POP	persistent organic pollutant
PVC	polyvinyl chloride
QC	quality control
RSD	relative standard deviation
SCB	Statistics Sweden (Statistiska Centralbyrån)
SD	standard deviation
SFA	Swedish Food Agency (Livsmedelsverket)
TEQ	toxic equivalent
TDI	tolerable daily intake
TWI	tolerable weekly intake
WHO	World Health Organization

Sammanfattning

I denna rapport sammanfattar vi biomonitoreringen av kemiska föroreningar i blod och urin från deltagarna i matvaneundersökningen *Riksmaten ungdom 2016–17*. Data från matvaneundersökningar som denna utgör en viktig vetenskaplig grund för Livsmedelsverkets risk- och nyttovärderingar. De är också viktiga när vi arbetar med riskhantering som kostråd, livsmedelskontroll och livsmedelslagstiftning.

Riksmaten ungdom 2016–17 är en nationell tvärsnittsstudie där Livsmedelsverket undersökt matvanorna hos barn och ungdomar i årskurserna 5 och 8 samt andra året på gymnasiet (dvs. i åldrarna 11–12, 14–15 respektive 17–18 år). Vi bjöd in representativa skolor över hela Sverige för att delta. Samtliga elever i en till två klasser vid de deltagande skolorna tillfrågades. I den här rapporten ingår den delgrupp av deltagare i *Riksmaten ungdom 2016–17* där blod- och urinprover samlades in. Vi rekryterade elever från skolorna mellan september 2016 och maj 2017. Av de 259 skolor som blev inbjudna för blod- och urinprovtagning deltog 62 (24 procent). Vid dessa skolor deltog 1 305 av de 2 377 tillfrågade eleverna, varav 1 105 elever (46 procent) både gav fullständig information om matvanor och lämnade blod- och urinprover. 56 procent av deltagarna var flickor. 30 procent gick i årskurs 5, 37 procent i årskurs 8 och 32 procent i årskurs 2 på gymnasiet. Skolorna fanns i regionerna för Sveriges sju arbets- och miljömedicinska enheter (Göteborg, Linköping, Lund, Stockholm, Umeå, Uppsala och Örebro).

I de blod- och urinprover som vi samlade in undersöktes ett brett spektrum av kemiska föroreningar: ämnen som inte längre är godkända för användning men ännu finns i miljön, ämnen som fortfarande produceras och används samt naturligt förekommande metaller. Vi analyserade följande grupper av substanser: klorerade och bromerade persistenta organiska föroreningar, högfluorerade ämnen, metaller, ftalatmetaboliter och flera fenolära ämnen (bl.a. bisfenoler och metaboliter av några bekämpningsmedel).

Analysen av klorerade organiska föroreningar i serum visade att halterna var högst av DDT-metaboliten DDE, följt av PCB-153, hexaklorbensen (HCB), PCB-138 och PCB-180. Av den totala halten PCB utgjorde PCB-153, PCB-138 och PCB-180 ca 70 procent. Pojkarna i undersökningen hade högre halter än flickorna av de flesta klorerade föreningar. Halterna av DDE och vissa typer av PCB var dessutom högre bland de äldre deltagarna.

Av de högfluorerade ämnena uppmätte vi högst serumhalter av PFOS och PFOA. De högsta halterna fanns hos ungdomar från Ronneby kommun (region Lund), där dricksvattnet tidigare varit kraftigt förorenat av dessa ämnen. Ämnena PFNA, PFHxS och PFOS fanns i högre halter hos pojkar än hos flickor.

Nästan alla deltagare hade mätbara halter i blodet av kadmium, kvicksilver, bly, krom, mangan, kobolt och nickel. Aluminium kunde mätas i serum hos hälften av deltagarna. Hos pojkar fanns högre halter av kvicksilver och bly, medan halterna av kadmium var högre hos flickor.

Hos 123 av deltagarna undersökte vi om det fanns arsenik i urinen. Alla prover hade mätbara halter av oorganisk arsenik eller minst en av de två analyserade organiska metaboliterna (monometylarsenik och

dimetylarsenik). Högst var halten av dimetylarsenik. Pojkar hade högre totalhalter av arsenik än flickor.

Vi analyserade även en rad ftalatmetaboliter och fenolära ämnen i urin från deltagarna. Nästan alla (94–100 procent) hade mätbara halter av ftalatmetaboliter och metaboliter till DiNCH (mjukgörare som används som ersättare till ftalater) i urinen. Högst halter fanns av monobutylftalat (MnBP), följt av monoetylftalat (MEP). Fenolära ämnen som kunde mätas i de flesta urinprover (mer än 90 procent) var bisfenol A och S, difenylfosfat (DPP, metabolit till ett fosforbaserat flamskyddsmedel), trikloropyridinol (metabolit till insekticiden klorpyrifos) och 3-fenoxybensoesyra (metabolit till pyretroider, en grupp bekämpningsmedel). Triklosan var mätbart hos 81 procent av deltagarna. Halterna av ftalatmetaboliter, DPP och bisfenol S var något högre hos flickor än hos pojkar. De ftalatmetaboliter och fenolära ämnen som undersöktes är kortlivade och försvinner snabbt ur kroppen. Att de ändå kunde mätas i urin tyder på en kontinuerlig exponering.

Sammanfattningsvis kunde vi mäta föroreningar ur alla undersökta substansgrupper hos de flesta deltagare i *Riksmaten ungdom 2016–17*. Nivåerna var generellt jämförbara med nivåerna i andra studier och inom de intervall som kunde förväntas. Vi såg vissa köns- och åldersrelaterade skillnader i halter. Detta tyder på att det kan finnas skillnader i exponering, upptag eller eliminering mellan kön och åldersgrupper. Deltagarna var dock i olika tillväxtfaser och stadier av puberteten.

Att göra en fullständig värdering av riskerna med de halter som uppmätts ligger utanför syftet med denna rapport. För några substanser finns det dock föreslagna halter i människokroppen under vilka det är osannolikt med negativa hälsoeffekter. Detta gäller vissa PCB-varianter, PFOS, PFOA, kvicksilver, bly, några ftalat- och DiNCH-metaboliter, bisfenol A samt triklosan. Våra resultat visar att de flesta svenska ungdomar har halter i sina kroppar som utifrån nuvarande kunskap sannolikt inte utgör någon risk för hälsan. Vissa individer hade emellertid högre halter av PFOS eller bly. Detta gällde framför allt bly. Där hade 7 procent av deltagarna blodhalter över Efsas referenspunkt för ökad risk för kronisk njursjukdom hos vuxna och 13 procent hade halter över referenspunkten för påverkan på hjärnans utveckling hos foster och små barn. Detta understryker vikten av att ytterligare minska exponeringen för bly från alla källor. Vi kan heller inte utesluta att det finns grupper i Sverige med högre exponering för kemiska föroreningar än deltagarna i *Riksmaten ungdom 2016–17*.

Halterna av föroreningar varierade mycket mellan de som deltog i undersökningen. För att kunna bedöma och hantera risker är det viktigt att undersöka möjliga orsaker till dessa variationer. Därför kommer Livsmedelsverket att fortsätta utvärdera data från *Riksmaten ungdom 2016–17.* Detta kommer vi att göra genom att studera samband mellan halter av kemiska föroreningar och faktorer som kost, sociodemografi och livsstil.

Summary

Dietary and biomonitoring data constitute an important scientific basis for risk and benefit assessments as well as for the development of risk management measures such as dietary advice, control programmes and food regulations. The present report summarises the results from biomonitoring of contaminants in blood and urine from participants in the dietary survey Riksmaten ungdom 2016–17 (*Riksmaten Adolescents 2016–17*).

Riksmaten Adolescents 2016–17 is a nationally representative, cross-sectional, school-based dietary survey of children and adolescents in grades 5 and 8, and high school grade 2 (approximately 11–12, 14–15 and 17–18 years of age). Representative schools across Sweden were invited to take part in the study. At the participating schools, all students in one or two classes were invited. The study population of the present report was a subgroup of *Riksmaten Adolescents 2016–17*, from whom blood and urine were collected. Adolescents were recruited from schools between September 2016 and May 2017. Sixty-two (24%) of the 259 schools invited for blood and urine sampling participated. At these schools, 1,305 of the 2,377 invited students participated. Complete dietary information and valid blood and urine samples were available from 1,105 students (46%) and these were included in the analyses in this report. Fifty-six percent of the participants were girls. The distribution of participants between grades were as follows; grade 5: 30%, grade 8: 37%, and high school grade 2: 32%. The participating schools were distributed across Sweden's seven Divisions of Occupational and Environmental Medicine (Gothenburg, Linköping, Lund, Stockholm, Umeå, Uppsala, and Örebro regions).

A wide range of contaminants were investigated in the collected blood and urine samples: substances the use of which has been restricted or banned but which continue to persist in the environment; substances that continue to be legally produced and used; and naturally occurring toxic metals and trace elements. The analysed substance groups included chlorinated and brominated persistent organic pollutants, per- and polyfluoroalkyl substances (PFAS), metals and metalloids, phthalate metabolites and phenolic substances (e.g. bisphenols and metabolites of some pesticides).

The chlorinated persistent organic pollutant with the highest concentration in serum was the DDT metabolite DDE, followed by PCB-153, hexachlorobenzene (HCB), PCB-138 and PCB-180. These three PCBs accounted for 70% of the total body burden of PCBs. Boys had higher serum concentrations of most chlorinated persistent organic pollutants than girls. Concentrations of DDE and some of the PCBs were higher in older age groups.

PFOS and PFOA were the PFAS detected in the highest concentrations in serum. The highest concentrations of PFAS were found in individuals from Ronneby municipality (region Lund) where previously the drinking water has been contaminated with these compounds. Higher concentrations of PFNA, PFHxS and PFOS were observed in boys than girls.

Almost all participants had detectable concentrations of cadmium, mercury, lead, chromium, manganese, cobalt and nickel in whole blood. Aluminium was detected in serum from half of the participants. Higher concentrations of mercury and lead were observed in boys than girls whereas blood concentrations of cadmium were higher in girls than boys.

Urinary arsenic was measured in a subsample of 123 participants. All participants had quantifiable urinary levels of inorganic arsenic or at least one of its two analysed metabolites (monomethylarsonic acid and dimethylarsinic acid). The highest concentration was observed for dimethylarsinic acid. Boys had higher urine concentrations of total arsenic than girls.

A number of phthalate metabolites and phenolic substances were analysed in urine from the participants. Almost all samples (94–100%) had measurable concentrations of phthalate metabolites and metabolites of the alternative plasticizer DiNCH. Of the phthalate metabolites, the highest concentrations were observed for mono-butyl phthalate (MnBP), followed by monoethyl phthalate (MEP). The phenolic substances detected in most samples (>90%) were bisphenol A and S, diphenyl phosphate (DPP, metabolite of a phosphorus-based flame retardant), trichloropyridinol (metabolite of the insecticide chlorpyrifos) and 3-phenoxybenzoic acid (metabolite of pyrethroids, a group of pesticides). Triclosan was detected in 81% of the samples. Concentrations of phthalate metabolites, DPP and bisphenol S were higher in girls than in boys. All the metabolites and phenolic substances analysed in urine are rapidly metabolised and excreted from the body. The detection of these substances thus suggests continuous exposure.

In summary, contaminants from all investigated substance groups (i.e. chlorinated persistent organic pollutants, PFAS, metals and metalloids, and phthalates and phenolic compounds) could be quantified in samples from most participants in *Riksmaten Adolescents 2016–17*. However, the levels were generally comparable to levels found in other studies and within the expected ranges. Despite the heterogeneity of the study population, which represented a mix of individuals at different pubertal stages and growth phases, we found some gender- and age-related differences in contaminant concentrations. This suggests that there may be differences in exposure, uptake and/or elimination between genders and age groups.

Full risk assessments of the observed concentrations were beyond the scope of this report. However, for some PCBs, PFOS, PFOA, mercury, lead, some phthalate and DiNCH metabolites, bisphenol A and triclosan, concentrations in the human body have been proposed below which it is unlikely that these contaminants cause adverse health effects. Our results show that, based on current knowledge, the levels of these contaminants in most Swedish adolescents are not a health concern. However, some participants exhibited a higher exposure to PFOS or lead. This was especially pronounced for lead, where 7% and 13% of the participants had blood concentrations above the EFSA reference points for increased risk of chronic kidney disease in adults and developmental neurotoxicity in small children, respectively. This underlines the importance of further reducing lead exposure from all sources, not only food. Moreover, we cannot exclude that there are Swedish populations with a higher exposure to contaminants than the populations covered by *Riksmaten Adolescents 2016–17*.

There were substantial individual variations in the levels of the investigated contaminants among Swedish adolescents. For risk assessment and risk management purposes, it is important to explore possible causes of this variation. Therefore, the Swedish Food Agency will continue to evaluate data from *Riksmaten Adolescents 2016–17* through further studies on the association between contaminant levels and factors such as diet, sociodemographics and lifestyle.

Introduction

Dietary data constitute an important scientific basis for risk and benefit assessments as well as for the development of risk management measures such as dietary advice, control programmes and food regulations. The Swedish Food Agency therefore regularly carries out national dietary surveys to gather information on dietary intake. The collected data are used to calculate nutrient and energy intake as well as to estimate exposure to unwanted substances via food, for example. In the most recent dietary surveys conducted by the Swedish Food Agency, *Riksmaten Adults 2010–11* and *Riksmaten Adolescents 2016–17*, the collection of dietary data was supplemented by taking blood and urine samples in participant subgroups for analysis of markers for nutritional status and contaminants. Biomonitoring data are important for assessing nutritional status and human exposure to contaminants that could negatively affect health.

In this report, we summarise results from the biomonitoring of contaminants in blood and urine from participants in the dietary survey *Riksmaten Adolescents 2016–17*. Differences in concentrations of contaminants in adolescents in different geographical regions and age groups (school grades) and by gender are investigated. More in-depth statistical analyses of other factors that may influence exposure, e.g. food consumption, life-style and socioeconomic factors, will be performed later. The following substance groups were analysed in samples from the participants in *Riksmaten Adolescents 2016–17:* chlorinated and brominated persistent organic pollutants (POPs), per- and polyfluoroalkyl substances (PFAS), metals and metalloids, phthalate metabolites, and phenolic substances (e.g. bisphenols and metabolites of some pesticides).

To the best of our knowledge, *Riksmaten Adolescents 2016–17* is to date the largest nationally representative study on diet and body burden of contaminants in Swedish adolescents. The studied population is of particular interest because children and adolescents are more sensitive to environmental chemical exposure than adults. Exposure during early life may have unpredictable adverse health effects later in life [1]. For example, it has been shown that exposure to some chemicals during gestation, infancy or childhood may increase the risk of neurodevelopmental disorders and obesity [2]. Examples of such chemicals include polychlorinated biphenyls (PCBs), hexachlorobenzene (HCB), 1,1-bis-(4-chlorophenyl)-2,2,2-trichlorethane (DDT), bisphenol A (BPA), phthalates, triclosan, PFAS and heavy metals [2-4].

Chlorinated and brominated persistent organic pollutants

Persistent organic pollutants (POPs) are a group of halogenated compounds that accumulate in the environment and the human body because of their lipid solubility and resistance to degradation. They have been intentionally produced for a variety of purposes or have been formed inadvertently during certain chemical processes.

Examples of chlorinated and brominated POPs include industrial chemicals (e.g. PCBs), compounds unintentionally formed during industrial processes (e.g. dioxins), chlorinated pesticides (e.g. DDT) and brominated flame retardants (BFRs). Many of these chlorinated and brominated POPs cause a number of adverse effects in animals, including effects on reproduction and development and the endocrine, nervous and immune systems [5, 6]. Similar, but more subtle effects have been indicated in

epidemiological studies with background exposure [7-9]. Early life stages, i.e. the fetus, infant and child, are most sensitive to exposure [10]. Exposure to PCBs, HCB and the DDT metabolite p,p'-DDE (1,1-dichloro-2,2-bis(4-chlorophenyl)ethane) has also been associated with an increased risk of obesity [11], type 2 diabetes [12], hypertension [13, 14] and all-cause mortality [15]. Exposure to chlorinated and brominated POPs is therefore of great concern, especially during adolescence, a period of growth and development. Many of these substances are listed in the global Stockholm Convention; their use is strictly regulated, and some are no longer in production.

Overall, human biomonitoring studies have demonstrated decreasing levels of many chlorinated and brominated POPs. In Sweden, the levels of PCBs, dioxins and p,p'-DDE in breast milk have decreased by at least 70% since the 1970s [16]. In addition, a study of first-time mothers conducted by the Swedish Food Agency (POPUP, Persistent Organic Pollutants in Uppsala primiparas) shows decreasing levels of PCBs, dioxins, chlorinated pesticides and polybrominated diphenyl ethers (PBDEs) in breast milk and serum between 1996 and 2016 [17, 18]. Nøst et al. [19] have also reported a decrease in the levels of chlorinated pesticides and PCBs in 30-year-old Norwegian men and women from 1986 to 2007.

PCBs are a group of anthropogenic POPs that were produced for decades before they were banned in the 1970s and 1980s because of their harmful effect on the environment and human health [20]. However, because of their resistance to degradation, PCBs are still present in the environment. PCBs are lipophilic and easily bioconcentrated and biomagnified in the food chain. There are 209 different PCB congeners with different numbers and positions of the chlorine atoms and thus different biological activities. These congeners are divided into 12 dioxin-like and 197 non-dioxin-like PCBs.

Food, especially food of animal origin, is the main source of human exposure to PCBs [21, 22]. In Sweden, fatty fish from the Baltic Sea has been recognized for a long time as an important source of exposure to PCBs [23, 24]. Vegetables can also be a source of PCB exposure [25], but they mainly contribute to exposure to lower chlorinated congeners with shorter half-lives and faster metabolism. Some exposure may be derived from air [26] and from house dust [27].

The European Food Safety Authority (EFSA) has established a tolerable weekly intake (TWI) for dioxins and dioxin-like PCBs of 2 pg toxic equivalents/kg body weight (bw) [28]. The TWI value indicates the weekly dose that can be consumed over the course of a lifetime without adverse health effects in humans. The TWI for dioxins and dioxin-like PCBs was based on effects on semen quality in an epidemiological study. No health-based tolerable intake level has been established for non-dioxin-like PCBs. However, based on liver and thyroid effects, EFSA estimated that 500 µg/kg is a conservative NOAEL (no observed adverse effect level) body burden for total non-dioxin-like PCBs [29]. NOAEL is the highest level that does not cause an adverse health effect in the most sensitive species. The German Human Biomonitoring Commission has estimated that there is no risk of adverse health effects of non-dioxin-like PCBs at serum levels (sum of PCB 138, PCB 153 and PCB 180 multiplied by 2) at or below 3.5 µg/L and that adverse health effects are possible above 7 µg/L [30]. These values are based on benchmark doses from epidemiological studies of prenatal exposure and cognitive effects [31].

Chlorinated pesticides, e.g. DDT and chlordanes, have been widely used as insecticides. Additionally, DDT is used in malaria control. In the environment, DDT is mainly degraded to p,p'-DDE, which is more persistent than the parent compound. HCB has been widely used as a fungicide for control of mould and fungi in cereal grains. HCB is also formed unintentionally as a contaminant

in chemical and combustion processes [20]. The use of these pesticides has been banned in Sweden for decades. However, due to their stability, high volume production, long-time use and long-range atmospheric transport they continue to be spread in the environment and are found in both wildlife and humans [32]. According to the Swedish Market Basket Survey 2015, fish consumption is the largest contributor to the total intake of p,p'-DDE followed by dairy and meat products. For HCB, dairy products are the largest contributor to exposure, followed by fish and meat [22].

The World Health Organisation (WHO) has proposed a health-based guidance value for HCB intake of 160 ng/kg bw/day, based on animal studies on cancer [33]. For DDT compounds, a provisional acceptable daily intake of 10 µg/kg bw has been established based on developmental toxicity in rats [34]. Based on the Swedish Market Basket Survey 2015, the average per capita intake of HCB and DDT compounds in the Swedish population are 100-1,000 times lower than these guidance values [22].

Brominated flame retardants (BFRs) are a diverse group of chemicals that are used to increase fire resistance in various materials. Polybrominated diphenylethers (PBDEs) and HBCDD (hexabromocyclododecane) are additive BFRs that have been used since the 1970s in goods such as plastics, textiles and electronic products [35]. Both PBDEs and HBCDD are toxic to the liver and affect thyroid hormone homeostasis and the reproductive and nervous systems [36, 37]. Due to their toxic properties and persistence, the use of PBDEs and HBCDD has been regulated within the EU since the beginning of the 2000s. Fish, meat, fats, egg and dairy products are the major contributors of PBDEs and HBCDD in the diet [22]. According to EFSA, dietary intake of PBDEs and HBCDD is unlikely to be a significant health concern [36, 37]. Moreover, total per capita intake of PBDEs and HBCDD decreased during the period 1999-2015 [22]. Similarly, the concentration of many PBDEs and HBCDD decreased in serum from Swedish first-time mothers between 1996 and 2017 [18]. Many emerging BFRs have been introduced on the market as substitute chemicals for PBDEs and HBCDD, e.g. BTBPE (2-bis(2,4,6-tribromophenoxy) ethane), DBDPE (decabromodiphenylethane), PBEB (pentabromoethylbenzene) and HBB (hexabromobenzene). Knowledge on emerging BFRs is limited.

Per- and polyfluoroalkyl substances

Per- and polyfluoroalkyl substances (PFAS) comprise a large group of synthetic substances that have been produced and used in a wide variety of products and applications due to their unique chemistry such as surface activity and resistance to chemical and biological degradation [38, 39]. Some PFAS (perfluorooctane sulfonate [PFOS] and perfluorooctanoate [PFOA]) are highly resistant to both physical and microbiological degradation [40] and are persistent in the environment, thus fulfilling international POP criteria [41]. However, in contrast to the classical POPs described above, the surface-active perfluoroalkyl acids (PFAA) do not accumulate in fatty tissues, but are bound to proteins and accumulate in the blood, liver and kidney, and in some cases in the thymus, lung and bone marrow [42-44].

Resistance to degradation has led to the accumulation of some PFAS in the environment, which are now not only detectable in surface and drinking water but also in food and wildlife worldwide [45-47]. PFOS and PFOA have been shown to be toxic in animal studies, producing liver toxicity, developmental effects in offspring, immune suppression, adverse effects on lipid metabolism and hormonal effects [41, 48]. A recent systematic review of children's health literature indicates that PFAS exposure is associated with dyslipidaemia and renal and immunity-related disorders [49].

Another possible effect of PFOS is poor serum antibody response to vaccinations in children [41]. Some PFAS have been associated with metabolic disorders, although the number of human studies is insufficient to draw any reliable conclusion [50].

Food and contaminated drinking water are important sources of human exposure to PFAS in Sweden and other countries [51-54]. PFAS have already caused undesirable consequences for public drinking water supplies in Sweden [55]. There have been a number of cases of PFAS contamination of drinking water and some wells have been closed due to elevated PFAS levels in both raw and produced drinking water. Careful monitoring of PFAS in drinking water is ongoing. Use of PFAS-containing products may also contribute to exposure through the indoor environment (dust and air) [48, 56]. Humans can in addition be exposed to PFAS through occupational exposure [57].

A recent preliminary risk assessment of PFOS and PFOA concluded that a substantial proportion of the European population exceeds the proposed TWIs of 13 ng/kg bw for PFOS and 6 ng/kg bw for PFOA [41]. The TWI for PFOS was based on increased serum cholesterol levels at a benchmark dose ($BMDL_5$) of 21–25 ng/mL in plasma in adults and a decreased antibody response after vaccination at a $BMDL_5$ of 10.5 ng/mL in 5-year-old children. The TWI for PFOA was based on increased serum cholesterol levels at a benchmark dose ($BMDL_5$) of 9.2–9.4 ng/mL in plasma/serum. According to the Swedish Market Basket Survey 2015 [22], the average per capita intake of PFOS and PFOA was highly unlikely to exceed the TWIs suggested by EFSA [41]. However, this survey did not consider the intake from drinking water which, if the drinking water is contaminated, is an important source of exposure. Recent biomonitoring studies in Sweden suggest that human exposure to PFOS and PFOA has decreased [58, 59], likely due to the phase-out of production and use of PFOA and related substances [60].

Metals and metalloids

Metals are abundant in the environment as a result of both anthropogenic and natural activities. Many metals are essential for humans and have important physiological functions. Essential metals are for example copper (Cu), zinc (Zn), iron (Fe) and manganese (Mn). Some metals and metalloids such as cadmium (Cd), mercury (Hg), lead (Pb) and arsenic (As) are non-essential and are regarded as toxic for biota, including humans. Metals can be present at various levels in the environment, e.g. the soil, water and atmosphere. Humans may be exposed to metals from the environment or by ingestion of contaminated food or water.

Cadmium (Cd) is found in the environment occurring both naturally and from industrial and agricultural use. Health effects from long-term exposure to Cd include renal dysfunction and osteoporosis [61]. Cd exposure may also be related to some forms of cancer [62]. The current TWI for Cd, 2.5 µg/kg bw, is based on epidemiological studies on the relationship between urinary Cd and a biomarker of renal tubular effects [63]. For non-smokers, food is the main source of exposure to Cd with cereals, vegetables and potatoes being the main contributors [22, 64]. Residents in locations with high Cd pollution have increased Cd levels in blood and urine, which are mainly explained by consumption of locally grown vegetables and contaminated water [65]. The intestinal absorption of Cd can be affected by dietary intake of essential minerals (iron, calcium, zinc, copper) and protein [66].

Mercury (Hg) exists in the environment in inorganic or organic forms. The organic form, methylmercury (MeHg), is toxic to the central nervous system. The fetal period, when the nervous

system is developing, is particularly sensitive to MeHg exposure. MeHg can accumulate in fish, shellfish and sea mammals and fish is the main contributor to Hg exposure from the diet [22]. High concentrations of MeHg may be found in fresh-water fish from contaminated lakes (perch, pike, pikeperch, burbot) and in large predatory fish species such as tuna, swordfish, Atlantic halibut, shark and ray. Therefore, the Swedish Food Agency recommends that women who are planning to get pregnant, are pregnant or nursing limit their consumption of such fish species to 2-3 portions per year. In Swedish adults, the concentrations of Hg were positively related to fish intake (in particular, shellfish and saltwater fish) [67]. EFSA [68] established a TWI for inorganic Hg of 4 µg/kg bw, and for MeHg of 1.3 µg/kg bw. The German Human Biomonitoring Commission has estimated that there is no risk of adverse health effects of Hg at concentrations in whole blood below 5 µg/L, whereas adverse health effects are possible at concentrations above 15 µg/L [30].

Lead (Pb) is a naturally occurring toxic metal found in the Earth's crust. Pb-containing gasoline was a source of environmental and human exposure to organic Pb components for many decades [69]. Because of its toxicity, this gasoline was gradually phased out in many countries during the 1990s. Pb can adversely affect the central nervous system in developing infants [70]. Pb may also affect blood pressure, reproductive and kidney function, and cause mutagenesis. Children are more vulnerable to Pb toxicity than adults, especially for neurological toxicity [71]. EFSA (2010) established a reference point of 12 µg/L of Pb in blood for developmental neurotoxicity and a reference point of 15 µg/L for an effect on the prevalence of chronic kidney disease in adults [72]. Humans are exposed to Pb via food, drinking water and air [73]. According to the Swedish Market Basket Survey 2015, Pb is measurable in all food categories with the main contributors to intake being the mixed category ‘sugar and sweets’ (including sauces and dressings), cereal products, vegetables, fruits and beverages [22]. Several studies have shown that in the Swedish population blood levels of Pb have decreased since the 1970s due to a gradual reduction of lead in petrol [74-76].

Chromium (Cr) is ubiquitous in the diet and found in meat and meat products, oils and fats, breads and cereals, fish, pulses and spices [77]. Cr in food is mostly trivalent and, due to poor absorption, trivalent Cr compounds have low toxicity after oral exposure [78]. EFSA derived a tolerable daily intake (TDI) of 300 µg trivalent Cr/kg bw and concluded that the mean dietary exposure levels in Europe are well below the TDI [77]. It was previously believed that trivalent Cr is involved in regulating carbohydrate and lipid metabolism by improving insulin responsiveness [79]. However, in 2014, EFSA questioned the essentiality of trivalent Cr for humans because of the lack of convincing evidence [77].

Manganese (Mn) is an essential mineral necessary for normal growth and development, particularly for normal brain development and function as well as for normal amino acid, lipid, protein and carbohydrate metabolism. As Mn is present in commonly consumed food, deficiency of Mn is rare. Absorption of Mn is tightly regulated in the gut and toxicity from dietary exposure has therefore not been reported [80]. Around the world, Mn toxicity is due to environmental exposure, including airborne exposure and drinking water [81]. High exposure may adversely affect brain function causing neurological, cognitive and neuropsychological effects [80]. The Swedish Market Basket Survey 2015 estimated a daily Mn intake of 4.2 mg per person and day [22], which is somewhat higher than the adequate intake (AI) of 3 mg per day proposed by EFSA [82].

Cobalt (Co) is a constituent of vitamin B12, but has no other known biological role. The primary exposure routes are through inhalation and ingestion of food and drinking water [83]. Paustenbach et

al. [84] discussed adverse health effects of high Co concentrations such as cardiomyopathy, vision or hearing impairment, reversible hypothyroidism and polycythaemia. However, these effects are unlikely to occur at Co levels below 300 µg/L in whole blood [85]. According to the Swedish Market Basket Survey 2015, the estimated daily intake of Co was 11 µg per person per day [22].

Nickel (Ni) is not essential for humans. According to EFSA, Ni in food can induce eczematous skin reactions among nickel-sensitized individuals and can affect reproduction and development in experimental animals [86]. Diet is likely the most important source of Ni exposure in the general population. Based on reproductive and developmental effects in experimental animals, EFSA [86] derived a TDI for Ni of 2.8 µg/kg bw. According to the Swedish Market Basket Survey 2015 [22], the average per capita intake of Ni from food (1.7 µg/kg bw) is below this TDI. The TDI may not be sufficiently protective for individuals sensitized to Ni since it has been reported that individuals with allergic contact dermatitis may develop eczematous skin reactions from oral exposure to nickel salts [86]. The current TDI for Ni has been questioned and a new evaluation is in progress.

Aluminium (Al) occurs naturally in the air, water and soil as a result of weathering of rocks and volcanic activity. Mining and processing Al also contribute to Al release into the environment. The general population is primarily exposed to Al from food, although minor exposure may occur from drinking water and inhalation of air [87]. Only a small portion of the ingested Al is absorbed. EFSA established a TWI for Al of 1 mg/kg bw [88]. According to the Swedish Market Basket Survey 2015 [22], the average per capita intake of Al in Sweden is 16 µg/kg bw/day or 0.11 mg/kg bw/week.

Arsenic (As) is a metalloid naturally present in the bedrock. It can be released by smelter operations and fossil-fuel combustion [89]. The toxicity of As depends on chemical structure. In its inorganic form (inorganic As, arsenite AsIII and arsenate AsV) As is highly toxic. As can also occur in organic forms (e.g. arsenobetaine (AB) and arsenosugars) that are supposed to be less harmful to health [90]. In aquatic environments, inorganic As is transformed to a range of organic metabolites, which are then found in tissues of aquatic species [91]. Inorganic As is mainly found in drinking water and enters the food chain through plant crops which absorb it from water and soils. EFSA reported that the main contributor to dietary exposure to inorganic As was the food group ‘grain-based processed products (non rice-based)’, in particular wheat bread and rolls [92]. Similarly, the Swedish Market Basket Survey 2015 demonstrated that cereals were the major contributor to exposure to inorganic As, although fish was the main contributor to total As [22, 93]. Rice is one of the foods that contains the highest concentrations of inorganic As, as well as some organic As [94]. The main adverse effects associated with long-term ingestion of inorganic As in humans are skin lesions, cancer, cardiovascular disease, developmental toxicity, neurotoxicity, abnormal glucose metabolism, and diabetes [95]. Neurotoxicity is mainly reported after acute exposure [95]. Exposure to As over many years can also increase the risk of some cancers, e.g. in lungs or bladder [96]. Epidemiological studies indicate that children may be more sensitive to inorganic As than adults [97]. In the human body inorganic As is metabolised to monomethylarsonic acid (MMA), which is further methylated to dimethylarsinic acid (DMA). From key epidemiological studies, EFSA has identified benchmark doses ($BMDL_{01}$) between 0.3 and 8 µg inorganic As/kg bw per day for cancer of the lung, skin and bladder as well as skin lesions [95]. According to the Swedish Market Basket Survey 2015, the estimated average dietary exposure to inorganic As in Sweden is 0.033 µg/kg bw/day, which is below this range [22].

Phthalate metabolites and phenolic compounds

Phthalates (diesters of phthalic acid) are a family of man-made chemical compounds used as plasticizers, solvents and additives in many industrial and personal care products. They continuously leach from products leading to contamination of food and environments, and to human exposure. Phthalates are commonly used in food packaging materials and can be released from packaging to food.

Phthalates have a short elimination half-life (estimated half-life of various phthalate metabolites is approximately 4 to 25 h); they are rapidly metabolised and excreted in urine [98]. Several phthalates are included on the EU candidate list of substances of very high concern due to reproductive toxicity. Five of these phthalates (dicyclohexyl phthalate, DCHP; di-ethylhexyl phthalate, DEHP; di-n-butyl phthalate, DnBP; benzyl butyl phthalate, BBP; diisobutyl phthalate, DIBP) are also classified as endocrine disruptors [99]. Due to their toxic properties, the use of phthalates has been restricted in toys and childcare articles in the EU since 2007. The use of some phthalates has therefore been or is being phased out and substituted with new chemicals with a similar function. For example, **di-iso-nonyl cyclohexane-1,2-dicarboxylate (DiNCH)** was introduced on the European market in 2002 to replace DEHP and other high-molecular weight phthalates in polyvinyl chloride (PVC) [100]. DiNCH rapidly became one of the most used non-phthalate plasticizers in Sweden [101]. Dietary sources are considered the major exposure route, although DiNCH also has been found in dust [102, 103]. In epidemiological studies, some phthalates are suspected to have long-term endocrine or neurotoxic effects on human health [104-106]. The German Human Biomonitoring Commission has estimated urine concentrations of some phthalate and DiNCH metabolites below which, according to current knowledge, there is no risk of adverse health effects [30]. Such concentrations are available for e.g. DEHP metabolites (sum of 5-OH-MEHP and 5oxo-MEHP) and DiNCH metabolites (sum of OH-MINCH and cx-MINCH) and are based on TDIs from risk assessments or from NOAEL values from critical animal studies.

Phosphorous flame retardants are used in a wide variety of consumer products. To a certain extent they replace brominated flame retardants that have been legislatively restricted and phased out. Triphenyl phosphate is an organophosphate ester used both as a flame retardant and plasticizer, and has been applied to polyurethane foam, resins, PVC, hydraulic fluids, lacquers, and nail polish [107]. Humans may be exposed to phosphorous flame retardants via dermal contact, diet and dust. A recent Swedish study on residues of phosphorus flame retardants in food reported that average per capita intakes are lower than health-based reference doses and that cereals, pastries, sugar/sweets and beverages are the main contributors to the total intake [108].

Bisphenol A (BPA) and its analogues **bisphenol S (BPS)** and **4,4-bisphenol F (BPF)** are well-known toxicants present in the environment. BPA is a high production-volume chemical used mainly in the manufacture of polycarbonate and epoxy plastics. This compound can migrate from food packaging materials to food [109]. BPS and BPF are currently used as alternatives to BPA because the use of BPA is strictly regulated. BPA is an endocrine-disrupting compound that has been shown to affect for example reproduction and development in animals. Possible effects of BPA exposure in humans are debated, but the substance might cause unwanted estrogenic effects in humans [110]. EFSA estimated dietary exposure to BPA in different population groups and concluded that dietary exposure to BPA is below the provisional TDI of 4 μg/kg bw per day in all groups, including infants and children [111]. However, EFSA underlined that there is considerable uncertainty in the exposure estimate for the non-

dietary sources, and more studies are needed to address these uncertainties [111]. The German Human Biomonitoring Commission has estimated that there is no risk of adverse health effects from BPA below urine levels of 0.1 mg/L in children and 0.2 mg/L in adults [30].

Polycyclic aromatic hydrocarbons (PAH) are formed during combustion processes and are found in complex mixtures in the environment, including foods. Among non-smokers, the main sources of human exposure are food and polluted air. The levels of PAH are especially high in smoked and grilled foods [112]. The Swedish Market Basket Survey 2015 [22] indicated that the estimated PAH intake from food has decreased during the last fifteen years. A possible explanation for this is improved production processes. As PAH are carcinogenic, the levels in food should be as low as possible as should be human exposure [112].

Organophosphate and pyrethroid insecticides are used in agriculture worldwide. These compounds were substitutes for organochlorine insecticides because of their higher susceptibility to environmental degradation. However, exposure to the organophosphate insecticide chlorpyrifos and pyrethroids may be associated with adverse central nervous system outcomes and developmental neurotoxicity [113-115]. In Sweden, no pesticide containing organophosphates has been approved since about 2010, and the use of pyrethroids has been restricted. Thus, exposure to residues of organophosphate and pyrethroid insecticides in the Swedish population occurs mainly from consumption of imported fruits and vegetables.

Phenolic substances such as triclosan (TCS, antibacterial agent), 3-tert-butyl-4-hydroxyanisole (BHA) and benzophenone-3 (BP-3, UV filter) are ingredients in cosmetic and personal care products. BHA is also used as an antioxidant and preservative in food. The use of these ingredients is of concern due to their potential side effects on human health. For example, animal *in vivo* and *in vitro* studies have shown that TCS and BHA adversely affect endocrine function and thyroid hormone homeostasis [116-118]. The German Human Biomonitoring Commission has estimated that there is no risk of adverse health effects of TCS at urine concentretions below 2000 ng/mL in children [30].

Aim

The aim of the present study was to estimate exposure to several contaminants in Swedish adolescents by analysing blood and urine concentrations in samples from participants in the national dietary survey *Riksmaten Adolescents 2016–17*. Examined contaminants included chlorinated and brominated POPs, PFASs, metals and metalloids, phthalate metabolites and phenolic compounds. Furthermore, we aimed to evaluate associations between blood/urine concentrations of contaminants and age, gender and geographical location.

Materials and methods

Study population

The study population for the present report was a subgroup of *Riksmaten Adolescents 2016–17*, from whom blood and urine was collected. *Riksmaten Adolescents 2016–17* is a nationally representative, cross-sectional, school-based dietary survey of children and adolescents. Details of the study design and sampling procedures are described elsewhere [119]. Briefly, students in grade 5, grade 8 and 2nd year of high school (referred to in the report as grade 11) were recruited from schools between September 2016 and May 2017. The survey included a web-based dietary assessment method, web questionnaires, weight and height measurements and physical activity assessment using accelerometers.

In the main study, 601 schools (approximately 200 from each school grade) were selected by Statistics Sweden (SCB) to represent Swedish students in the three age groups. Selection was based on geographical area, type of municipality as classified by the Swedish Association of Local Authorities and Regions, type of school (independent school with public funding or public), and size of school. Exclusion criteria were schools with fewer than 10 students in a school grade and high schools with only language introduction classes. Approximately 40% of the schools (n=259) were randomly selected for blood and urine sampling.

Schools were invited to participate through emails addressed to the principal. The email invitations were followed-up by telephone calls. To include equal numbers of participants from each school grade, recruitment was rotated between the grades. One or two classes were included from each school and when the desired number of classes in a school grade had been recruited, no further schools in that school grade were contacted. To achieve an even distribution over the year from different areas of Sweden, schools were recruited both in the spring and in the autumn in all the regions.

Names and addresses of students in the included classes were collected from the schools, and letters with information about the study and its aims were sent out to students' legal guardians approximately 3 weeks before the school visit. For schools with blood and urine sampling, consent forms for all students, and guardians of children younger than 16 years were included in the letters. Prior to the data collection, the class teacher was asked to show a short information film to the students. Written consent was collected by teachers and provided to the research team on the day of the school visit.

An outline of the recruitment of participants is presented in Figure 1. Of the 259 schools selected for blood and urine sampling, 62 (24%) participated, 79 (31%) declined participation, 7 (3%) were excluded and 111 (43%) were not contacted by phone as the desired number of classes had been reached. In total, 1,305 of the 2,377 invited students participated and complete dietary information and valid blood and urine samples were available from 1,105 students (46%). Ethical approval for the study was obtained from the Regional Ethical Review Board in Uppsala (No. 2015/190). Written informed consent was obtained from all participants their guardians if younger than 16 years.

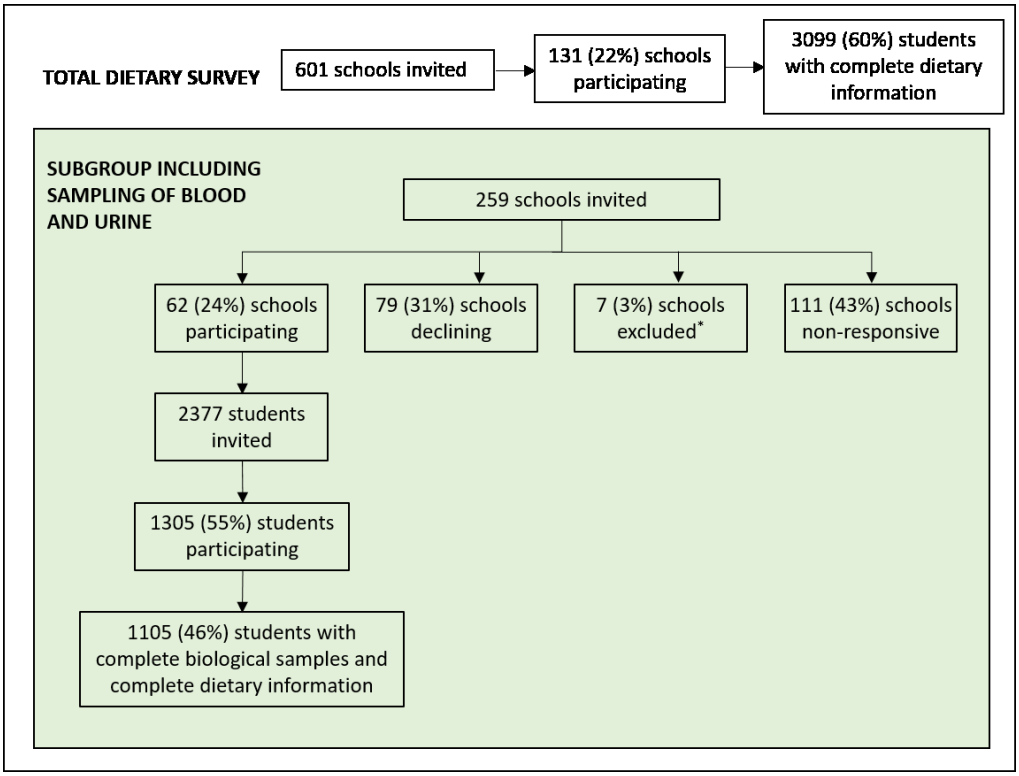

Figure 1. Overview flow diagram of participants in the main study and the biomonitoring subgroup of *Riksmaten Adolescents 2016–17*. *Schools were excluded if there were <10 students in a grade or they only taught language introduction.

Data collection

School visits

Field staff from the Swedish Food Agency planned and coordinated the school visits. They also visited the classes during school hours and instructed the students on how to record their diet, complete the on-line questionnaires and how to wear the accelerometer for the physical activity assessment. Trained staff from the regional Occupational and Environmental Medicine Divisions (AMM) in Gothenburg, Linköping, Lund, Stockholm and Umeå collected the blood and urine samples (Figure 2). The AMM clinic in Linköping also carried out the sample collection in the regions of Örebro and Uppsala.

At the school visit, the students recorded their dietary intake and completed the on-line questionnaires. The students were also instructed to ask their parents to complete one on-line questionnaire later at home and also to record another diet day at home. Non-fasting blood and spot urine samples were collected at the school health facilities or equivalent. After sampling, the participants were offered a snack of juice and fruit. Body weight and height, in light clothing, were also measured during the visit using a standardised protocol. Blood samples were centrifuged, and serum was stored at -20°C together with whole blood, plasma and urine. After the school visit, samples were transported to the AMM clinic and stored at -80°C.

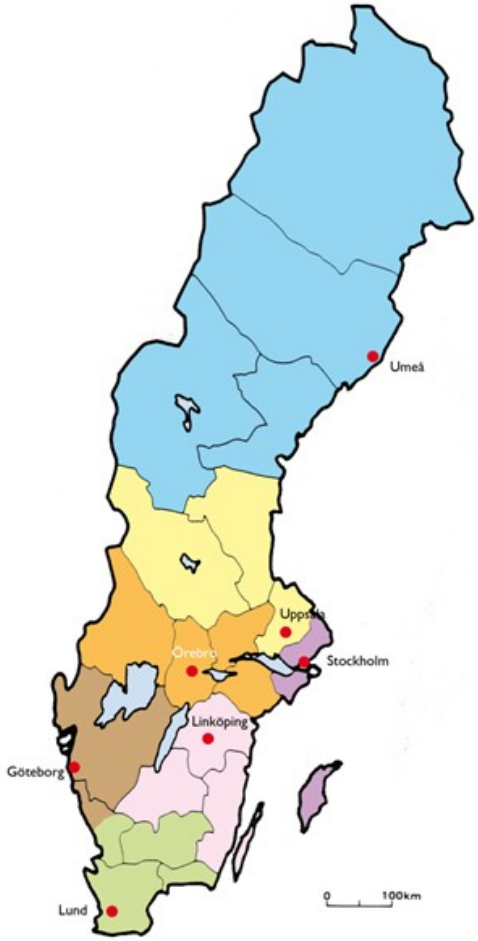

Figure 2. Regions and capitals of Occupational and Environmental Medicine Divisions (AMM) in Sweden.

Sample handling and storage

When a region had completed sampling in all schools, the samples were transported to the Swedish Food Agency and then stored at -80°C until they were distributed to various laboratories for analysis of contaminants. The concentration of contaminants was determined using either individual samples or pooled samples. Pooled samples of blood serum and urine were used for analyses of brominated flame retardants and arsenic. Thirty pools were constructed according to participant gender and grade (Table 1). Additionally, 12 pools were constructed according to income level in the birth country [120] of the participants and their mothers.

Variables in this report

Region and municipality were defined according to the location of the school. Regions were classified according to the area of the AMM clinics (Figure 2). Each region included 4–10 counties as specified in Table 6. Type of municipality was classified according to the Swedish Association of Local Authorities and Regions 2017 [121].

Information on age, gender, parental education and birth country was derived from the on-line questionnaires. Parental education was determined as the highest attained education level of either the mother or the father. Birth country of the participant and both parents were recorded in text and then coded as Nordic country or non-Nordic. Birth region was defined as Nordic if (1) the participant was

born in a Nordic country and at least one parent was born in a Nordic country or (2) the child was born outside the Nordic countries but both parents were born in the Nordic countries.

Body mass index was calculated (kg/m^2) and weight status was determined using the IOTF (International Obesity Task Force) cut-offs. For participants 18 years and above, the WHO cut-offs for weight status were used [122].

Table 1. Description of pooled samples from *Riksmaten Adolescents 2016–17* used for analyses of brominated flame retardants and arsenic species.

	Grade 5		Grade 8		Grade 11	
	Boys	**Girls**	**Boys**	**Girls**	**Boys**	**Girls**
Number of pools	5 pools	5 pools	5 pools	5 pools	5 pools	5 pools
(individual samples in each pool)	(20)	(20)	(20)	(20)	(20)	(20)
	Country of birth of participant/mother					
	Sweden/Sweden		**Sweden/Medium or low-income country***		**Medium or low-income country /Medium or low-income country***	
Number of pools	4 pools		4 pools		4 pools	
(individual samples in each pool)	(20)		(9)		(6)	

*Classification based on UN, 2014.

Analytical methods

Detailed descriptions of the analytical methods are provided in Appendix 1.

Chlorinated and brominated persistent organic pollutants

The analyses of chlorinated and brominated POPs (Table 2) in individual serum samples were performed by the Finnish National Institute for Health and Welfare, Department of Health Security (Appendix A1.1). The method used has been described previously [123]. In brief, concentrations were measured using gas chromatography - triple quadrupole mass spectrometry (GC-MS/MS). The instrument used was an Agilent 7010 GC-MS/MS system (Wilmington, DE, U.S.), GC column DB-5MS UI (J&W Scientific, 20m, ID 0.18 mm, 0.18 μm). Limits of quantification (LOQ) ranged from 5 pg/mL for PCB congeners and trans-nonachlor to 40 pg/mL for p,p'-DDE. Two blank samples and two control samples (NIST SRM 1958) were included in each batch of samples. Measured concentrations of chlorinated and brominated POPs in SRM1958 were 80-105% of the certified/reference concentrations. The coefficient of variation (CV%) from SRM 1958 (n=18) was <3.6% for all compounds.

Brominated flame retardants (Table 2) were analysed in 42 pooled serum samples at the Swedish Food Agency (Appendix A1.2). These pools were constructed as described in Table 1. The analyses were performed according to a previously described method using capillary gas chromatography and mass selective detection in electron capture negative ionization and selected ion monitoring modes (GC/LRMS/ECNI-SIM) [18]. LOQ values ranged from 0.6 to 20 ng/kg serum depending on the analyte. Measurement uncertainty was ≤50%.

Table 2. Chlorinated and brominated persistent organic pollutants analysed in serum samples from *Riksmaten Adolescents 2016–17*.

Chlorinated compounds analysed in individual samples	**Abbreviation in the report**
Pentachlorobenzene	PeCB
Hexachlorobenzene	HCB
α-hexachlorocyclohexane	α-HCH
β-hexachlorocyclohexane	β-HCH
γ-hexachlorocyclohexane	γ-HCH
1α,2β,4β,5,6,7β,8,8-octachloro-2,3α -epoxy-3a α,4,7,7a α -tetrahydro-4,7- methanoindan	Oxychlordane
1,2,3,4,5,6,7,8,8-nonachloro-3a,4,7,7a-tetrahydro-4,7-methanoindan	Trans-nonachlor
1,1-bis-(4-chlorophenyl)-2,2,2-trichlorethane	p,p'-DDT
1,1-dichloro-2,2-bis(p-chlorophenyl)ethylene	p,p'-DDE
2,4,4',5-tetrachlorobiphenyl	PCB-74
2,2',4,4',5-pentachlorobiphenyl	PCB-99
2,3',4,4',5-pentachlorobiphenyl	PCB-118
2,2',3,4,4',5'-hexachlorobiphenyl	PCB-138
2,2',4,4',5,5'-hexachlorobiphenyl	PCB-153
2,3,3',4,4',5-hexachlorobiphenyl	PCB-156
2,2',3,3',4,4',5-heptachlorobiphenyl	PCB-170
2,2',3,4,4',5,5'-2,2',3,4,4',5,5'-heptachlorobiphenyl	PCB-180
2,2',3,4,4',5',6-heptachlorobiphenyl	PCB-183
2,2',3,4',5,5',6-heptachlorobiphenyl	PCB-187
Brominated flame retardants analysed in individual samples	
2,2',4,4'-tetrabromodiphenylether	BDE-47
2,2',4,4',5-pentabromodiphenylether	BDE-99
2,2',4,4',5,5'-hexabromodiphenylether	BDE-153
Brominated flame retardants analysed in pooled samples	
2,4,4'-tribromodiphenylether	BDE-28
2,2',4,4'-tetrabromodiphenylether	BDE-47
2,3',4,4'-tetrabromodiphenylether	BDE-66
2,2',4,4',5-pentabromodiphenylether	BDE-99
2,2',4,4',6-pentabromodiphenylether	BDE-100
2,2',4,4',5,5'-hexabromodiphenylether	BDE-153
2,2',4,4',5,6'-hexabromodiphenylether	BDE-154
2,2',3,4,4',5',6-heptabromodiphenylether	BDE-183
2,2',3,3',4,4',5,5',6,6'-decabromodiphenylether (“deca-BDE”)	BDE-209
Hexabromocyclododecane	HBCDD
Hexabromobenzene	HBB
Pentabromoethylbenzene	PBEB
Decabromodiphenylethane	DBDPE
2-bis(2,4,6-tribromophenoxy)ethane	BTBPE

Per- and polyfluoroalkyl substances

PFAS analyses in serum samples were performed at the Department of Environmental Science and Analytical Chemistry (ACES), Stockholm University (Appendix A1.3). Full names and abbreviations of PFAS in this study are provided in Table 3 and Table 4. Sample extraction was carried out using a method adapted from Powley et al. [124]. The samples were analysed using a Waters Acquity ultra performance liquid chromatograph (UPLC) coupled to a Waters Xevo TQS triple quadrupole mass spectrometer. The mass spectrometer was operated in negative electrospray ionisation, multiple reaction monitoring (MRM) mode. Quantification was based on isotope dilution. LOQs are summarized in Table 12 and Table 13.

Table 3. PFAS analysed in serum samples from *Riksmaten Adolescents 2016–17*.

Compound	Abbreviation in the report
Perfluorohexanoic acid	PFHxA
Perfluoroheptanoic acid	PFHpA
Perfluorooctanoic acid, linear isomer	lin-PFOA
Perfluorooctanoic acid, branched isomer	br-PFOA
Perfluorononanoic acid	PFNA
Perfluorodecanoic acid	PFDA
Perfluoroundecanoic acid	PFUnDA
Perfluorododecanoic acid	PFDoDA
Perfluorotridecanoic acid	PFTriDA
Perfluorotetradecanoic acid	PFTeDa
Perfluoropentadecanoic acid	PFPeDA
Perfluorohexadecanoic acid	PFHxDA
Perfluorooctadecanoic acid	PFOcDA
Perfluorobutane sulfonic acid	PFBS
Perfluoropentane sulfonic acid	PFPeS
Perfluorohexane sulfonic acid, linear isomer	lin-PFHxS
Perfluorohexane sulfonic acid, branched isomer	br-PFHxS
Perfluoroheptane sulfonic acid	PFHpS
Perfluorooctane sulfonic acid, linear isomer	lin-PFOS
Perfluorooctane sulfonic acid, branched isomer	br-PFOS

Table 4. Extra target PFAS analysed in serum samples from *Riksmaten Adolescents 2016–17*.

Compound	Abbreviation in the report
Perfluorononane sulfonic acid	PFNS
Perfluorodecane sulfonic acid, linear isomer	lin-PFDS
Perfluorodecane sulfonic acid, branched isomer	br-PFDS
Perfluoroundecane sulfonic acid	PFUnDS
Perfluorooctane sulfonamide, linear isomer	lin-FOSA
Perfluorooctane sulfonamide, branched isomer	br-FOSA
Perfluorooctane sulfonamidoacetic acid, linear isomer	lin-FOSAA
Perfluorooctane sulfonamidoacetic acid, branched isomer	br-FOSAA
N-methylperfluorooctansulfonamide acetic acid, linear isomer	lin-MeFOSAA
N-methylperfluorooctansulfonamide acetic acid, branched isomer	br-MeFOSAA
N-Ethyl Perfluorooctane sulfonamidoacetic acid, linear isomer	lin-EtFOSAA
N-Ethyl Perfluorooctane sulfonamidoacetic acid, branched isomer	br-EtFOSAA
9-chlorohexadecafluoro-3-oxanonane-1-sulfonic acid	9Cl-PF3ONS
11-chloroeicosafluoro-3-oxanonane-1-sulfonic acid	11Cl-PF3OUdS
Ammonium 4,8-dioxa-3H-perfluorononanoate	ADONA
2,3,3,3-tetrafluoro-2-(heptafluoropropoxy)propionic acid	HFPO-DA
3:3 Fluorotelomer carboxylic acid	3:3 FTCA
5:3 fluorotelomer carboxylic acid	5:3 FTCA
7:3 fluorotelomer carboxylic acid	7:3 FTCA
1H,1H,2H,2H-perfluorohexane sulfonate	4:2 FTSA
1H,1H,2H,2H-perfluorooctane sulfonate	6:2 FTSA
1H,1H,2H,2H-perfluorodecane sulfonate	8:2 FTSA
6:2 Fluorotelomer phosphate diester	6:2 diPAP
8:2 Fluorotelomer phosphate diester	8:2 diPAP
6:2/8:2 Fluorotelomer phosphate diester	6:2/8:2 diPAP

Metals and metalloids

Analyses of Pb, Cd, Hg, Cr, Mn, Co and Ni in whole blood and Al in serum were performed at the division of Occupational and Environmental Medicine, Lund University (Appendix A1.4). The samples were treated as previously described [125]. The concentrations were determined by inductively coupled plasma mass spectrometry (ICP-MS; iCAP Q, Thermo Fisher Scientific, Bremen, GmbH) equipped with collision cell with kinetic energy discrimination and helium as collision gas. The limit of detection (LOD) varied from 0.05 to 5.0 µg/L. Method precision varied from 2.8 to 15% depending on analyte.

Levels of total arsenic and arsenic species were determined in 123 individual urine samples and in 42 pooled urine samples at the Swedish Food Agency (Appendix A1.4). The As species determined were: inorganic As (sum of arsenite and arsenate) and the organic As species dimethyl arsenate (DMA), monomethyl arsonate (MMA) and arsenobetain (AB). Levels of inorganic As and MMA were

determined according to the European standard for inorganic As based on anion-exchange HPLC-ICP-MS [126], and levels of DMA and AB according to a method based on cation–exchange HPLC-ICP-MS [127]. Total arsenic was determined by ICP-MS according to the European Standard for total As [128]. The methods for total As, inorganic As and MMA are accredited by SWEDAC (Sweden's national accreditation body). An Agilent 7700x ICP-MS was used in all three methods. The LOD values were 0.2 µg/kg for inorganic As and MMA, 0.4 µg/kg for DMA and AB, and 2.2 µg/kg for total As. The expanded uncertainty was ±34% for inorganic As and MMA, and ±20% for total As, DMA and AB. Relative standard deviations (RSD) for analysis of the urine certified reference materials (NIST 2669 Arsenic species in frozen human urine Level 1 and 2, and Seronorm Trace Elements Urine L-1) varied from 2.4 to 8.1% depending on the As species analysed (Appendix A1.4). The relative bias (the differences between the concentration of the reference material and the concentration found using the analytical method) varied from -12% to +19% (Appendix A1.4).

Phthalate metabolites and phenolic compounds

Phthalate metabolites and phenolic compounds were analysed in urine by the Division of Occupational and Environmental Medicine, Lund University, as previously described [129] with some modifications (Appendix A1.5). The measured compounds and their parent compounds are listed in Table 5. The samples were analysed on a Shimadzu UFLC system (Shimadzu Corporation, Kyoto, Japan) coupled to a QTRAP5500 triple quadrupole linear ion trap mass spectrometer equipped with a TurboIon Spray source (LC-MS/MS; AB Sciex, Foster City, CA, U.S.). The LOD values are summarized in Table 22. The CV% of the quality control sample did not exceed 20% for almost all compounds with the exception of cx-MiDP (27% at concentration of 0.6 ng/mL), BPS (25% at concentration of 0.8 ng/mL), DBP (41% at concentration of 0.1 ng/mL) and BHA (21% at concentration of 0.8 ng/mL).

Urine concentrations were adjusted to the mean urine density in the *Riksmaten Adolescents 2016–17* population (1.022 kg/L) according to Carnerup et al. [130].

Table 5. Phthalate metabolites and phenolic substances analysed in urine samples from *Riksmaten Adolescents 2016–17*.

Compound	Abbreviation in the report	Parent compound*
Phthalate metabolites		
Monoethyl phthalate	MEP	DEP
Mono-butyl phthalate (sum of MnBP and MiBP, mono-isobutylphthalate)	MnBP	DBP
Monobenzyl phthalate	MBzP	BBzP
Mono-(2-ethylhexyl) phthalate	MEHP	DEHP
Mono-(2-ethyl-5-hydroxylhexyl) phthalate	5-OH-MEHP	DEHP
Mono-(2-ethyl-5-oxohexyl) phthalate	5-oxo-MEHP	DEHP
Mono-(2-(carboxymethyl)hexyl) phthalate	2-cx-MEHP	DEHP
Mono-(2-ethyl-5-carboxypentyl) phthalate	5-cx-MEPP	DEHP
Mono-(4-methyl-7-hydroxyloctyl) phthalate	OH-MiNP	DiNP
Mono-(4-methyl-7-oxooctyl) phthalate	oxo-MiNP	DiNP
Mono-(4-methyl-7-carboxyheptyl) phthalate	cx-MiNP	DiNP
Monocarboxyisononyl phthalate	cx-MiDP	DiDP/DPHP
6-hydroxy monopropylheptylphthalate	OH-MPHP	DiDP/DPHP
DiNCH metabolites		
Cyclohexane-1,2-dicarboxylate-mono(oxo-isononyl) ester	oxo-MiNCH	DiNCH
1,2-Cyclohexanedicarboxylic acid mono 4-methyl-7-carboxy-heptyl ester	cx-MiNCH	DiNCH
2-(((Hydroxy-4-methyloctyl)oxy)carbonyl)cyclohexanecarboxylic acid	OH-MiNCH	DiNCH
Phosphorous flame retardants		
Diphenyl phosphate	DPP	TPP
Dibutyl phosphate	DBP	TBP
Bis(2-butoxyethyl)phosphate	BBOEP	TBOEP
Bisphenols		
Bisphenol A	BPA	
Bisphenol S	BPS	
4,4-Bisphenol F	4,4-BPF	
Polycyclic aromatic hydrocarbons (PAH) metabolites		
Hydroxyphenanthrene (sum of 2-OH and 3-OH phenanthrene)	2-OH-PH	Phenanthrene
1-Hydroxypyrene	1-HP	Pyrene
Insecticide metabolites		
Trichloropyridinol	TCP	Chlorpyrifos
3-Phenoxybenzoic acid	3-PBA	Pyrethroids
Other phenolic substances		
Triclosan	TCS	
3-tert-Butyl-4-hydroxyanisole	BHA	
Bensophenone-3 / Oxybenzone	BP-3	

*DEP, di-ethyl phthalate; DBP, di-butyl phthalate, BBzP, butylbenzyl phthalate; DEHP, di-ethylhexyl pthtalate; DiNP, di-iso-nonyl phthalate; DiDP, di-iso-decyl phthalate; DPHP, di-propylheptyl phthalate; DiNCH, di(isononyl) cyclohexane-1,2-dicarboxylate; TPP, triphenyl phosphate; TBP, tributyl phosphate; TBOEP, tris(2-butoxyethyl) phosphate.

Statistical analysis

The statistical software package STATA (version 15; StataCorp, College Station, TX, U.S.) was used to perform all statistical analyses. Concentrations of contaminants were described by number of participants and percentages with concentrations below LOD or LOQ, arithmetic means, standard deviation (SDs), 5th percentile, median and 95th percentile. All descriptive statistical analyses were run on untransformed data. In all calculations, concentrations of compounds below LOD or LOQ were substituted with a value equal to LOD/√2 or LOQ/√2, respectively.

Differences in concentrations between regions, grades and genders were investigated for compounds with detectable levels in more than 50% of the samples using analysis of variance (ANOVA).

The factors included in the statistical models are listed below. The choice of factors was based on their relevance for the substance group.

Chlorinated and brominated persistent organic pollutants – region, grade and gender, interactions between the factors, and weight status as categorical variable.

Per- and polyfluoroalkyl substances – region, grade and gender, interactions between region and gender, interactions between grade and gender.

Metals, phthalate metabolites and phenolic compounds – region, grade and gender, interactions between all these factors.

Arsenic – grade and gender, or country of birth of participant/mother (for pooled samples).

For each compound, differences between grades were also estimated separately for boys and girls using the same factors as in the main module excluding gender.

Additionally, differences in metal concentrations associated with smoking habits were evaluated among participants in grades 8 and 11 using a model with the factors region, gender and smoking habits.

Tukey's multiple comparison test was used to estimate differences between groups. All ANOVA tests were performed on natural log-transformed data to normalize the distribution. The results are reported as back-transformed least squares means with 95% confidence intervals (CI) for the purpose of clearer data presentation.

Results and discussion

Characteristics of the study population

The distribution of participants according to region, type of municipality, gender, grade, birth region, parental education and weight status is presented in Table 6. More girls than boys (56% vs 44%, Table 6) participated in the study, especially in grades 8 and 11 (Table 7). The largest disparity between genders was observed in grade 11 in Umeå and Uppsala. Number of participants by gender within grade and region is presented in Table 7. Age and anthropometric details of the study population combined for both genders are presented in Table 8. The participants were aged between 10 and 21 years old, with an average age of 14.7 years.

Table 6. Main characteristics of the participants in the *Riksmaten Adolescents 2016–17* subgroup that donated biological samples (n=1,105).

	Number of participants	Percentage (%)
Region (counties randomly selected within region)		
Gothenburg (Gothenburg, Åmål, Borås, Lidköping, Härryda, Lysekil)	223	20.2
Linköping (Linköping, Eksjö, Tranås, Norrköping, Motala, Mjölby)	157	14.2
Lund (Lund, Älmhult, Landskrona, Burlöv, Kävlinge, Sjöbo, Ronneby, Malmö, Lessebo, Ljungby)	210	19.0
Stockholm (Stockholm, Danderyd, Gotland, Nacka)	168	15.2
Umeå (Umeå, Boden, Lycksele, Piteå, Östersund)	131	11.9
Uppsala (Uppsala, Falun, Sandviken, Tierp)	84	7.6
Örebro (Eskilstuna, Hallstahammar, Hammarö, Karlskoga, Nyköping, Trosa, Västerås)	132	12.0
Type of municipality*		
Large cities and municipalities near large cities	281	25.4
Medium-sized towns and municipalities near medium-sized towns	585	52.9
Smaller towns/urban areas and rural municipalities	239	21.6
Gender		
Boys	483	43.7
Girls	622	56.3
Grade		
5	333	30.1
8	413	37.4
11 (2nd year high school)	359	32.5
Birth region**		
Nordic	924	83.6
Non-nordic	173	15.6
Unknown	8	0.8
Parental education***		
Primary school	44	4.0
High school	361	32.7
University degree	645	58.4
Unknown	55	5.0
Weight status******		
Underweight	74	6.7
Normal	796	72.0
Overweight	192	17.4
Obese	43	3.9

*According to the classification of Swedish municipalities by the Swedish Association of Local Authorities and Regions, 2017.

** Nordic birth region is defined as i) the child was born in the Nordic countries and at least one parent was born in the Nordic countries or ii) the child was born outside of the Nordic countries but both parents were born in the Nordic countries.

***Parental education is determined as the highest attained education level of either the mother or the father.

**** According to IOTF: International Obesity Task Force at the age of < 18 and WHO: World Health Organization at the age ≥ 18.

Table 7. Number of participants in the *Riksmaten Adolescents 2016–17* subgroup that donated biological samples per gender within grade and region (n=1,105).

Region	**Grade 5**		**Grade 8**		**Grade 11**	
	Boys	**Girls**	**Boys**	**Girls**	**Boys**	**Girls**
Gothenburg	38	36	38	50	27	34
Linköping	11	9	32	43	21	41
Lund	36	35	36	49	31	23
Stockholm	20	36	36	41	19	16
Umeå	23	22	6	11	12	57
Uppsala	10	8	14	13	9	30
Örebro	28	21	18	26	18	21
Total per gender	166	167	180	233	137	222

Table 8. Age and anthropometric measurements of participants in the *Riksmaten Adolescents 2016–17* subgroup that donated biological samples (n=1,105). Data are presented as mean±standard deviation (minimum; maximum in brackets).

Characteristic	**Grade 5**	**Grade 8**	**Grade 11**	**All participants**
	n=333	**n=413**	**n=359**	**n=1,105**
Age, years	11.6±0.4	14.5±0.4	17.8±0.7	14.7±2.5
	(10.6; 13.1)	(11.6; 15.7)	(16.8; 21.1)	(10.6; 21.1)
Height, cm	151.4±7.9	168.1±8.4	171.3±9.9	164.1±12.2
	(130.0; 177.0)	(146.6; 192.0)	(141.0; 198.2)	(130.0; 198.2)
Body weight, kg	43.8±10.4	58.6±10.6	68.2±13.5	57.3±15.1
	(23.1; 100.7)	(37.7; 108.5)	(40.5; 124.4)	(23.4; 124.4)

Chlorinated and brominated persistent organic pollutants

Chlorinated and brominated POPs were analysed in serum samples from 1096 participants, and the results are shown in Table 9. Full summary statistics per region, grade and gender are provided in Appendix 2, Tables A2.1–A2.3.

Table 9. Serum concentrations of chlorinated and brominated persistent organic pollutants (pg/mL) in Swedish adolescents (n=1,096). Concentrations below LOQ were replaced by LOQ/√2 in the calculations.

Compound	% of samples						
	LOQ	<LOQ	Mean	SD	5th percentile	Median	95th percentile
PeCB	10	99.8	7	0.3	<LOQ	<LOQ	<LOQ
HCB	10	0	52	224	22	43	75
α-HCH	20	100	14	0	<LOQ	<LOQ	<LOQ
β-HCH	15	95	14	25	<LOQ	<LOQ	15
γ-HCH	20	99.5	14	3	<LOQ	<LOQ	<LOQ
Oxychlordane	25	99.8	18	0.5	<LOQ	<LOQ	<LOQ
Trans-nonachlor	5	77	5	4	<LOQ	<LOQ	10
p,p'-DDT	15	97	13	20	<LOQ	<LOQ	<LOQ
p,p'-DDE	40	7	191	417	<LOQ	93	575
PCB-74	5	76	6	32	<LOQ	<LOQ	10
PCB-99	5	68	5	5	<LOQ	<LOQ	11
PCB-118	5	30	8	6	<LOQ	6	17
PCB-138	5	1	32	23	6	27	76
PCB-153	5	0.1	53	39	13	43	129
PCB-156	5	60	6	5	<LOQ	<LOQ	15
PCB-170	5	16	16	14	<LOQ	12	42
PCB-180	5	4	32	29	5	24	87
PCB-183	5	91	4	2	<LOQ	<LOQ	6
PCB-187	5	46	8	7	<LOQ	6	19
Sum of PCB 138, 153 and 180			118	89	27	95	283
Total PCB*			170	128	52	137	390
BDE-47	15	98	11	8	<LOQ	<LOQ	<LOQ
BDE-99	15	99	11	7	<LOQ	<LOQ	<LOQ
BDE-153	15	99.5	11	5	<LOQ	<LOQ	<LOQ

*Total PCB concentration was calculated by summing concentrations of all individual PCB congeners in each sample.

The highest median concentrations were found for p,p′-DDE, followed by PCB-153, HCB, PCB-138 and PCB-180 (Table 9), which is in agreement in general with previous Swedish studies on adults of both genders [131] and pregnant women [132]. These compounds were observed at quantifiable levels in more than 90% of the participants. PCB-138, PCB-153 and PCB-180 accounted for 70% of the total PCB concentrations. Concentrations of α-HCH, γ-HCH, oxychlordane, p,p'-DDT, BDE-47, BDE-99 and BDE-153 were below LOQ in more than 96% of the samples.

Although p,p′-DDT has been prohibited as a pesticide for decades in most countries, its major metabolite p,p′-DDE is still detected in a majority of human blood samples (93% in the present study). There were large variations in the p,p′-DDE concentrations ranging from <LOQ to 6134 pg/mL.

Six out of 10 analysed PCBs were quantified in more than 50% of the participants (Table 9). The higher chlorinated di-ortho congeners PCB-138, PCB-153 and PCB-180 have also previously been reported as the most abundant PCBs in Swedish men aged 40-75 years from both urban and rural areas of Sweden [133], and in Swedish women aged 50–74 years [134]. The German Human Biomonitoring Commission has estimated that there is no risk of adverse health effects of non-dioxin-like PCBs at serum levels (sum of PCB 138, PCB 153 and PCB 180 multiplied by 2) at or below 3.5 µg/L and that adverse health effects are possible above 7 µg/L [30]. No individual in our study exceeded 3.5 µg/L and the highest sum concentration was 1.6 µg/L.

PBDE accumulation in human blood and milk is well documented [135, 136]. In the present study, serum concentrations of BDE-47, BDE-99 and BDE-153 were only quantified in a few samples (≤ 2%) (Table 9). In an earlier Swedish study on adults, higher quantification frequencies for BDE-47, BDE-99 and BDE-153 were reported [137]. It should however be emphasised that Bjermo et al. [137] calculated LOQ values using ng/g lipid weight, in contrast to the pg/mL of serum used in the present study, limiting comparison.

Table 10. Concentrations of brominated flame retardants in pooled serum samples (ng/kg) in Swedish adolescents (n=41). Concentrations below LOQ were replaced by LOQ/√2 in the calculations.

Compound	% of samples >LOD	LOQ	% of samples <LOQ	Mean	SD	5th percentile	Median	95th percentile
BDE-28	0	0.9-1.2	100	0.8	0.05	<LOQ	<LOQ	<LOQ
BDE-47	76	5.6-7.2	80	6.5	3.88	<LOQ	<LOQ	14.5
BDE-66	0	0.5-0.7	100	0.4	0.02	<LOQ	<LOQ	<LOQ
BDE-99	76	7.9-10	85	7.9	3.40	<LOQ	<LOQ	14.4
BDE-100	59	1.9-2.4	88	1.8	0.51	<LOQ	<LOQ	2.9
BDE-153	98	2.4-3.0	83	2.2	0.50	<LOQ	<LOQ	3.2
BDE-154	7	1.9-2.6	100	1.7	0.11	<LOQ	<LOQ	<LOQ
BDE-183	15	0.5-0.7	85	1.1	3.96	<LOQ	<LOQ	1.3
BDE-209	63	16-21	88	16	3.79	<LOQ	<LOQ	25
HBCDD	49	2.0-2.7	88	2.0	0.87	<LOQ	<LOQ	3.2
BTBPE	12	0.9-1.2	95	0.8	0.20	<LOQ	<LOQ	<LOQ
DBDPE	0	16-22	100	14	0.91	<LOQ	<LOQ	<LOQ
HBB	0	0.5-0.7	100	0.4	0.02	<LOQ	<LOQ	<LOQ
PBEB	0	0.5-0.7	100	0.4	0.02	<LOQ	<LOQ	<LOQ

In addition to the PBDEs in individual samples, 9 PBDE-congeners, HBCDD and four emerging BFRs were analysed in 42 pooled serum samples. Concentrations of BFRs in these pooled serum samples are shown in Table 10. One of the pools was lost during sample extraction and, consequently, the total number of analysed pools is 41. Many of the analysed compounds had concentrations below LOQ in most samples. The highest quantification and detection frequencies were found for those substances that were also analysed in the individual serum samples (i.e. BDE-47, BDE-99 and BDE-153). Quantification frequencies for these substances were however higher in the pooled samples than in the individual samples, probably due to lower LOQs in the analytical method used for the pools. Levels of emerging BFRs (BTBPE, DBDPE, HBB, and PBEB) were below LOD and LOQ in all samples, with the exception of BTBPE which could be detected in 5 pools and quantified in 2 pools. Due to the low

number of samples with BFR levels above LOQ, no further statistical analyses or comparisons with other studies were performed.

Differences in concentrations of chlorinated persistent organic pollutants between regions

Summary statistics of chlorinated and brominated POPs per region are presented in Appendix 2, Table A2.1. The concentrations of HCB and p,p'-DDE differed between regions ($p<0.001$ for both) (Figure 3). Regional differences were also observed for total PCB (Figure 3) as well as individual PCBs (PCB-118, PCB-138, PCB-153, PCB-170, PCB-180 [$p<0.001$ for all] and PCB-187 [$p= 0.007$]) (Figure 4). There was no clear pattern in the differences between regions. In contrast, in a study on Swedish adults (*Riksmaten adults 2010–11*), the highest concentrations of PCBs were observed in the Lund region [131]. In the same study, the concentrations of HCB were generally higher in Lund and Stockholm. Only small regional differences in the concentrations of PCBs and PBDEs have been previously observed in mother's milk among primiparous women in Sweden, suggesting similar organohalogen compound exposure in the studied regions (Lund, Gothenburg, Uppsala and Lycksele) [138].

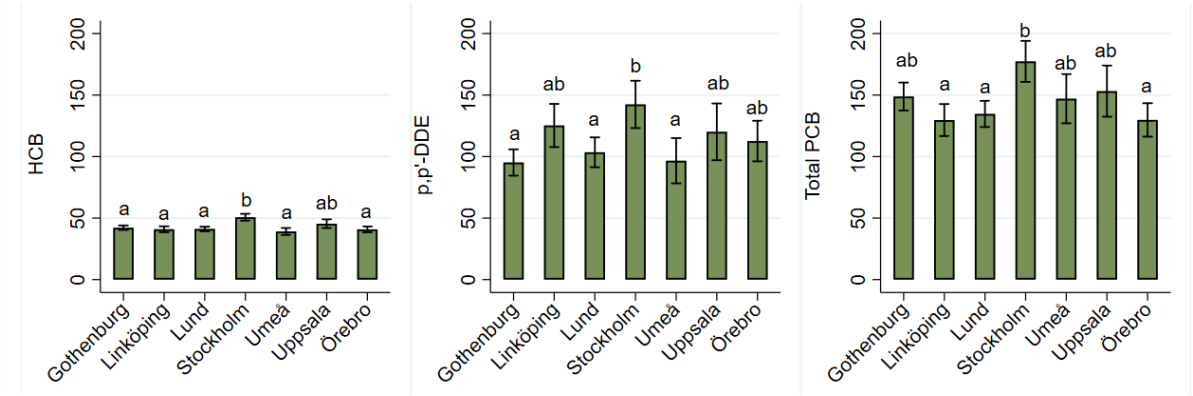

Figure 3. Serum concentrations of HCB, p,p'-DDE and total PCB (pg/mL) in Swedish adolescents per region (back-transformed least squares means with 95% confidence intervals from the analysis of log values). Differences between the least squares means were estimated by ANOVA (main factors included region, grade and gender, and the interactions between the factors; weight status was also included in the model). One individual with an extreme HCB level (7420.6 pg/mL) was excluded from the statistical analysis. Different letters indicate significant differences between groups ($p<0.05$) according to Tukey's multiple comparison test. Number of observations per region: Gothenburg (n=221), Linköping (n=155), Lund (n=210), Stockholm (n=164), Umeå (n=131), Uppsala (n=84), Örebro (n=130).

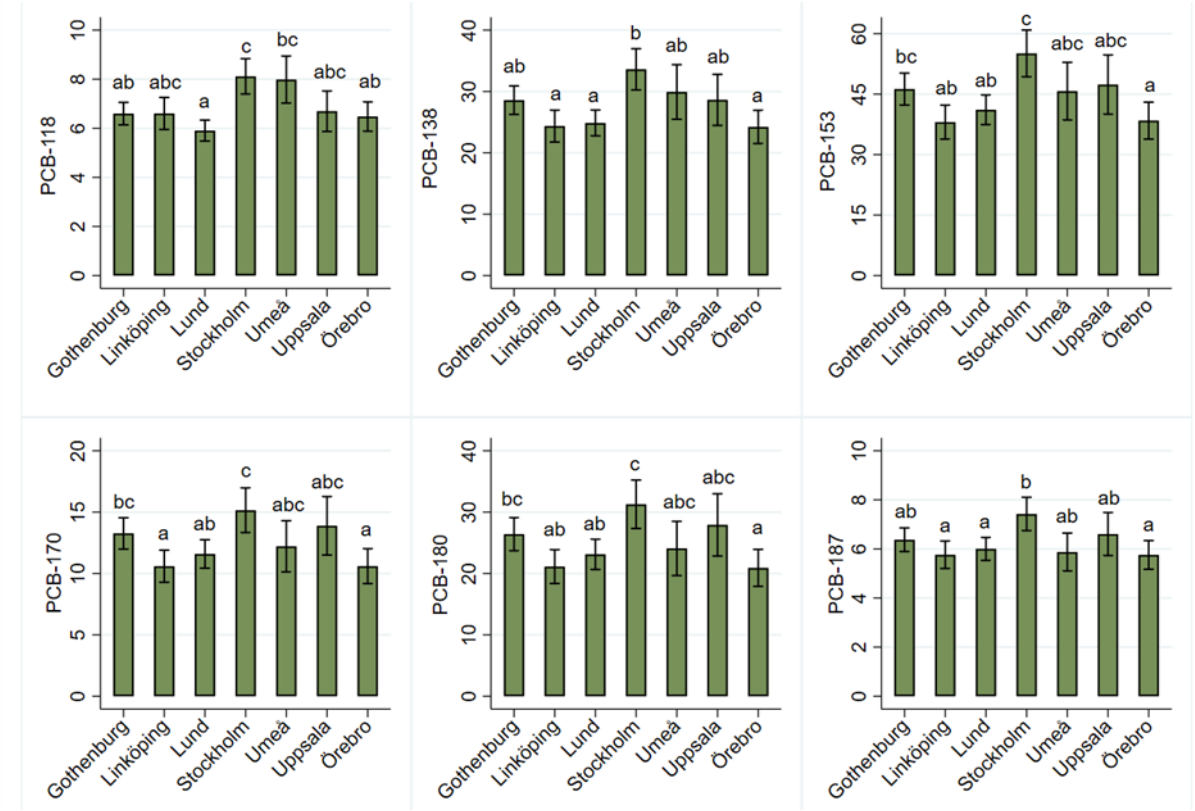

Figure 4. Serum concentrations of individual PCBs (pg/mL) in Swedish adolescents per region (back-transformed least squares means with 95% confidence intervals from the analysis of log values). Differences between the least squares means were estimated by ANOVA (main factors included region, grade and gender, and the interactions between the factors; weight status was also included in the model). Different letters indicate significant differences between groups (P<0.05) according to Tukey's multiple comparison test. Number of observations per region: Gothenburg (n=221), Linköping (n=155), Lund (n=210), Stockholm (n=164), Umeå (n=131), Uppsala (n=84), Örebro (n=130).

Differences in concentrations of chlorinated persistent organic pollutants between genders and grades

Summary statistics of chlorinated and brominated POPs per gender and grade are presented in Appendix 2, Tables A2.2 and A2.3. Boys had significantly higher concentrations of HCB, p,p'-DDE and PCBs compared with girls, with the exception of PCB-118 which did not differ between genders (Table 11). This suggests, for example, the possibility of gender-related differences in dietary habits or physiology. Consumption of fish, an important contributor to PCB exposure, was higher in boys than in girls in *Riksmaten Adolescents 2016–17* [139]. Higher levels of chlorinated POPs in men than in women have been observed in several studies on adult populations [131, 140].

Concentrations of p,p'-DDE, PCB-153, PCB-170 and PCB-180 differed significantly between grades and tended to increase with increasing age (Table 11). When estimating age differences within each gender separately, increasing PCB levels with increasing age were only observed in boys. Higher body burdens of POPs in older individuals have been reported in several studies on adults [131, 134, 140]. This is expected since these persistent compounds accumulate in the body during life. The lack of an obvious age association may be due to small differences in age between participants and/or the increase in body weight during adolescence. In contrast, Gallo et al. [141] observed that levels of PCBs and p,p'-DDE decreased with age among 151 individuals aged 13 to 18 years.

Table 11. Back-transformed least squares means (pg/mL) with 95% confidence interval of chlorinated and brominated persistent organic pollutants in Swedish adolescents per grade and gender*.

Compound	Boys				Girls					
	Grade 5 n=166	Grade 8 n=180	Grade 11 n=136	p-value, grade within boys	Grade 5 n=166	Grade 8 n=229	Grade 11 n=219	p-value, grade within girls	p-value, overall effect of gender	p-value, overall effect of grade
HCB	46 (43-48)	49 (46-51)	48 (45.5-51)	0.485	42[a] (39-44)	38[b] (36-40)	39[b] (37-41)	0.012	0.001	0.454
p,p'-DDE	116 (100-133)	121 (106-137)	147 (126-169)	0.058	102 (88-116)	94 (84-104)	108 (94-122)	0.167	0.001	0.017
PCB-118	6.8 (6.2-7.4)	6.9 (6.3-7.5)	7.6 (6.9-8.3)	0.256	6.7 (6.1-7.3)	6.6 (6.1-7.2)	6.8 (6.2-7.3)	0.653	0.123	0.269
PCB-138	27[a] (25-30)	32[ab] (29-35)	34[b] (31-38)	0.019	24 (21-26)	26 (23-28)	25 (23-27)	0.672	0.001	0.070
PCB-153	44[a] (39-48)	53[ab] (48-58)	55[b] (49-61)	0.025	37 (33-41)	42 (38-46)	40 (36-44)	0.408	0.001	0.035
PCB-170	12[a] (11-14)	15[ab] (14-17)	16[b] (14-18)	0.018	10 (7-11)	12 (10-13)	11 (10-13)	0.230	0.001	0.007
PCB-180	24[a] (21-27)	31[b] (27-35)	33[b] (29-37)	0.011	19 (17-21)	23 (20-25)	22 (20-25)	0.146	0.001	0.003
PCB-187	6.0[a] (5.5-6.6)	7.3[ab] (6.7-8.0)	7.7[b] (6.9-8.4)	0.030	5.5 (5.0-6.0)	5.9 (5.4-6.4)	5.7 (5.2-6.2)	0.655	0.001	0.093
Total PCB	145[a] (131-158)	169[ab] (154-184)	179[b] (162-196)	0.036	124 (112-135)	136 (124-147)	134 (121-146)	0.636	0.001	0.056

*Differences between the least squares means were estimated by ANOVA (main factors included region, grade and gender, and the interactions between the factors). Weight status was included in the model. Effect of grade was calculated separately for each gender using the same factors as in the main module without gender effect. Different letters indicate significant differences between the grades ($p<0.05$) according to Tukey's multiple comparison test.

Per- and polyfluoroalkyl substances

Concentrations of 11 individual PFAS, total PFHxS and total PFOS in serum samples are presented in Table 12. Full summary statistics per region, grade and gender are provided in Appendix 2, Tables A2.4–A2.6.

Table 12. Serum concentrations of PFAS (ng/g) in Swedish adolescents (n=1,096-1,098). Concentrations below LOQ were replaced by LOQ/√2 in the calculations.

Compound	LOQ	% of samples <LOQ	Mean	SD	5th percentile	Median	95th percentile
PFHpA	0.06-0.29	92	0.09	0.07	<LOQ	<LOQ	<LOQ
lin-PFOA	0.02-0.29	0.2	1.35	0.75	0.59	1.20	2.46
br-PFOA	0.02-0.29	100	0.06	0.05	<LOQ	<LOQ	<LOQ
PFNA	0.06-0.29	7	0.43	0.25	<LOQ	0.38	0.86
PFDA	0.06-0.29	38	0.20	0.15	<LOQ	0.16	0.44
PFUnDA	0.06-0.29	53	0.15	0.11	<0.06	<LOQ	0.34
PFBS	0.04-0.54	85	0.09	0.14	<0.04	<0.07	0.37
lin-PFHxS	0.02-0.46	8	1.72	10.7	<LOQ	0.40	2.62
br-PFHxS	0.02-0.19	96	0.10	0.22	<LOQ	<LOQ	<LOQ
total PFHxS			1.82	10.9	0.19	0.47	2.68
lin-PFOS	0.06-0.56	0	3.05	5.82	0.80	2.00	7.03
br-PFOS	0.06-0.56	2	1.55	4.69	0.38	0.92	2.90
total PFOS			4.61	10.4	1.23	2.89	9.72

Of the measured PFASs, lin-PFOA, PFNA, lin-PFHxS, lin-PFOS and br-PFOS showed the largest number of samples with concentrations above LOQ (Table 12); in contrast concentrations of PFHxA, PFDoDA, PFTriDA, PFTeDa, PFPeDa, PFDS and FOSA were below LOQ in more than 99% of the samples (data not shown). Lin-PFOS showed the highest median and 95th percentile concentration, followed by lin-PFOA, br-PFOS and lin-PFHxS. Lin-PFOA was quantified in more than 99% of the samples, whereas br-PFOA was not quantified in any of the samples (Table 12). This is not surprising for two reasons. Firstly, br-PFOA is eliminated faster than lin-PFOA [142]. Secondly, br-PFOA is only produced by electrochemical fluorination, a process that was phased out for the most part by 3M in 2002 [143]. Since then, mostly telomerised PFOA has been produced, which is strictly linear [143].

The levels of PFOA in the present study were similar to these reported in Swedish adolescents from Skåne (age 17–21 years), where the mean concentration was 1.15 ng/mL [144]. The levels of PFOA were also comparable to levels in a study of adolescents from the U.S. (age 12–19 years, 2013–2014), where the median concentration of lin-PFOA was 1.6 ng/mL [145], and from Norway (age 15–19 years, 2010–2011), where the median concentration of total PFOA was 1.9 ng/ml [146]. In a preliminary opinion, EFSA suggests a TWI of 6 ng/kg bw for PFOA [41]. This TWI was based on increased serum cholesterol levels at a benchmark dose ($BMDL_5$) of 9.2–9.4 ng/mL in plasma/serum [41]. In our study, the 95th percentile of lin-PFOA in serum was 2.5 ng/g (Table 12), which is lower than $BMDL_5$ used by EFSA. Only one sample had concentrations of PFOA slightly above the benchmark dose (9.8 ng/g).

The concentrations of PFNA in the present study were comparable with those measured in adolescents from southern Sweden (mean concentration 0.41 ng/mL) [144] and the U.S. (median concentration 0.5 ng/mL) [145]. The median concentration of PFNA in Swedish adults, sampled 2010–2011, was 0.80 ng/mL [147].

Lin-PFHxS was quantified in 92% of the samples, whereas br-PFHxS was quantified in only 4% of the samples. This finding was expected because it is known that linear forms of PFHxS predominate over branched isomers in humans, and branched isomers are eliminated faster [142, 143]. The median concentration of lin-PFHxS was 0.4 ng/g (<LOQ for br-PFHxS) which is comparable with adolescents from Sweden (0.29 ng/mL) [144] and Norway (0.55 ng/mL) [146], but lower than the 1.1 ng/mL reported in adolescents from the U.S. [145].

The samples contained a higher percentage of linear relative to branched isomers of total PFOS (approximately 70% linear isomers of total PFOS). This is consistent with the isomer pattern of technical PFOS [148] and with the pattern in humans reported in previous studies [145, 149, 150]. It should be noted, however, that there are larger variations in the relative percentage of linear and branched PFOS isomers in human samples compared with technical PFOS, suggesting possible pharmacokinetic differences between branched and linear PFOS [151] or isomer-specific biotransformation of PFAA-precursors [143].

The median total PFOS concentration was 2.9 ng/g, which is similar to 2.5 ng/mL in Swedish adolescents from Skåne [144], slightly lower than 3.6 µg/L in adolescents in the U.S. [145], and more than 2-fold lower than 6.2 ng/mL in Norwegian adolescents [146]. In a preliminary opinion, EFSA suggests a TWI of 13 ng/kg bw/week for PFOS [41]. This TWI was based on increased serum cholesterol levels at a benchmark dose ($BMDL_5$) of 21–25 ng/mL in plasma in adults and decreased antibody response after vaccination at a $BMDL_5$ of 10.5 ng/mL in 5-year-old children [41]. In our study, the 95th percentile of total PFOS in serum was 9.7 ng/g (Table 12). Eighteen individuals (1.6%) had PFOS concentrations above 21 ng/g serum and 49 individuals (4.5%) had concentrations above 10.5 ng/g. Among the 18 individuals with PFOS concentrations above 21 ng/g serum, all but one was from a school in Ronneby municipality that had known previous contamination of the drinking water.

PFUnDA and PFBS were detected in less than 50% of the samples. Median concentrations for both compounds were below LOQ, which is similar to the results for adolescents from the U.S. [145]. Norén et al. [144] reported median concentrations of 0.13 ng/mL for PFUnDA in Swedish adolescents, and Averina et al. [146] reported median concentrations of 0.16 ng/mL for PFUnDA and 0.04 ng/mL for PFBS in Norwegian adolescents. In serum from Swedish adults, the median concentration of PFUnDA was 0.33 ng/mL [131]. It should be noted however that LOD was 0.08 ng/mL in that study [131], whereas in the present study LOQ varied from 0.058 to 0.288. Thus, it is difficult to compare the median concentrations between the studies.

The serum concentrations of PFHpA were below LOQ in 92% of the samples with median <LOQ (Table 12). In contrast, levels were below LOD only in 19–25% of the samples in other studies on Swedish and Norwegian adolescents [144, 146]. PFHpA concentrations in Swedish adults (18–80 years) were below LOD in 88% (258 of 292) of the samples [131].

Detection and quantification frequencies for PFAS in biomonitoring studies are strongly affected by the wide sensitivity range of analytical methods used for their analysis, thus making it difficult to

compare the results from different studies. Nevertheless, it appears that PFAS levels in the present study are comparable to these reported in previous studies on adolescents.

In the present study we additionally measured several extra target PFAS (Table 3 and Table 13). Overall, the quality of these data was lower than for the standard targets due to the absence of exactly matched isotopically labelled internal standards. Due to these analytical difficulties, only six of the 25 analysed compounds could be quantified with adequate quality (Table 13). The highest quantification frequency was observed for L-Me-FOSAA (14 samples, 1.3%). 9Cl-PF3ONS was quantified in 2 samples, and ADONA in 1 sample.

Table 13. Serum concentrations of extra target PFAS (ng/g) in Swedish adolescents (n=1,098). Concentrations below LOQ were replaced by LOQ/√2 in the calculations.

Compound	LOQ	% of samples <LOQ	Mean	SD	5th percentile	Median	95th percentile
L-Me-FOSAA	0.08-3.57	99	0.30	0.57	<LOQ	<LOQ	<LOQ
Br-Me-FOSAA	0.08-3.57	100	0.29	0.54	<LOQ	<LOQ	<LOQ
9Cl-PF3ONS	0.02-3.50	99.8	0.57	0.88	<LOQ	<LOQ	<LOQ
11Cl-PF3OUdS	0.06-3.49	100	0.51	0.79	<LOQ	<LOQ	<LOQ
ADONA	0.02-3.57	99.9	0.29	0.66	<LOQ	<LOQ	<LOQ
4:2 FTSA	0.06-0.29	100	0.14	0.07	<LOQ	<LOQ	<LOQ

Differences in concentrations of per- and polyfluoroalkyl substances between regions

Summary statistics of PFAS per region are provided in Appendix 2, Table A2.4. There were small but significant differences in concentrations of PFOA, PFNA, PFOS and PFHxS between regions. There was no consistent pattern in the differences between regions, but in general the concentrations in the Umeå region were among the lowest. PFOA and PFNA concentrations were among the highest in the Stockholm region (Figure 5). Total PFHxS concentrations appeared to be higher in the samples from the Lund, Stockholm, Uppsala and Örebro regions (Figure 5). The concentrations of lin- and br-PFOS were highest in the Lund and Stockholm regions.

Exposure via contaminated drinking water is an important determinant for PFHxS levels in some human populations [52, 152]. For PFOS and PFOA, it has been suggested that other sources of exposure, such as diet, influence these PFAS levels [59, 147, 152]. In 2012, it was discovered that certain drinking water production wells in the City of Uppsala were contaminated with PFHxS, and to a lesser degree with PFOS, PFOA and PFBS [52]. In the present study, only one school was situated in Uppsala city, and as could be expected, the samples from this school had higher PFHxS levels than samples from other schools within the Uppsala region. In 2011, elevated levels of several PFAS were detected in drinking water in Stockholm compared with the levels in other major European cities [153].The same year (2011), higher levels of PFAS were found in drinking water from Tullinge in Botkyrka municipality [154]. However, in the present study we did not observe any relevant differences between the schools within the Stockholm region. On the contrary, within the Lund region, the concentrations of PFOS, PFHxS and PFOA were highest in samples from Ronneby municipality. The highest concentrations of PFOS, PFHxS and PFOA in the entire *Riksmaten Adolescents 2016–17* population were also found among participants from Ronneby. This is not surprising since very high

levels of PFOS and PFHxS were detected in one of two waterworks in Ronneby municipality in 2013 [155]. Even though distribution of contaminated water has stopped, the exposed population still has elevated levels of PFAS.

Thus, one important reason for regional variations in PFAS levels in Swedish adolescents is variations in drinking water levels. However, dietary habits, such as consumption of fish, is also an important source of exposure [147].

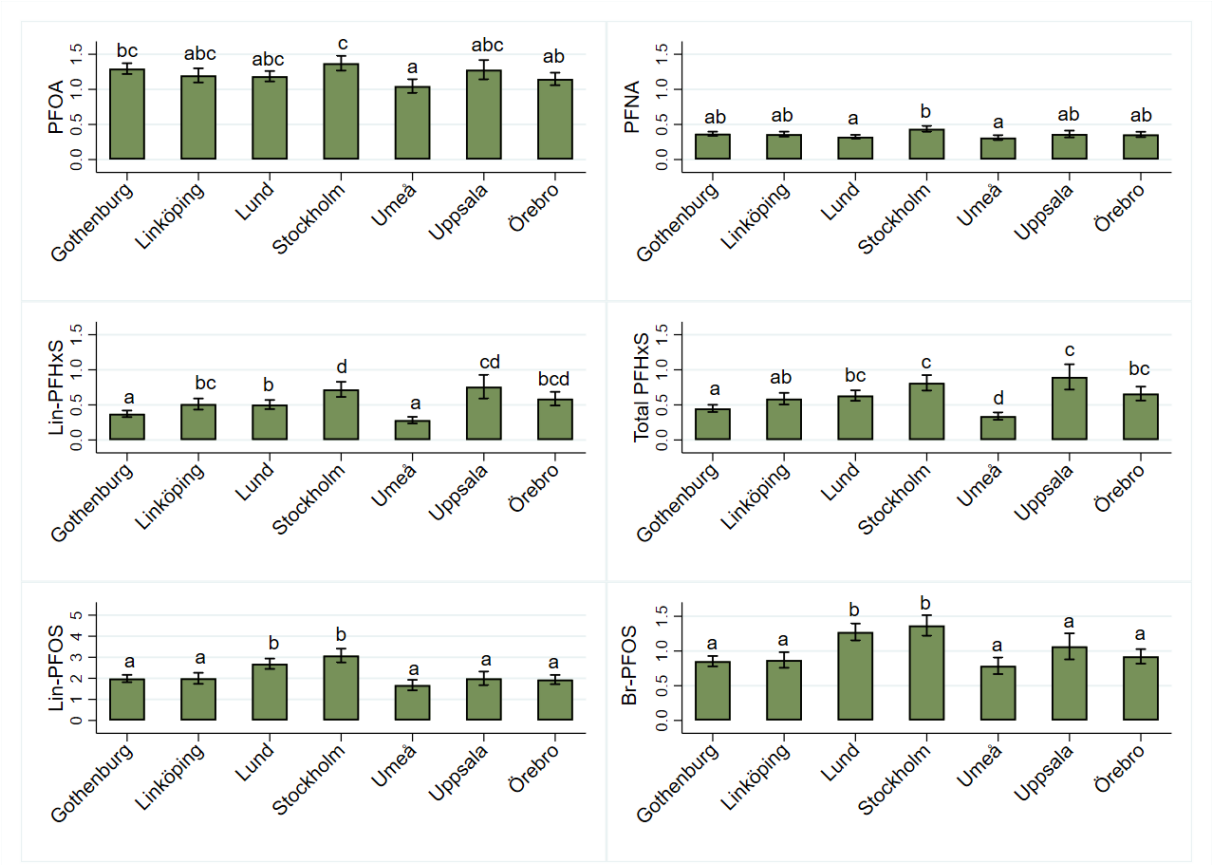

Figure 5. Serum concentrations of PFOA, PFNA, lin-PFHxS, total PFHxS, lin-PFOS and br-PFOS (ng/g) in Swedish adolescents per region (back-transformed least squares means with 95% confidence intervals from the analysis of log values). Differences between the least squares means were estimated by ANOVA (main factors included region, grade and gender, and the interactions between region and gender, and between grade and gender). Different letters indicate significant differences between regions ($p<0.05$) according to Tukey's multiple comparison test. Number of observations per region: Gothenburg (n=220–221), Linköping (n=156), Lund (n=209), Stockholm (n=166–167), Umeå (n=131), Uppsala (n=84), Örebro (n=130).

Differences in concentrations of per- and polyfluoroalkyl substances between genders and grades

Summary statistics of PFAS per gender and grade are presented in Appendix 2, Tables A2.5 and A2.6. In the present study, significant gender differences in serum concentrations were observed for PFNA, PFHxS and PFOS (Table 14). The concentrations of these compounds were lower in girls compared to boys. Differences in blood concentrations of PFAS between genders have been consistently observed

in previous studies on adolescents [146, 156] and adults [147, 156, 157]. For example, lower concentrations of PFNA, PFHxS, PFOS and PFOA were observed in Swedish adult females compared with males [147]. Similarly, Kärrman et al. [158] reported higher concentrations of PFOS, PFOA, and PFHxS in pooled serum samples from Australian adult males compared with females. Physiological differences such as menstrual excretion [159] may be responsible for lower PFAS concentrations in females of reproductive age. Additionally, it has also been suggested that renal elimination of some PFAS is gender-dependent and might contribute to final concentrations of PFAS in blood [160]. In contrast, in the present study the concentrations of PFOA were similar in Swedish adolescent boys and girls.

Serum concentrations of lin-PFOA, PFNA, PFHxS and PFOS differed significantly between grades, with a tendency for higher levels in grade 5 compared with grades 8 and 11 (Table 14). Similarly, serum concentrations of PFOS, PFOA, PFNA and PFHxS decreased in Finnish children from 1 to 10.5 years of age [161]. Koponen et al. [161] suggested that this decrease might be due to so called 'growth dilution', a dilution of the PFAS concentration in serum as a result of an increase in body mass and blood volume during growth. Age-related decreases of PFOA and PFHxS were also found by Toms et al. [162] when comparing the concentrations in children below 15 years and adults in Australia. In adult populations, an increase of PFAS levels is usually observed with age [147, 157] suggesting an ongoing bioaccumulation process. However, as discussed by Nøst et al. [163], estimating age-associated variations in PFAS concentrations may be complicated because exposure to PFAS has changed over time, and thus it is difficult to distinguish between the effect of changes in exposure and accumulation due to age.

Table 14. Back-transformed least squares means (ng/g) with 95% confidence interval of PFAS in serum in Swedish adolescents per grade and gender*.

Compound	Boys				Girls					
	Grade 5 n=166	Grade 8 n=180	Grade 11 n=136	p-value, grade within boys	Grade 5 n=166	Grade 8 n=230	Grade 11 n=218-220	p-value, grade within girls	p-value, overall effect of gender	p-value, overall effect of grade
lin-PFOA	1.5[a] (1.4-1.6)	1.1[b] (1.0-1.2)	1.2[b] (1.1-1.3)	0.001	1.2 (1.2-1.3)	1.1 (1.0-1.2)	1.2 (1.1-1.3)	0.045	0.092	0.001
PFNA	0.42[a] (0.38-0.46)	0.35[b] (0.32-0.38)	0.42[a] (0.38-0.47)	0.005	0.33 (0.30-0.36)	0.32 (0.29-0.34)	0.36 (0.33-0.39)	0.218	0.001	0.003
PFDA	0.16 (0.15-0.18)	0.15 (0.13-0.16)	0.17 (0.15-0.18)	0.309	0.15 (0.14-0.17)	0.15 (0.14-0.17)	0.16 (0.15-0.18)	0.634	0.893	0.317
lin-PFHxS	0.91[a] (0.77-1.04)	0.42[b] (0.36-0.48)	0.54[b] (0.45-0.62)	0.001	0.69[a] (0.59-0.79)	0.35[b] (0.31-0.40)	0.33[b] (0.28-0.37)	0.001	0.001	0.001
total PFHxS	1.0[a] (0.89-1.2)	0.49[b] (0.43-0.56)	0.62[b] (0.53-0.71)	0.001	0.80[a] (0.69-0.91)	0.43[b] (0.38-0.48)	0.41[b] (0.36-0.46)	0.001	0.003	0.001
lin-PFOS	2.9[a] (2.6-3.2)	2.1[b] (1.9-2.4)	2.5[ab] (2.2-2.8)	0.001	2.3[a] (2.1-2.5)	1.8[b] (1.6-1.9)	2.0[ab] (1.8-2.2)	0.001	0.001	0.001
br-PFOS	1.4[a] (1.2-1.5)	0.97[b] (0.88-1.1)	1.1[b] (0.99-1.3)	0.001	1.1[a] (0.99-1.2)	0.75[b] (0.69-0.82)	0.85[b] (0.77-0.94)	0.001	0.001	0.001
total PFOS	4.3[a] (3.8-4.7)	3.2[b] (2.9-3.6)	3.7[ab] (3.3-4.1)	0.001	3.5[a] (3.1-3.8)	2.5[b] (2.3-2.8)	2.9[b] (2.6-3.1)	0.001	0.001	0.001

*Differences between the least squares means were estimated by ANOVA (main factors included region, grade and gender, and the interactions between region and gender, and between grade and gender). Effect of grade was estimated separately for each gender using the same factors as in the main model without gender effect. Different letters indicate significant differences between the grades ($p<0.05$) according to Tukey's multiple comparison test.

Metals and metalloids

Metals

The concentrations of metals in blood are shown in Table 15. Nearly all participants had detectable concentrations of Cd, Hg, Pb, Cr, Mn, Co and Ni in whole blood. Al was detected in serum from 520 of 1,095 samples (47.5%). Full summary statistics per region, grade and gender are provided in Appendix 2, Tables A2.7-A2.9.

Table 15. Concentrations of metals in whole blood (cadmium, mercury, lead, chromium, manganese, cobalt, nickel) and serum (aluminium) (µg/L) in Swedish adolescents (n=1,099, n=1,095 for Al). Concentrations below LOD were replaced by LOD/√2 in the calculations.

Metal	LOD	% of samples <LOD	Mean	SD	5th percentile	Median	95th percentile
Cd	0.05	2.7	0.16	0.22	0.06	0.12	0.35
Hg	0.05	0.7	0.90	0.88	0.13	0.72	2.06
Pb	0.07	0.1	8.35	6.38	3.71	7.13	16.32
Cr	0.20	0.3	0.59	0.20	0.34	0.58	0.92
Mn	0.16	0.1	11.06	3.38	6.74	10.51	17.15
Co	0.05	2.1	0.15	0.10	0.06	0.12	0.36
Ni	0.21	0.1	0.65	0.20	0.41	0.62	0.93
Al	5	52.5	6	4.5	<LOD	<LOD	12

Concentrations of Cd in the present study (geometric mean: 0.16 µg/L) were slightly higher compared with previously published data on Swedish children in Lessebo municipality and Landskrona (geometric means: 0.09 µg/L and 0.10 µg/L, respectively) [164]. It should be noted, however, that the children in that study were younger (grades 2–4) than the participants from *Riksmaten Adolescents 2016–17*.

Concentrations of Hg in the present study (geometric mean: 0.66 µg/L) were similar to previously published data on Swedish children in Lessebo municipality and Landskrona (geometric means: 0.70 µg/L and 0.77 µg/L, respectively) [164]. According to the German Human Biomonitoring Commission, there is no risk of adverse health effects at Hg concentrations in whole blood ≤5 µg/L, whereas adverse health effects are possible at concentrations ≥15 µg/L [30]. In the present study, the 95th percentile of Hg was below both values (Table 15). Only 4 individuals had concentrations of Hg between 5 and 15 µg/mL.

Concentrations of Pb in the present study (geometric mean: 7.3 µg/L) were somewhat lower compared to previously published data on Swedish children in Lessebo municipality and Landskrona (geometric means: 9.9 µg/L and 7.8 µg/L) [164]. In a risk assessment, EFSA uses a reference point of 12 µg/L blood for developmental neurotoxicity in small children and a reference point of 15 µg/L for effects on prevalence of chronic kidney disease in adults [72]. In the present study, the 95th percentile of the measured Pb levels was 16 µg/L. Approximately 13% of the participants had Pb concentrations above 12 µg/L and 7% had concentrations above 15 µg/L. Even though exposure to Pb has been reduced in the Swedish population since the 1970s, our findings demonstrate that exposure is still high and underlines the importance of a further reduction from all sources.

In the present study, concentrations of Cr ranged from 0.14 to 2.95 µg/L with geometric mean of 0.57 µg/L. These concentrations were somewhat higher than the geometric means reported in Belgium among Flemish adolescents aged 14–15 years (0.33 µg/L and 0.26 µg/L) [165].

Concentrations of Mn varied from 0.01 to 33 µg/L with a median of 10.5 µg/L (Table 15). This is comparable to the medians (9.8 µg/L and 6.7 µg/L) measured in first-grade schoolchildren aged about 7 years in South Africa [166], and to the median (9.9 µg/L) measured in 12–19 years old adolescents in the U.S. [145]. The use of blood Mn as a biomarker of Mn exposure has, however, been questioned [167, 168].

Concentrations of Co varied from 0.04 to 0.73 µg/L with a median of 0.12 µg/L (Table 15). The geometric mean in the present study (0.13 µg/L) was comparable to that in a German adult population (0.14 µg/L) [169]. In contrast, higher means have been reported in a French adult population (20–59 years) (0.30 µg/L) [170] and among Swedish adolescents (15–17 years) in Trollhättan and Uppsala in 1993–1996 (approximately 0.31 µg/L) [171].

Concentrations of Ni varied from 0.15 to 3.62 µg/L with a median of 0.62 µg/L (Table 15). In a French adult population, the median concentration in blood was twofold higher, 1.28 µg/mL [170].

Al was quantifiable in 47.5% of the samples (Table 15). In a French adult population, Al was detectable in 86% of samples, but the mean Al concentration (4.3 µg/L) was lower than that found in our study [170].

Differences in concentrations of metals between regions

Summary statistics of metal concentrations in blood per region are presented in Appendix 2, Table A2.7. The only significant differences in metal concentrations between regions were observed for Cr and Ni (Figure 6). The highest concentrations of Cr were observed in samples from the Örebro region followed by Uppsala and Linköping. In contrast, Ni concentrations were higher in the Stockholm region compared to Linköping, Lund, Uppsala and Örebro. The reasons for these small differences in concentrations are unknown.

According to the ANOVA test, the concentrations of Cd also significantly differed between the regions ($p=0.044$); however, but Tukey's multiple comparison test did not reveal any differences. Similarly to the present results, Hg, Pb and Cd concentrations did not differ between the same Swedish regions in a previous study on Swedish adults [67]. However, Bárány et al. [171] found significantly lower concentrations of Pb and Cd, and higher concentrations of Hg in blood from teenagers (15–17 years) in Uppsala compared to Trollhättan. The differences could be explained by different industrial emissions, diet and individual metabolic factors affecting the absorption and kinetics of metals.

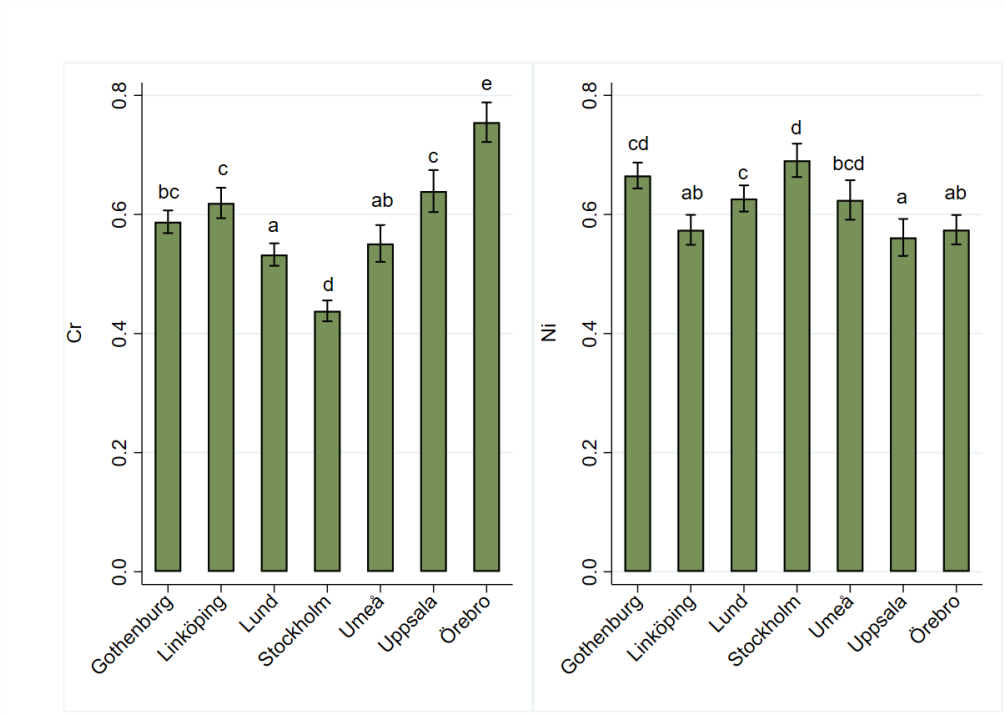

Figure 6. Concentrations of chromium and nickel (µg/L) in whole blood in Swedish adolescents per region (back-transformed least squares means with 95% confidence intervals from the analysis of log values). Differences between least squares means were estimated by ANOVA (main factors included region, grade and gender, and the interactions between the factors). Different letters indicate significant differences between groups (P<0.05) according to Tukey's multiple comparison test. Number of observations per region: Gothenburg (n=222), Linköping (n=157), Lund (n=210), Stockholm (n=166), Umeå (n=130), Uppsala (n=84), Örebro (n=130).

Differences in metal concentrations between genders and grades

Summary statistics of metal concentrations in blood by grade and gender are presented in Appendix 2, Tables A2.8 and A2.9.

Concentrations of Cd were higher in girls than boys (Table 16). These results are in agreement with previous observations of higher concentrations of Cd in blood, urine, and kidneys in women than in men [67, 76, 172, 173]. This might be due to generally higher gastrointestinal absorption of dietary Cd by females compared to males due to low iron status among women [173, 174]. Almost one third of the girls in grades 8 and 11 in *Riksmaten Adolescents 2016–17* had low iron depots (ferritin levels in blood) [139]. Cd accumulations also increased with age (Table 16), as previously reported [173].

The concentrations of total Hg were higher in boys than girls (Table 16), which is in agreement with results from studies of Swedish and Canadian adult populations [67, 175]. There was no overall association between grade and concentrations of total Hg. However, among girls, levels were higher in grade 5 compared with grade 11. In contrast, Jung et al. [176] observed a positive correlation between Hg concentrations and age in male, but not female, college students. Bárány et al. [171] did not observe any age- or gender-associated variations in the concentrations of Hg from Swedish teenagers (15–17 years). A previous study demonstrated that the levels of total Hg were positively associated with age in the Swedish population [67]. However, the participants in Bjermo et al. [67] were adults with a broader age range (18–80 years) compared to the present study. Differences in fish

consumption are probably responsible for the observed differences in total Hg between the groups in that study [67].

The concentrations of Pb were higher in boys than girls. This is in accordance with previous studies on children, adolescents and adults [67, 75, 171, 177]. Tukey's multiple comparison test did not reveal any differences between grades within boys but showed lower Pb concentrations in girls from grade 11 compared to grade 5. Again, a previous study on adults showed a positive association between Pb concentrations and age [67].

The concentrations of Cr were slightly higher in both boys and girls from grade 11 compared to grade 5 (Table 16). In contrast, Heitland and Köster [169] observed lower concentrations of Cr in urine from participants aged 12–17 years compared to participants aged 7–11 years. Girls in the present study had overall lower concentrations of Cr than boys.

The concentrations of Mn were significantly higher in participants from grade 8 compared to grades 5 and 11 (Table 16). Previous studies have shown higher Mn levels in children than adults [169, 178]. Overall, girls had higher concentrations of Mn than boys. Previous studies from other countries also demonstrated higher blood concentrations of Mn in females than males [179-181]. These differences were explained by females having lower concentrations of ferritin [182].

Similarly to Mn, the concentrations of Co were highest in participants from grade 8, and were generally higher in girls (Table 16). Bárány et al [171] also reported higher levels of Co in female adolescents compared to males. Similarly to Cd and Mn, low iron deposits can lead to increased absorption of Co in the gastrointestinal tract, which may explain the higher levels in girls [183].

The concentrations of Ni were similar among boys and girls in the present study. Significant differences between grades were observed in boys but not in girls (Table 16). In regard to Mn and Co, the concentrations of Ni were highest in the boys from grade 8.

The underlying reasons for the small differences in blood concentrations of Mn, Co and Ni between grades are not known.

Table 16. Back-transformed least squares means (µg/L) with 95% confidence interval of metals in whole blood in Swedish adolescents per grade and gender*.

Metal	Boys				Girls					
	Grade 5 n=165	Grade 8 n=179	Grade 11 n=136	p-value, grade within boys	Grade 5 n=165	Grade 8 n=231	Grade 11 n=222	p-value, grade within girls	p-value, overall effect of gender	p-value, overall effect of grade
Cd	0.09[a] (0.08-0.10)	0.12[b] (0.11-0.13)	0.14[b] (0.12-0.15)	0.001	0.10[a] (0.09-0.11)	0.13[b] (0.12-0.14)	0.17[c] (0.15-0.18)	0.001	0.009	0.001
Hg	0.70 (0.61-0.79)	0.73 (0.64-0.82)	0.93 (0.79-1.1)	0.056	0.70[a] (0.60-0.79)	0.62[ab] (0.55-0.69)	0.50[b] (0.44-0.56)	0.003	0.001	0.272
Pb	7.6 (7.0-8.2)	8.7 (8.0-9.3)	8.5 (7.9-9.2)	0.070	7.4[a] (6.8-8.1)	6.5[ab] (6.078-6.9)	6.5[b] (6.0-6.9)	0.028	0.001	0.769
Cr	0.56[a] (0.53-0.58)	0.58[ab] (0.56-0.60)	0.65[b] (0.62-0.67)	0.001	0.54[a] (0.52-0.57)	0.56[ab] (0.54-0.58)	0.60[b] (0.57-0.62)	0.014	0.009	0.001
Mn	9.4[a] (8.9-9.9)	11.1[b] (10.4-11.7)	9.5[a] (8.9-10.1)	0.001	11.0[ab] (10.3-11.6)	11.6[a] (11.0-12.2)	10.4[b] (9.9-11.0)	0.001	0.001	0.001
Co	0.10[a] (0.09-0.11)	0.14[b] (0.13-0.15)	0.08[c] (0.07-0.09)	0.001	0.12[a] (0.11-0.13)	0.16[b] (0.15-0.17)	0.14[ab] (0.13-0.15)	0.002	0.001	0.001
Ni	0.60[a] (0.57-0.62)	0.65[b] (0.62-0.67)	0.61[ab] (0.59-0.64)	0.009	0.61 (0.59-0.64)	0.63 (0.61-0.65)	0.64 (0.61-0.66)	0.392	0.566	0.016

*Differences between the least squares means were estimated by ANOVA (main factors included region, grade and gender, and the interactions between the factors). Effect of grade was calculated separately for each gender using the same factors as in the main module without gender effect. Different letters indicate significant differences between the grades within gender (P<0.05) according to Tukey's multiple comparison test.

Associations between metal concentrations in blood and smoking habits

In the present study, participants in grades 8 and 11 were asked about smoking habits (n=767). Significant associations were found for smoking habits and concentrations of Cd and Pb in blood (p=0.001 and p=0.009, respectively) (Figure 7), but not for the other metals. Blood concentrations of Cd were highest in current smokers and lowest in never-smokers. This was expected as smoking is a well-known exposure route for Cd; these results are in agreement with a previous study of Swedish adolescents [171]. Never-smokers had lower Pb concentrations in blood compared with current smokers (Figure 7).

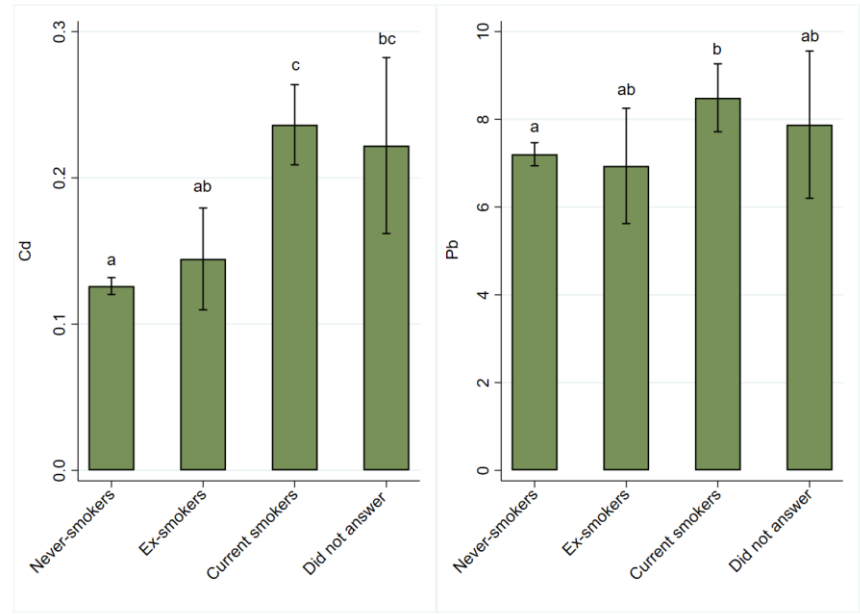

Figure 7. Concentrations of cadmium and lead (µg/L) in whole blood from Swedish adolescents per smoking category (back-transformed least squares means with 95% confidence intervals from the analysis of log values). Differences between the least squares means were estimated by ANOVA (main factors included region, gender and smoking). Different letters indicate significant differences between groups (P<0.05) according to Tukey's multiple comparison test. Number of observations in each group: never-smokers (n=625), ex-smokers (n=23), current smokers (n=101), did not answer (n=18).

Arsenic

Concentrations of As species were measured in individual urine samples from 74 girls and 49 boys (grade 5, n=22; grade 8, n=51; grade 11, n=50) (Table 17). All individuals selected for As analysis lived in households without a municipal water supply, i.e. with drinking water from a private or jointly owned well. Concentrations of As species were also measured in 42 pooled urine samples (Table 18). All individual samples had detectable concentrations of DMA and AB in urine. Total As was detected in 86% of the individual samples, and inorganic As and MMA in approximately two thirds (Table 17). All 42 pools had detectable concentrations of total As, DMA and MMA. Inorganic As was detected in 41 of 42 pools, and MMA in 39 pools (Table 18).

The profile of inorganic As and its metabolites in adult human urine has been reported to be 10–30% inorganic As, 10–20% MMA and 60–80% DMA [184]. In the present study, the profile in adolescents was found to be similar to this, although there are indications that As metabolism differs between children and adults [185].

In the present study concentrations of As are reported in µg As per kg urine (equal to density-adjustment) in contrast to many other studies which use creatinine-adjusted concentrations. This approach was chosen because creatinine-adjustment could be problematic for As biomonitoring, as urinary creatinine is associated with concentrations of As metabolites in urine [186]. Even though there is good agreement between adjusted and unadjusted As [187], a standard method for reporting As is still to be established. It should be noted that for the determination of inorganic As in the urine reference materials there is a 15–19% bias (Annex A1.4, Table A1.4.1), which may lead to an overestimation of inorganic As in studied samples.

Table 17. Concentrations of arsenic species in individual urine samples (µg/kg) from Swedish adolescents (n=123). Concentrations below LOD were replaced by LOD/√2 in the calculations.

As species	LOD	% of samples <LOD	Mean	SD	5th percentile	Median	95th percentile
Total As*	2.2	14	34.5	65.5	<LOD	13.2	163.6
Inorganic As	0.2	31	0.5	0.4	<LOD	0.5	1.2
Monomethyl arsenate (MMA)	0.2	36	0.5	0.4	<LOD	0.5	1.4
Dimethyl arsinate (DMA)	0.4	0	3.1	2.6	0.6	2.4	7.0
Sum of inorganic As, MMA and DMA			4.1	3.0	0.9	3.5	8.8
Arsenobetaine (AB)	0.4	0	27.0	57.1	0.4	8.7	126.2

*n=122.

Table 18. Concentrations of arsenic species in pooled urine samples (µg/kg) from Swedish adolescents (n=42 pools). Concentrations below LOD were replaced by LOD/√2 in the calculations.

As species	LOD	% of samples <LOD	Mean	SD	5th percentile	Median	95th percentile
Total As	2.2	0	40.7	18.9	16.5	39.1	70.7
Inorganic As	0.2	2	0.6	0.2	0.5	0.6	0.8
Monomethyl arsenate (MMA)	0.2	7	0.6	0.2	<LOD	0.6	0.8
Dimethyl arsinate (DMA)	0.4	0	3.8	1.6	2.5	3.5	5.5
Sum of inorganic As, MMA and DMA			5.0	1.8	3.5	4.7	7.0
Arsenobetaine (AB)	0.4	0	29.3	14.6	11.4	26.6	59.0

The concentrations of total As in the present study were higher than in an American study, where the median of unadjusted total urinary As among adolescents (12–19 years, NHANES data from 2013–2014) was 5.3 µg/L [145]. This is much lower than the medians of 13 and 39 µg/kg measured in individual and pooled samples, respectively, in the present study (Table 17 and Table 18). The reasons for this difference are unknown, but could be due to a higher consumption of fish in Sweden. It should be underlined that although a high consumption of fish yields high concentrations of total As, it does not necessarily cause elevated exposure to toxic inorganic As. This is because AB, which is supposed to be less harmful, is the major As species in many seafoods [90]. In line with such speculation, the sums of inorganic As, DMA and MMA in the present study (medians 3.5 µg/kg and 4.7 µg/kg in individual and pooled samples, respectively) were comparable to that measured in the adolescents from the U.S. (median concentration: of 4.4 µg/L) [145]. Concentrations of DMA in the adolescents from the U.S. (median 3.2 µg/L) were also similar to those in our study. In Flemish adolescents (14–15 years), the geometric means of the sum of inorganic As and major metabolites (i.e. As(III), As(V)), MMA and DMA) in urine was 4.8 µg/L [188]. The geometric mean of the sum of inorganic As, DMA

and MMA in urine from Swedish children from Lessebo municipality was 6.1 µg/L [164]. In the present study, the corresponding values from 7 municipalities were 4.8 µg/kg in the pooled samples and 3.2 µg/kg in the individual samples.

Differences in arsenic concentrations between genders and grades

Boys had significantly higher concentrations of total As and AB in urine compared to girls in both individual (Table 19) and pooled samples (Table 20). Other studies have also suggested that males have higher levels of total As in urine than females [189, 190]. These gender-associated differences have been explained by gender-associated differences in As metabolism [191].

Concentrations of all measured As species in the individual urine samples showed no difference ($p>0.05$) between grades (results not shown). However, the concentrations of total As and AB were lower in pooled samples from older individuals (Table 20). When differences between grades were estimated for each gender separately, grade appeared to be associated with lower concentrations of total As only in girls (Table 20). This observation is in agreement with a previous study, where total As concentrations in urine declined with age in girls (from 7 to 17 years) [190].

Neither gender- nor grade-associated differences were observed for inorganic As, DMA and MMA in the present study.

Table 19. Back-transformed least squares means (µg/kg) with 95% confidence intervals of arsenic species in individual urine samples (µg/kg) from Swedish adolescents per gender*.

As species	Boys n=49	Girls n=74	p-value, gender
Total As	19 (11-26)	10 (6.9-13)	0.020
Inorganic As	0.43 (0.33-0.52)	0.37 (0.30-0.44)	0.367
MMA	0.43 (0.32-0.53)	0.35 (0.28-0.42)	0.222
DMA	2.6 (2.0-3.2)	2.1 (1.7-2.5)	0.135
AB	11 (5.5-17)	5.5 (3.3-7.6)	0.034

*Differences between the least squares means were estimated by ANOVA (main factors included grade and gender).

Associations between arsenic concentrations and country of birth

Concentrations of As species in pooled urine samples in relation to country of birth of participants and their mothers are shown in Table 21. No significant differences were observed between the groups.

Table 20. Back-transformed least squares means (µg/kg) with 95% confidence intervals of arsenic species in pooled urine samples from Swedish adolescents per grade and gender*.

As species	Boys			p-value, grade within boys	Girls			p-value, grade within girls	p-value, overall effect of gender	p-value, overall effect of grade
	Grade 5 5 pools	Grade 8 5 pools	Grade 11 5 pools		Grade 5 5 pools	Grade 8 5 pools	Grade 11 5 pools			
Total As	43 (27-59)	63 (39-86)	35 (22-48)	0.133	43[a] (33-53)	36[ab] (28-44)	27[b] (21-33)	0.055	0.051	0.032
Inorganic As	0.44 (0.28-0.60)	0.69 (0.45-0.94)	0.63 (0.41-0.86)	0.223	0.69 (0.59-0.79)	0.58 (0.49-0.66)	0.63 (0.54-0.73)	0.283	0.487	0.588
MMA	0.40 (0.26-0.54)	0.62 (0.40-0.83)	0.60 (0.39-0.81)	0.198	0.63 (0.50-0.75)	0.53 (0.43-0.63)	0.55 (0.44-0.66)	0.483	0.561	0.589
DMA	3.4 (2.8-4.0)	3.7 (3.0-4.4)	3.3 (2.7-3.9)	0.651	4.8 (3.3-6.3)	3.0 (2.1-3.9)	3.1 (2.2-4.1)	0.101	0.839	0.230
AB	34 (21-46)	45 (28-629)	23 (14-32)	0.089	28 (22-35)	28 (22-35)	19 (14-23)	0.041	0.041	0.005

*Differences between the least squares means were estimated by ANOVA (main factors included grade and gender). Effect of grade was calculated separately for each gender using grade as the only factor in the model. Different letters indicate significant differences between the grades (P<0.05) according to Tukey's multiple comparison test.

Table 21. Back-transformed least squares means (µg/kg) with 95% confidence intervals of arsenic species in pooled urine samples from Swedish adolescents in relation to country of birth.*

As species	Country of birth of participant/mother of the participant			p-value
	Sweden/Sweden 4 pools	Sweden/Medium or low-income country** 4 pools	Medium or low-income country /Medium or low-income country** 4 pools	
Total As	30 (14-47)	29 (13-45)	33 (15-51)	0.943
Inorganic As	0.50 (0.37-0.63)	0.59 (0.43-0.74)	0.73 (0.54-0.92)	0.185
MMA	0.28 (0.12-0.43)	0.58 (0.26-0.90)	0.80 (0.36-1.24)	0.069
DMA	3.2 (2.3-4.0)	3.8 (2.8-4.9)	4.5 (3.3-5.7)	0.234
AB	23 (4.3-42)	20 (3.7-36)	18 (3.3-32)	0.895

*Differences between the least squares means were estimated by ANOVA (main factors included country of birth of participant/mother) followed by Tukey's multiple comparison test.
**Classification based on UN, 2014 [120].

Phthalate metabolites and phenolic compounds

Density-adjusted levels of phthalate metabolites and phenolic compounds in urine are presented in Table 22. Full summary statistics per region, grade and gender are provided in Appendix 2.

Table 22. Density-adjusted urine concentrations of phthalate metabolites and phenolic compounds (ng/mL) in Swedish adolescents (n=1,095-1,104). Concentrations below LOD were replaced by LOD/√2 in the calculations.

Compound	LOD	% of samples <LOD	Mean	SD	5th percentile	Median	95th percentile
Phthalate metabolites							
MEP	0.2	0	112.3	395.8	9.5	32.8	353.4
MnBP	1.6	0.1	52.0	48.1	15.0	39.7	118.9
MBzP	0.2	0.2	13.1	19.5	1.3	6.5	44.8
MEHP	0.3	1.6	2.5	9.0	0.6	1.6	5.9
5-OH-MEHP	0.1	0.2	11.7	36.9	3.1	7.7	26.0
5-oxo-MEHP	0.2	0.2	9.4	33.2	2.2	6.0	20.8
2-cx-MEHP	0.05	0	3.0	5.1	0.8	2.0	6.9
5-cx-MEPP	0.07	0	10.5	29.3	2.6	6.9	25.0
OH-MiNP	0.05	0	11.2	39.4	1.2	3.7	37.3
oxo-MiNP	0.05	0	4.5	13.6	0.7	1.8	13.7
cx-MiNP	0.05	0	16.1	59.8	2.1	5.9	48.9
cx-MiDP	0.1	5.4	0.6	1.1	0.1	0.4	1.5
OH-MPHP	0.08	0.8	1.9	5.1	0.4	1.0	4.7
DiNCH metabolites							
oxo-MiNCH	0.08	0.7	4.5	34.7	0.3	1.0	12.5
cx-MiNCH	0.1	2.2	3.1	14.4	0.2	0.8	10.9
OH-MiNCH	0.1	3.3	3.5	22.3	0.2	0.8	11.3
Phosphorous flame retardants							
DPP	0.07	0	2.7	3.3	0.7	1.9	6.2
DBP	0.05	15	0.2	0.5	<LOD	0.1	0.6
BBOEP	0.05	54	0.1	0.1	<LOD	0.1	0.3
Bisphenols							
BPA	0.2	7	1.4	2.6	0.3	0.9	3.8
BPS	0.03	7	0.3	1.6	0.04	0.1	0.7
4,4-BPF	0.03	21	0.8	7.1	<LOD	0.1	2.1
PAH metabolites							
2-OH-PH	0.1	32	0.3	0.4	<LOD	0.2	0.9
1-HP	0.1	64	0.1	0.2	<LOD	<LOD	0.3
Insecticide metabolites							
TCP	0.07	0.1	1.8	2.9	0.5	1.2	4.5
3-PBA	0.05	3	0.4	0.6	0.1	0.3	1.1
Other							
TCS	0.1	19	3.8	43.8	<LOD	0.3	2.1
BHA	0.02	14	12.0	36.6	<LOD	0.9	63.0
BP-3	0.2	15	11.6	74.0	<LOD	0.8	26.6

Phthalate and DiNCH metabolites

In the present study, MnBP (i.e. sum of MnBP and MiBP) was found to be the phthalate metabolite with highest median concentration, followed by MEP (Table 22). The highest detected MEP concentration of 8304 ng/mL was almost double the second highest concentration of 5241 ng/mL. The highest MnBP concentration was 916 ng/mL (the second highest 370 ng/mL). In previous Swedish studies on adolescents and first-time mothers, MnBP and/or MEP were also found to be the phthalate metabolites with the highest urine concentrations [129, 144].

The highest quantified MEHP concentration was 290 ng/mL (the second highest 41 ng/mL). Generally, individuals with high levels of MEHP in their urine also had high levels of the other DEHP metabolites (5-OH-MEHP, 5oxo-MEHP, 2cx-MEHP and 5cx-MEPP). Levels of these metabolites in urine are often considered as biomarkers of DEHP exposure [192]. The German Human Biomonitoring Commission has estimated that there is no risk of adverse health effects at urine concentrations of 5-OH-MEHP and 5oxo-MEHP (sum) below 500 ng/mL in children (6–13 years), 300 ng/mL in women of child-bearing age and below 750 ng/mL in the remaining general population [30]. In the present study, only two urine samples (0.2%) exceeded 300 ng/mL and the 95th percentile of this sum was 46.9 ng/mL. For 5cx-MEPP, the German Human Biomonitoring Commission has estimated that there is no risk of adverse health effects at urine concentrations below 1.8 mg/L (1800 ng/mL) in children and below 2.8 mg/L (2800 ng/mL) in adults [30]. None of the samples in the present study had concentrations exceeding these values, and the maximum concentration was 912 ng/mL.

The concentrations of OH-MiNP, oxo-MiNP, cx-MiNP, cx-MiDP and OH-MPHP varied greatly between individuals, although no extreme values were observed.

Overall, the range of observed concentrations of phthalate metabolites in the present study was similar to that in a recent report on Swedish adolescents (17–21 years old) from Skåne [144], and in a study on Swedish first-time mothers [129].

The DiNCH metabolites (oxo-MiNCH, cx-MiNCH and OH-MiNCH) were detected in >99% of the participants. The observed median concentration of oxo-MiNCH was close to that reported in adolescents by Norén et al. [144] (1.0 vs 0.7 ng/mL). The median concentration in Swedish preschool children was reported to be 1.8 ng/mL [103]. Increasing temporal trends in DiNCH exposure were observed in studies on Swedish first-time mothers from 2009 to 2014 [129], German young adults from 1999 to 2017 [193] and U.S. adolescents from 2011 to 2016 [194]. Increasing trends of DiNCH-exposure are expected as this chemical has been introduced as a substitute for phthalates. The German Human Biomonitoring Commission has estimated that there is no risk of adverse health effects at urine concentrations of DiNCH metabolites (sum of cx-MiNCH and OH-MiNCH) below 3 mg/L (3000 ng/mL) in children and below 4.5 mg/L (4500 ng/mL) in adults [30]. None of the samples in the present study had concentrations exceeding 3000 ng/mL and the highest analysed concentration was 1063 ng/mL.

Phosphorus flame retardants

Among the phosphorus-based flame retardants, the highest concentrations and the highest detection frequency was observed for di-phenylphosphate (DPP). The median DPP concentration (1.9 ng/mL) was higher than the median level in adolescents from Skåne (0.95 ng/mL) and first-time mothers in

Uppsala (1.0 ng/mL), whereas the concentrations of DBP and BBOEP were in a similar range [129, 144]. DPP is a urinary metabolite of triphenyl phosphate, and often used as a biomarker for triphenyl phosphate exposure.

Bisphenols

BPA and BPS were detected in 93% of the samples. The detection frequency for 4,4-BPF was lower (Table 22). These results are in agreement with those from a study of adolescents from Skåne [144]. The median concentration of 4,4-BPF was lower in the present study (0.1 ng/mL) compared to the median concentration of 2.2 ng/mL reported by Norén et al. [144]. The German Human Biomonitoring Commission has estimated that there is no risk of adverse health effects of BPA below urine levels of 100 ng/mL in children [30]. In our study, the 95th percentile of BPA (3.8 ng/mL) was much below this value (Table 22). None of the samples had concentrations exceeding 100 ng/mL.

Polycyclic aromatic hydrocarbons metabolites

The PAH metabolites hydroxyphenanthrene (2-OH-PH) and hydroxypyrene (1-HP) were detected in 68% and 36% of the samples, respectively (Table 22). The median level of 2-OH-PH was slightly higher in the present study (0.2 ng/mL) than median levels reported in Swedish adolescents (0.13 ng/mL) [144] and first-time mothers (0.12 ng/mL) [195].

Insecticide metabolites

Metabolites of chlorpyrifos (TCP) and pyrethroids (3-PBA) were detected in almost all samples (Table 22). Levels were similar to those in Swedish first-time mothers [195]. Another population of Swedish adolescents had similar 3-PBA levels, but somewhat higher TCP levels, in urine [144]. The median TCP concentration in the present study was lower (1.2 vs 1.8–3.1 ng/mL) than that found in studies on adult urine in 2010 or thereabouts [196, 197]. This indicates that exposure to chlorpyrifos has decreased since the EU restricted uses on a number of crops, and many maximum residue levels were lowered in 2016. The concentrations of 3-PBA in the present study were similar to those reported for other Swedish populations [196, 197] as well as for adolescents in the U.S. [198] and children in Germany [199]. Previous estimates of pyrethroid exposure from food have also indicated that it has been at a fairly constant low level during recent years [114].

Other phenolic compounds

The concentrations and detection frequency of the antibacterial substance triclosan (TCS) (Table 22) were comparable to these reported for adolescents from Skåne and first-time mothers in Uppsala [144, 195]. The concentration of TCS in urine below which, according to an assessment by the German Human Biomonitoring Commission, there is no risk of adverse health effects, was estimated to be 2000 ng/mL in children [30]. In our study, the 95th percentile of TCS was almost 1000 times lower than this value. None of the samples had concentrations exceeding 2000 ng/mL.

To the best of our knowledge, the concentration of BHA, a commonly used food preservative, has not previously been studied in Swedish adolescents. The median level of BHA in the present study (Table 22) was slightly higher than in Swedish first-time mothers (0.9 vs 0.5 ng/mL) [195].

BP-3, a UV filter used in the cosmetic industry, was detected in 85% of the samples with a median of 0.8 ng/mL (Table 22), which is somewhat lower than the 1.9 ng/mL reported in Swedish first-time mothers [195] and 3.6 ng/mL reported for Belgian adolescents [200].

Differences in concentrations of phthalate metabolites and phenolic compounds between regions

Summary statistics of phthalate metabolites and phenolic compounds per region are presented in Appendix 2, Table A2.10. Concentrations of the phthalate metabolites MEP, MEHP, 5-OH-MEHP, 5-oxo-MEHP, 2-cx-MEHP, OH-MiNP, oxo-MiNP and OH-MPHP were similar in all regions ($p>0.05$). Concentrations of MnBP, MBzP, 5-cx-MEPP and cx-MiDP differed between regions, but the differences did not follow any consistent pattern (Figure 8). Among bisphenols, only BPA displayed region-associated differences being higher in the Örebro region than Gothenburg (Figure 8). Concentrations of all other substances, i.e. DiNCH metabolites (oxo-MiNCH, cx-MiNCH and OH-MiNCH), phosphorus flame retardants (DPP and DBP), PAH metabolites (2-OH-PH), insecticide metabolites (TCP and 3-PBA), TCS, BHA and BP-3 did not differ between regions.

Observed differences between regions were small and may be explained by chance or by differences in life-style or food consumption.

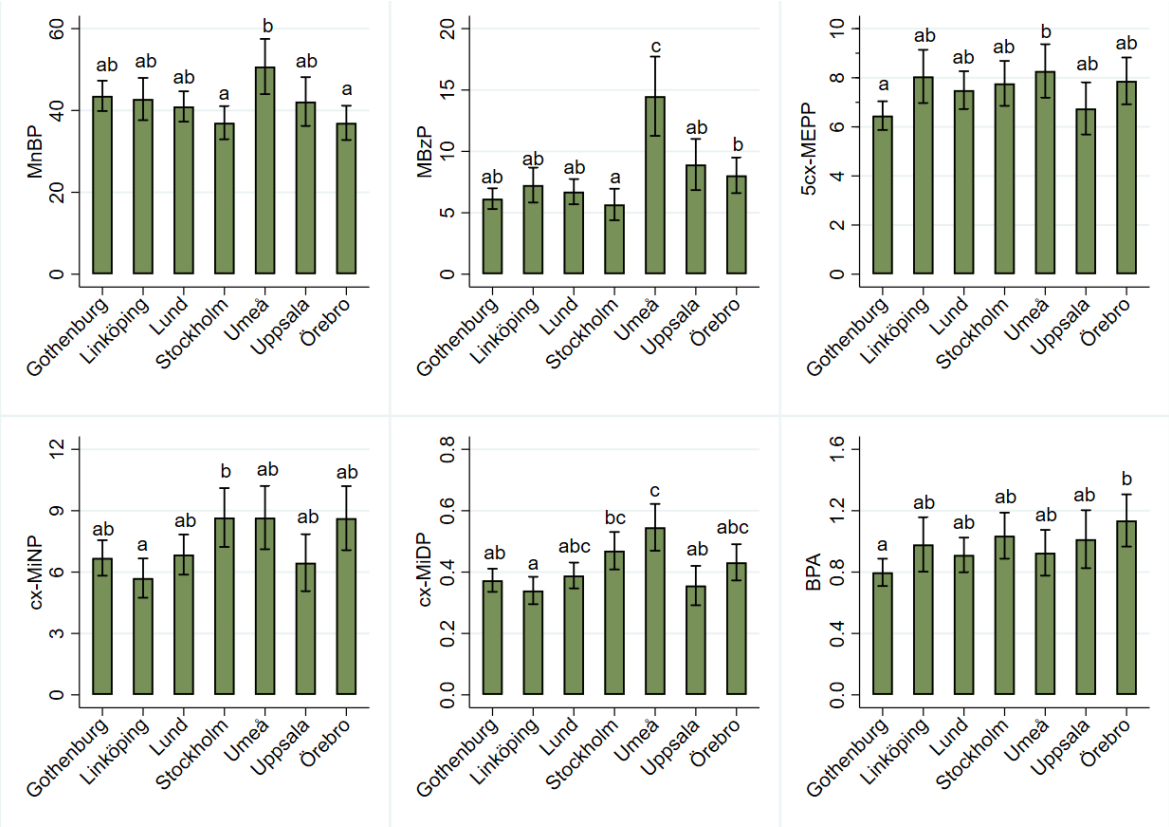

Figure 8. Density-adjusted urine concentrations of phthalate metabolites and bisphenol A (ng/mL) in Swedish adolescents per region (back-transformed least squares means with 95% confidence intervals from the analysis of log values). Differences between the least squares means were estimated by ANOVA (main factors included region, grade and gender, and the interactions between the factors). Different letters indicate significant differences between regions ($P<0.05$) according to Tukey's multiple comparison test. Number of observations per region: Gothenburg (n=221–222), Linköping (n=155–157), Lund (n=209–210), Stockholm (n=168), Umeå (n=130–131), Uppsala (n=84), Örebro (n=131–132).

Differences in phthalate metabolites and phenolic compound concentrations between genders and grades

Summary statistics of phthalate metabolites and phenolic compounds per gender and grade are presented in Appendix 2, Table A2.11 and A2.12. Higher concentrations of phthalates are usually observed in children than in adolescents and adults [201]. Less is known about gender-associated differences during adolescence. Larsson et al. [103], however, did not observe any significant age- or gender-associated differences in the levels of phthalates, non-phthalate plasticizers and bisphenols in urine from Swedish preschool children.

Overall, boys had significantly lower concentrations of most phthalate metabolites in urine than girls (Table 23). There were no consistent associations between grade and phthalate metabolite concentrations; the direction of the associations varied between compounds. Frederiksen et al. [202] observed a significant decrease in median urine levels of MnBP, MiBP, DEHP metabolites and DiNP metabolites with increasing age from 5–9 to 14–20 years in the Danish population. Similarly to the present study, the concentrations of MEP increased with age in the Danish population [202].

The concentrations of DiNCH metabolites were similar in boys and girls and increased with age, although for cx-MiNCH and OH-MiNCH this increase was observed in boys only (Table 23). Similarly, Fromme et al [102] did not detect any gender-related differences in German children.

The concentrations of both DPP and DBP were highest in boys in grade 5. The number of studies on DPP and DBP in urine is increasing, but the majority have been performed on adults. Similarly to a study in the U.S. [203], we found higher concentrations of DPP in girls than boys (Table 24).

In the present study, concentrations of BPA and 4,4-BPF were similar in boys and girls, whereas concentrations of BPS were higher in girls (Table 24). Among girls, the concentrations of BPA and BPS were lower in grade 8 compared with grades 5 and 11. Levels of 4,4-BPF were higher in older participants of both genders. Data on variability of bisphenols across ages and genders, especially among adolescents, are sparse. According to Frederiksen et al. [202], Danish children (5–9 years) excreted significantly higher amounts of BPA than adolescents (14–20 years) and the concentrations were similar in males and females.

The concentrations of the PAH metabolite 2-OH-PH were similar in boys and girls (Table 24). Levels were lower in grade 5 than in grades 8 and 11, but the differences were very small. In contrast, there was a tendency for higher concentrations of the PAH metabolites 3-hydroxyphenanthrene and 1-hydroxypyrene in younger children (6–11 years) compared with adolescents and adults (≥12 years) in a U.S. study [204].

Urine concentrations of the insecticide metabolite TCP were similar in boys and girls and were higher in girls in grade 11 compared to those in grade 8. No grade-associated differences were observed in TCP concentrations in boys (Table 24). The concentrations of 3-PBA were higher in girls than boys, probably due to higher consumption of fruits and vegetables by girls [139]. A lower 3-PBA concentration was observed in boys in grade 11 compared to boys in grade 5. Biomonitoring of pyrethroid insecticides in the general U.S. population demonstrated that children (6–11 years) had higher 3-PBA concentrations than adolescents and adults [198].

Overall, the concentrations of TCS, BHA and BP-3 were higher in girls (Table 24). Similarly, higher BP-3 concentrations were observed in females in Belgium, most likely due to the presence of BP-3 in

personal care products [200]. In contrast, the concentrations of TCS did not differ between genders in Canadian [205] and Danish [202] children, adolescents and adults.

The highest TCS, BHA and BP-3 concentrations were observed in the participants in grade 11. Danish adolescents (14–20 years) also had significantly higher urine levels of TCS and BP-3 than children (5–13 years) [202].

Table 23. Back-transformed least squares means (ng/mL, density-adjusted) with 95% confidence interval of phthalate and DiNCH metabolites in Swedish adolescents per grade and gender*.

Compound	Boys			p-value, grade within boys	Girls			p-value, grade within girls	p-value, overall effect of gender	p-value, overall effect of grade
	Grade 5 n=166	Grade 8 n=179-180	Grade 11 n=134-136		Grade 5 n=166-167	Grade 8 n=231-233	Grade 11 n=219-222			
Phthalate metabolites										
MEP	26[a] (22-31)	32[ab] (26-38)	42[b] (34-50)	0.001	34[a] (27-41)	54[b] (46-63)	70[b] (58-82)	0.001	0.001	0.001
MnBP	43[a] (39-47)	42[a] (38-46)	32[b] (28-35)	0.001	45 (40-50)	41 (38-45)	45 (41-50)	0.308	0.002	0.058
MBzP	7.8 (6.5-9.0)	7.4 (5.6-9.2)	6.2 (4.9-7.4)	0.456	8.5 (6.9-10)	7.0 (5.8-8.3)	9.4 (8.0-11)	0.018	0.010	0.741
MEHP	1.9[a] (1.7-2.1)	1.4[b] (1.3-1.6)	1.6[ab] (1.4-1.8)	0.005	1.9[ab] (1.7-2.1)	1.6[a] (1.44-1.74)	1.9[b] (1.7-2.2)	0.015	0.001	0.025
5-OH-MEHP	10[a] (8.9-11)	7.6[b] (6.7-8.4)	6.2[b] (5.5-7.0)	0.001	9.9[a] (8.8-11)	7.4[b] (6.7-8.1)	7.6[b] (6.8-8.4)	0.002	0.125	0.001
5oxo-MEHP	7.9[a] (7.0-8.8)	5.9[b] (5.3-6.6)	4.6[c] (4-0-5.1)	0.001	8.0[a] (7.0-8.9)	5.7[b] (5.2-6.2)	7.0[a] (6.2-7.7)	0.001	0.002	0.001
2cx-MEHP	2.6[a] (2.3-2.9)	2.0[b] (1.8-2.2)	1.6[c] (1.4-1.8)	0.001	2.5[a] (2.2-2.8)	2.0[b] (1.8-2.1)	2.2[ab] (2.0-2.4)	0.020	0.063	0.001
5cx-MEPP	8.9[a] (7.9-9.9)	6.6[b] (5.9-7.3)	5.4[b] (4.7-6.0)	0.001	9.7[a] (8.6-11)	6.8[b] (6.2-7.4)	7.6[b] (6.7-8.4)	0.001	0.001	0.001
OH-MiNP	4.2 (3.5-4.9)	4.3 (3.6-4.9)	4.5 (3.7-5.3)	0.635	5.1 (4.2-6.0)	4.6 (3.9-5.2)	5.6 (4.8-6.5)	0.101	0.001	0.001
oxo-MiNP	1.9 (1.6-2.2)	1.9 (1.7-2.2)	2.0 (1.7-2.3)	0.744	2.3[ab] (2.0-2.7)	2.1[a] (1.9-2.4)	2.7[b] (2.3-3.1)	0.031	0.001	0.082
cx-MiNP	6.2 (5.2-7.2)	6.4 (5.4-7.3)	6.8 (5.6-7.9)	0.798	7.9 (6.6-9.2)	7.4 (6.5-8.4)	8.7 (7.4-10)	0.180	0.001	0.258
cx-MiDP	0.42[a] (0.37-0.47)	0.35[ab] (0.31-0.40)	0.35[b] (0.30-0.39)	0.031	0.44 (0.39-0.50)	0.41 (0.37-0.46)	0.47 (0.41-0.52)	0.400	0.001	0.368
OH-MPHP	1.1 (0.97-1.3)	1.1 (1.0-1.3)	1.0 (0.89-1.2)	0.391	1.1 (0.98-1.3)	1.1 (0.99-1.2)	1.3 (1.1-1.4)	0.156	0.264	0.945
DiNCH metabolites										
oxo-MiNCH	1.0[a] (0.81-1.2)	1.3[ab] (1.1-1.5)	1.6[b] (1.2-1.9)	0.022	1.3[ab] (1.1-1.6)	1.2[a] (1.0-1.4)	1.7[b] (1.4-2.0)	0.040	0.056	0.003
cx-MiNCH	0.73[a] (0.59-0.87)	0.91[ab] (0.73-1.1)	1.2[b] (0.96-1.5)	0.012	0.96 (0.77-1.1)	0.94 (0.78-1.1)	1.2 (0.97-1.42)	0.283	0.076	0.005
OH-MiNCH	0.76[a] (0.60-0.91)	1.0[ab] (0.81-1.2)	1.2[b] (0.94-1.5)	0.022	0.99 (0.79-1.2)	0.89 (0.74-1.0)	1.2 (0.94-1.34)	0.175	0.301	0.010

*Differences between the least squares means were estimated by ANOVA (main factors included region, grade and gender, and the interactions between the factors). Effect of grade was estimated separately for each gender using the same factors as in the main model without gender effect. Different letters indicate significant differences between the grades (P<0.05) according to Tukey's multiple comparison test.

Table 24. Back-transformed least squares means (ng/mL, density-adjusted) with 95% confidence interval of phosphorous flame retardant, bisphenols, polycyclic aromatic hydrocarbons and insecticide metabolites, and other phenolic compounds in Swedish adolescents per grade and gender*.

Compound	Boys				Girls					
	Grade 5 n=166	Grade 8 n=180	Grade 11 n=136	p-value, grade within boys	Grade 5 n=167	Grade 8 n=232-233	Grade 11 n=221-222	p-value, grade within girls	p-value, overall effect of gender	p-value, overall effect of grade
Phosphorous	**Flame retardants**									
DPP	1.9[a] (1.7-2.1)	1.9a[b] (1.7-2.1)	1.7[b] (1.5-1.8)	0.029	2.0 (1.8-2.3)	2.1 (1.9-2.2)	2.2 (2.0-2.5)	0.267	0.001	0.644
DBP	0.19[a] (0.16-0.22)	0.12[b] (0.11-0.14)	0.12[b] (0.10-0.14)	0.001	0.17 (0.14-0.19)	0.17 (0.15-0.19)	0.18 (0.15-0.20)	0.752	0.001	0.008
Bisphenols										
BPA	0.91 (0.79-1.0)	0.87 (0.75-0.98)	0.99 (0.85-1.13)	0.764	1.1[a] (0.93-1.29)	0.78[b] (0.69-0.87)	1.1[a] (0.96-1.24)	0.001	0.247	0.006
BPS	0.13 (0.11-0.15)	0.12 (0.10-0.14)	0.14 (0.12-0.16)	0.325	0.16[a] (0.13-1.2)	0.14[b] (0.13-0.16)	0.20[ab] (0.17-0.22)	0.009	0.001	0.008
4,4-BPF	0.08[a] (0.06-0.09)	0.13[b] (0.10-0.16)	0.21[b] (0.15-0.26)	0.001	0.08[a] (0.07-0.11)	0.13[b] (0.10-0.15)	0.20[b] (0.15-0.25)	0.001	0.596	0.001
PAH metabolites										
2-OH-PH	0.15[a] (0.13-0.17)	0.19[b] (0.17-0.22)	0.20[ab] (0.17-0.22)	0.027	0.18[a] (0.15-0.21)	0.19[ab] (0.17-0.22)	0.22[b] (0.19-0.25)	0.005	0.167	0.001
Insecticide metabolites										
TCP	1.2 (1.1-1.4)	1.2 (1.1-1.4)	1.2 (1.1-1.3)	0.932	1.3[ab] (1.2-1.4)	1.2[a] (1.1-1.3)	1.4[b] (1.3-1.6)	0.012	0.109	0.212
3-PBA	0.30[a] (0.26-0.34)	0.26[ab] (0.23-0.29)	0.24[b] (0.21-0.27)	0.038	0.31 (0.27-0.35)	0.30 (0.27-0.33)	0.30 (0.26-0.34)	0.664	0.016	0.043
Other										
TCS	0.25[a] (0.20-0.30)	0.29[ab] (0.23-0.34)	0.38[b] (0.29-0.47)	0.017	0.31[a] (0.23-0.40)	0.39[ab] (0.32-0.46)	0.44[b] (0.36-0.53)	0.012	0.017	0.001
BHA	0.29[a] (0.18-0.41)	0.44[ab] (0.25-0.63)	0.65[b] (0.35-0.94)	0.020	0.90[a] (0.43-1.4)	1.9[b] (1.2-2.6)	2.3[b] (1.5-3.1)	0.001	0.001	0.001
BP-3	0.55[a] (0.41-0.70)	0.70[ab] (0.53-0.87)	0.91[b] (0.67-1.15)	0.036	0.75[a] (0.53-0.97)	1.9[b] (1.5-2.3)	2.5[b] (1.9-3.1)	0.001	0.001	0.001

*Differences between the least squares means were estimated by ANOVA (main factors included region, grade and gender, and the interactions between the factors). Effect of grade was estimated separately for each gender using the same factors as in the main model without gender effect. Different letters indicate significant differences between the grades (P<0.05) according to Tukey's multiple comparison test.

Concluding remarks

The present report summarizes blood and urine concentrations of contaminants in a unique, national representative study of Swedish adolescents with a geographic distribution from Boden in the north to Malmö in the south. The results provide an overview of the exposure to several contaminants in Swedish adolescents aged approximately 11–12, 14–15 and 17–18 years. A wide range of contaminants were investigated: substances the use of which has been restricted or banned but which still persist in the environment; substances that continue to be produced and used; and naturally occurring toxic metals and trace elements. Contaminants from all investigated substance groups (i.e. chlorinated and brominated POPs, PFAS, metals and metalloids, phthalates and phenolic compounds) could be quantified in samples from most participants in the study. The levels were generally comparable to those found in other studies and were within the expected range.

Full risk assessments of the observed concentrations were beyond the scope of this report. However, for some PCBs, PFOS, PFOA, mercury, lead, some phthalate and DiNCH metabolites, bisphenol A and triclosan, concentrations in the human body have been proposed below which it is unlikely that these contaminants would cause adverse health effects. Our results show that, based on current knowledge, the levels of these contaminants in most Swedish adolescents are not a health concern. However, a few percent of the participants had a higher exposure to PFOS or lead and single individuals had a higher exposure to PFOA, mercury or DEHP metabolites. This was especially pronounced for lead, where 7% and 13% of the participants had blood concentrations above the EFSA reference points for increased risks of chronic kidney disease in adults and developmental neurotoxicity in small children, respectively. This underlines the importance of further reducing lead exposure from all sources, not only food. In addition, we cannot exclude that some Swedish populations may have a higher exposure to contaminants than those included in *Riksmaten Adolescents 2016–17*.

Despite the heterogeneity of the study population, which represents a mix of individuals in different pubertal stages and growth phases, we found some gender- and age-related differences in contaminant concentrations. This suggests that there may be differences in exposure, uptake and/or elimination between genders and age groups.

Even though some regional differences were observed in concentrations of the investigated compounds, there was no consistent pattern. Some differences were statistically significant, but most were small and could be random findings. Thus, in most areas of Sweden exposure to the studied contaminants appears to be quite similar. However, a higher PFAS exposure was detected in adolescents in Ronneby municipality (region Lund) where the drinking water had previously been highly contaminated with these compounds.

There were substantial individual variations in levels of the investigated contaminants among Swedish adolescents. For risk assessment and risk management purposes, it is important to explore possible causes of such variation. Therefore, the Swedish Food Agency will continue to evaluate data from *Riksmaten Adolescents 2016–17* through further studies on the association between contaminant levels and factors such as diet, sociodemographics and lifestyle.

References

1. *Felter SP, Daston GP, Euling SY, Piersma AH, Tassinari MS. Assessment of health risks resulting from early-life exposures: Are current chemical toxicity testing protocols and risk assessment methods adequate? Crit Rev Toxicol. 2015;45(3):219-44.*
2. *Braun JM. Early-life exposure to EDCs: role in childhood obesity and neurodevelopment. Nat Rev Endocrinol. 2017;13(3):161-73.*
3. *Heindel JJ, Blumberg B, Cave M, Machtinger R, Mantovani A, Mendez MA, et al. Metabolism disrupting chemicals and metabolic disorders. Reprod Toxicol. 2017;68:3-33.*
4. *Al Osman M, Yang F, Massey IY. Exposure routes and health effects of heavy metals on children. Biometals. 2019;32(4):563-73.*
5. *Darnerud PO. Toxic effects of brominated flame retardants in man and in wildlife. Environ Int. 2003;29(6):841-53.*
6. *Larsen JC. Risk assessments of polychlorinated dibenzo- p-dioxins, polychlorinated dibenzofurans, and dioxin-like polychlorinated biphenyls in food. Mol Nutr Food Res. 2006;50(10):885-96.*
7. *Langer P. Persistent organochlorinated pollutants (PCB, DDE, HCB, dioxins, furans) and the thyroid--review 2008. Endocr Regul. 2008;42(2-3):79-104.*
8. *Lundqvist C, Zuurbier M, Leijs M, Johansson C, Ceccatelli S, Saunders M, et al. The effects of PCBs and dioxins on child health. Acta Paediatr Suppl. 2006;95(453):55-64.*
9. *Magliano DJ, Loh VH, Harding JL, Botton J, Shaw JE. Persistent organic pollutants and diabetes: a review of the epidemiological evidence. Diabetes Metab. 2014;40(1):1-14.*
10. *Baccarelli A, Pesatori AC, Consonni D, Mocarelli P, Patterson DG, Jr., Caporaso NE, et al. Health status and plasma dioxin levels in chloracne cases 20 years after the Seveso, Italy accident. Br J Dermatol. 2005;152(3):459-65.*
11. *Lee DH, Lind L, Jacobs DR, Jr., Salihovic S, van Bavel B, Lind PM. Associations of persistent organic pollutants with abdominal obesity in the elderly: The Prospective Investigation of the Vasculature in Uppsala Seniors (PIVUS) study. Environ Int. 2012;40:170-8.*
12. *Lee DH, Steffes MW, Sjodin A, Jones RS, Needham LL, Jacobs DR, Jr. Low dose of some persistent organic pollutants predicts type 2 diabetes: a nested case-control study. Environ Health Perspect. 2010;118(9):1235-42.*
13. *Ha MH, Lee DH, Son HK, Park SK, Jacobs DR, Jr. Association between serum concentrations of persistent organic pollutants and prevalence of newly diagnosed hypertension: results from the National Health and Nutrition Examination Survey 1999-2002. J Hum Hypertens. 2009;23(4):274-86.*
14. *Lind PM, Penell J, Salihovic S, van Bavel B, Lind L. Circulating levels of p,p'-DDE are related to prevalent hypertension in the elderly. Environ Res. 2014;129:27-31.*
15. *Lind PM, Salihovic S, Stubleski J, Karrman A, Lind L. Association of Exposure to Persistent Organic Pollutants With Mortality Risk: An Analysis of Data From the Prospective Investigation of Vasculature in Uppsala Seniors (PIVUS) Study. JAMA Netw Open. 2019;2(4):e193070.*
16. *Noren K, Meironyte D. Certain organochlorine and organobromine contaminants in Swedish human milk in perspective of past 20-30 years. Chemosphere. 2000;40(9-11):1111-23.*
17. *Gyllenhammar I, Glynn A, Fridén U, Cantillana T, Aune M, Lignell S. Levels of persistent halogenated organic pollutants (POP) in mother's milk from first-time mothers in Uppsala, Sweden: results from year 2015 and 2016, and temporal trends for the time period 1996-2016. Uppsala, Sweden: Swedish Food Agency; 2017.*
18. *Ålander J, Gyllenhammar I, Bergh A, Cantillana T, Lignell S, Ankarberg EH, et al. Concentrations of brominated flame retardants (HBB, PBEB, BTBPE, DBDPE, PBDEs and HBCD) in blood serum from firsttime mothers in Uppsala 1996-2017. Uppsala, Sweden: Swedish Food Agency; 2019.*
19. *Nost TH, Berg V, Hanssen L, Rylander C, Gaudreau E, Dumas P, et al. Time trends of persistent organic pollutants in 30 year olds sampled in 1986, 1994, 2001 and 2007 in Northern Norway: Measurements, mechanistic modeling and a comparison of study designs. Environ Res. 2019;172:684-92.*
20. *Bernes C. Persistent organic pollutants : a Swedish view of an international problem. Stockholm, Sweden: Swedish Environmental Protection Agency; 1998.*

21. *Kim M, Kim S, Yun S, Lee M, Cho B, Park J, et al. Comparison of seven indicator PCBs and three coplanar PCBs in beef, pork, and chicken fat. Chemosphere. 2004;54(10):1533-8.*

22. *National Food Agency. Swedish Market Basket Survey 2015 – per capita-based analysis of nutrients and toxic compounds in market baskets and assessment of benefit or risk. Uppsala, Sweden; 2017.*

23. *Darnerud PO, Atuma S, Aune M, Bjerselius R, Glynn A, Grawe KP, et al. Dietary intake estimations of organohalogen contaminants (dioxins, PCB, PBDE and chlorinated pesticides, e.g. DDT) based on Swedish market basket data. Food Chem Toxicol. 2006;44(9):1597-606.*

24. *Bjermo H, Darnerud PO, Lignell S, Pearson M, Rantakokko P, Nalsen C, et al. Fish intake and breastfeeding time are associated with serum concentrations of organochlorines in a Swedish population. Environ Int. 2013;51:88-96.*

25. *Grassi P, Fattore E, Generoso C, Fanelli R, Arvati M, Zuccato E. Polychlorobiphenyls (PCBs), polychlorinated dibenzo-p-dioxins (PCDDs) and dibenzofurans (PCDFs) in fruit and vegetables from an industrial area in northern Italy. Chemosphere. 2010;79(3):292-8.*

26. *Knobeloch L, Turyk M, Imm P, Anderson H. Polychlorinated biphenyls in vacuum dust and blood of residents in 20 Wisconsin households. Chemosphere. 2012;86(7):735-40.*

27. *Abb M, Breuer JV, Zeitz C, Lorenz W. Analysis of pesticides and PCBs in waste wood and house dust. Chemosphere. 2010;81(4):488-93.*

28. *EFSA. EFSA Panel on Contaminants in the Food Chain (CONTAM). Scientific Opinion on the risk for animal and human health related to the presence of dioxins and dioxin-like PCBs in feed and food. The EFSA Journal. 2018;16(11):5333.*

29. *EFSA. Opinion of the scientific panel on contaminants in the food chain on a request from the commission related to the presence of non-dioxin-like polychlorinated biphenyls (PCB) in feed and food. The EFSA journal. 2005;284:1-137.*

30. *Apel P, Angerer J, Wilhelm M, Kolossa-Gehring M. New HBM values for emerging substances, inventory of reference and HBM values in force, and working principles of the German Human Biomonitoring Commission. Int J Hyg Environ Health. 2017;220(2 Pt A):152-66.*

31. *Kommission H. Human-Biomonitoring-(HBM)-Werte für Polychlorierte Biphenyle (PCB) im Blut. Bundesgesundheitsbl GesundheitsforschGesundheitsschutz. 2012;55(8):1069–70.*

32. *Koureas M, Rousou X, Haftiki H, Mouchtouri VA, Rachiotis G, Rakitski V, et al. Spatial and temporal distribution of p,p'-DDE (1dichloro2,2bis (pchlorophenyl) ethylene) blood levels across the globe. A systematic review and meta-analysis. Sci Total Environ. 2019;686:440-51.*

33. *World Health Organization & International Programme on Chemical Safety. Hexachlorobenzene : health and safety guide. https://apps.who.int/iris/handle/10665/42018 (access date 10 Feb 2020); 1998.*

34. *JMPR (Joint Meeting on Pesticide Residues). DDT. Report of the Joint Meeting of the WHO Panel of Experts on the Pesticide Residues in Food and the WHO Core Assessment Group. WHO, Geneva. pp 59-63. Geneva, Switzerland.: WHO; 2001.*

35. *Alaee M, Arias P, Sjodin A, Bergman A. An overview of commercially used brominated flame retardants, their applications, their use patterns in different countries/regions and possible modes of release. Environ Int. 2003;29(6):683-9.*

36. *EFSA. EFSA Panel on Contaminants in the Food Chain (CONTAM); Scientific Opinion on Hexabromocyclododecanes (HBCDDs) in Food. The EFSA Journal. 2011;9(7):2296.*

37. *EFSA. EFSA Panel on Contaminants in the Food Chain (CONTAM); Scientific Opinion on Polybrominated Diphenyl Ethers (PBDEs) in Food. The EFSA Journal. 2011;9(5):2156.*

38. *Buck RC, Franklin J, Berger U, Conder JM, Cousins IT, de Voogt P, et al. Perfluoroalkyl and polyfluoroalkyl substances in the environment: terminology, classification, and origins. Integr Environ Assess Manag. 2011;7(4):513-41.*

39. *Krafft MP, Riess JG. Selected physicochemical aspects of poly- and perfluoroalkylated substances relevant to performance, environment and sustainability-part one. Chemosphere. 2015;129:4-19.*

40. *Kissa E. Fluorinated surfactants and repellents. 2nd Edition. Surfactant Science Series. Volume 97 ed: Marcel Dekker Inc., New York; 2001.*

41. *EFSA. EFSA Panel on Contaminants in the Food Chain (CONTAM); Risk to human health related to the presence of perfluorooctane sulfonic acid and perfluorooctanoic acid in food. The EFSA Journal. 2018;16:5194.*

42. *Bogdanska J, Borg D, Bergstrom U, Mellring M, Bergman A, DePierre J, et al. Tissue distribution of (14)C-labelled perfluorooctanoic acid in adult mice after 1-5 days of dietary exposure to an experimental dose or a lower dose that resulted in blood levels similar to those detected in exposed humans. Chemosphere. 2020;239:124755.*

43. *Bogdanska J, Borg D, Sundstrom M, Bergstrom U, Halldin K, Abedi-Valugerdi M, et al. Tissue distribution of (3)(5)S-labelled perfluorooctane sulfonate in adult mice after oral exposure to a low environmentally relevant dose or a high experimental dose. Toxicology. 2011;284(1-3):54-62.*

44. *Bogdanska J, Sundstrom M, Bergstrom U, Borg D, Abedi-Valugerdi M, Bergman A, et al. Tissue distribution of 35S-labelled perfluorobutanesulfonic acid in adult mice following dietary exposure for 1-5 days. Chemosphere. 2014;98:28-36.*

45. *Kelly BC, Ikonomou MG, Blair JD, Surridge B, Hoover D, Grace R, et al. Perfluoroalkyl contaminants in an Arctic marine food web: trophic magnification and wildlife exposure. Environ Sci Technol. 2009;43(11):4037-43.*

46. *Giesy JP, Kannan K. Global distribution of perfluorooctane sulfonate in wildlife. Environ Sci Technol. 2001;35(7):1339-42.*

47. *Gebbink WA, Glynn A, Darnerud PO, Berger U. Perfluoroalkyl acids and their precursors in Swedish food: The relative importance of direct and indirect dietary exposure. Environ Pollut. 2015;198:108-15.*

48. *DeWitt JC. Toxicological Effects of Perfluoroalkyl and Polyfluoroalkyl Substances: Springer International Publishing, Switzerland. Publisher: Humana Press, Cham; 2015.*

49. *Rappazzo KM, Coffman E, Hines EP. Exposure to Perfluorinated Alkyl Substances and Health Outcomes in Children: A Systematic Review of the Epidemiologic Literature. Int J Environ Res Public Health. 2017;14(7).*

50. *Ballesteros V, Costa O, Iniguez C, Fletcher T, Ballester F, Lopez-Espinosa MJ. Exposure to perfluoroalkyl substances and thyroid function in pregnant women and children: A systematic review of epidemiologic studies. Environ Int. 2017;99:15-28.*

51. *Vestergren R, Berger U, Glynn A, Cousins IT. Dietary exposure to perfluoroalkyl acids for the Swedish population in 1999, 2005 and 2010. Environ Int. 2012;49:120-7.*

52. *Gyllenhammar I, Berger U, Sundstrom M, McCleaf P, Euren K, Eriksson S, et al. Influence of contaminated drinking water on perfluoroalkyl acid levels in human serum--A case study from Uppsala, Sweden. Environ Res. 2015;140:673-83.*

53. *Sjogren P, Montse R, Lampa E, Salihovic S, van Bavel B, Lind L, et al. Circulating levels of perfluoroalkyl substances are associated with dietary patterns - A cross sectional study in elderly Swedish men and women. Environ Res. 2016;150:59-65.*

54. *Boone JS, Vigo C, Boone T, Byrne C, Ferrario J, Benson R, et al. Per- and polyfluoroalkyl substances in source and treated drinking waters of the United States. Sci Total Environ. 2019;653:359-69.*

55. *Banzhaf S, Filipovic M, Lewis J, Sparrenbom CJ, Barthel R. A review of contamination of surface-, ground-, and drinking water in Sweden by perfluoroalkyl and polyfluoroalkyl substances (PFASs). Ambio. 2017;46(3):335-46.*

56. *Sunderland EM, Hu XC, Dassuncao C, Tokranov AK, Wagner CC, Allen JG. A review of the pathways of human exposure to poly- and perfluoroalkyl substances (PFASs) and present understanding of health effects. J Expo Sci Environ Epidemiol. 2019;29(2):131-47.*

57. *Fromme H, Tittlemier SA, Volkel W, Wilhelm M, Twardella D. Perfluorinated compounds--exposure assessment for the general population in Western countries. Int J Hyg Environ Health. 2009;212(3):239-70.*

58. *Axmon A, Axelsson J, Jakobsson K, Lindh CH, Jonsson BA. Time trends between 1987 and 2007 for perfluoroalkyl acids in plasma from Swedish women. Chemosphere. 2014;102:61-7.*

59. *Glynn A, Benskin JP, Gyllenhammar I, Aune M, Cantillana T, Sandblom O. Temporal trends of perfluoroalkyl substances (PFAS) in individual serum samples from first-time mothers in Uppsala 1996-2016. Uppsala, Sweden: Swedish Food Agency; 2017.*

60. *Lindstrom AB, Strynar MJ, Libelo EL. Polyfluorinated compounds: past, present, and future. Environ Sci Technol. 2011;45(19):7954-61.*

61. *Akesson A, Barregard L, Bergdahl IA, Nordberg GF, Nordberg M, Skerfving S. Non-renal effects and the risk assessment of environmental cadmium exposure. Environ Health Perspect. 2014;122(5):431-8.*

62. *Akesson A, Julin B, Wolk A. Long-term dietary cadmium intake and postmenopausal endometrial cancer incidence: a population-based prospective cohort study. Cancer Res. 2008;68(15):6435-41.*

63. EFSA. EFSA Panel on Contaminants in the Food Chain (CONTAM); Statement on tolerable weekly intake for cadmium. The EFSA Journal. 2011;9:1975.

64. EFSA. Cadmium dietary exposure in the European population. The EFSA Journal. 2012;10(1):2551.

65. Ahn SC, Chang JY, Lee JS, Yu HY, Jung AR, Kim JY, et al. Exposure factors of cadmium for residents in an abandoned metal mine area in Korea. Environ Geochem Health. 2017;39(5):1059-70.

66. Andersen O, Nielsen JB, Nordberg GF. Nutritional interactions in intestinal cadmium uptake--possibilities for risk reduction. Biometals. 2004;17(5):543-7.

67. Bjermo H, Sand S, Nalsen C, Lundh T, Enghardt Barbieri H, Pearson M, et al. Lead, mercury, and cadmium in blood and their relation to diet among Swedish adults. Food Chem Toxicol. 2013;57:161-9.

68. EFSA. EFSA Panel on Contaminants in the Food Chain (CONTAM); Scientific Opinion on the risk for public health related to the presence of mercury and methylmercury in food. The EFSA Journal. 2012;10(12):2985.

69. Patrick L. Lead toxicity, a review of the literature. Part 1: Exposure, evaluation, and treatment. Altern Med Rev. 2006;11(1):2-22.

70. Bellinger DC. Very low lead exposures and children's neurodevelopment. Curr Opin Pediatr. 2008;20(2):172-7.

71. Bjorklund G, Crisponi G, Nurchi VM, Cappai R, Buha Djordjevic A, Aaseth J. A Review on Coordination Properties of Thiol-Containing Chelating Agents Towards Mercury, Cadmium, and Lead. Molecules. 2019;24(18).

72. EFSA. Panel on Contaminants in the Food Chain (CONTAM); Scientific Opinion on Lead in Food. The EFSA Journal. 2010;8:1570.

73. Florea AM, Busselberg D. Occurrence, use and potential toxic effects of metals and metal compounds. Biometals. 2006;19(4):419-27.

74. Stromberg U, Lundh T, Schutz A, Skerfving S. Yearly measurements of blood lead in Swedish children since 1978: an update focusing on the petrol lead free period 1995-2001. Occup Environ Med. 2003;60(5):370-2.

75. Stromberg U, Schutz A, Skerfving S. Substantial decrease of blood lead in Swedish children, 1978-94, associated with petrol lead. Occup Environ Med. 1995;52(11):764-9.

76. Wennberg M, Lundh T, Bergdahl IA, Hallmans G, Jansson JH, Stegmayr B, et al. Time trends in burdens of cadmium, lead, and mercury in the population of northern Sweden. Environ Res. 2006;100(3):330-8.

77. EFSA. NDA Panel (EFSA Panel on Dietetic Products, Nutrition and Allergies); Scientific opinion on dietary reference values for chromium The EFSA Journal. 2014;12(10):3845.

78. EFSA. EFSA Panel on Contaminants in the Food Chain (CONTAM); Scientific Opinion on the risks to public health related to the presence of chromium in food and drinking water. The EFSA Journal. 2014;12(3):3595.

79. Hoffman NJ, Penque BA, Habegger KM, Sealls W, Tackett L, Elmendorf JS. Chromium enhances insulin responsiveness via AMPK. J Nutr Biochem. 2014;25(5):565-72.

80. O'Neal SL, Zheng W. Manganese Toxicity Upon Overexposure: a Decade in Review. Curr Environ Health Rep. 2015;2(3):315-28.

81. Kullar SS, Shao K, Surette C, Foucher D, Mergler D, Cormier P, et al. A benchmark concentration analysis for manganese in drinking water and IQ deficits in children. Environ Int. 2019;130:104889.

82. EFSA. NDA Panel (EFSA Panel on Dietetic Products, Nutrition and Allergies); Scientific Opinion on Dietary Reference Values for manganese. The EFSA Journal. 2013;11(11):3419.

83. ATSDR (Agency for Toxic Substances and Disease Registry). Toxicological profile for cobalt. Atlanta, U.S.: U.S. Department of Health and Human Services, Public Health Service; 2004.

84. Paustenbach DJ, Tvermoes BE, Unice KM, Finley BL, Kerger BD. A review of the health hazards posed by cobalt. Crit Rev Toxicol. 2013;43(4):316-62.

85. Leyssens L, Vinck B, Van Der Straeten C, Wuyts F, Maes L. Cobalt toxicity in humans-A review of the potential sources and systemic health effects. Toxicology. 2017;387:43-56.

86. EFSA. EFSA Panel on Contaminants in the Food Chain (CONTAM); Scientific Opinion on the risks to animal and public health and the environment related to the presence of nickel in feed. The EFSA Journal. 2015;13(4):4074.

87. Tietz T, Lenzner A, Kolbaum AE, Zellmer S, Riebeling C, Gurtler R, et al. Aggregated aluminium exposure: risk assessment for the general population. Arch Toxicol. 2019;93(12):3503-21.

88. EFSA. Scientific Opinion of the Panel on Food Additives, Flavourings, Processing Aids and Food Contact Materials on a request from European Commission on Safety of aluminium from dietary intake. The EFSA Journal. 2008;754:1-34.

89. Smedley PL, Kinniburgh DG. A review of the source, behavior and distribution of arsenic in natural waters. Applied Geochemistry. 2002;17:517–68.

90. Hughes MF. Arsenic toxicity and potential mechanisms of action. Toxicol Lett. 2002;133(1):1-16.

91. Taylor V, Goodale B, Raab A, Schwerdtle T, Reimer K, Conklin S, et al. Human exposure to organic arsenic species from seafood. Sci Total Environ. 2017;580:266-82.

92. EFSA. Scientific opinion on dietary exposure to inorganic arsenic in the European population. The EFSA Journal. 2014;12(3):3597.

93. Kollander B, Sand S, Almerud P, Ankarberg EH, Concha G, Barregard L, et al. Inorganic arsenic in food products on the Swedish market and a risk-based intake assessment. Sci Total Environ. 2019;672:525-35.

94. Ankarberg EH, Fogelberg P, Gustafsson K, Nordenfors H, Bjerselius R. Inorganic arsenic in rice and rice products on the Swedish Market 2015. Uppsala, Sweden: Swedish Food Agency; 2015:Report no. 16/2015.

95. EFSA. EFSA Panel on Contaminants in the Food Chain (CONTAM); Scientific Opinion on Arsenic in Food. The EFSA Journal. 2009;7(10):1351.

96. Martin S, Griswold W. Human health effects of heavy metals. Environmental Science and Technology Briefs for Citizens. 2009;15:1-6.

97. Steinmaus C, Ferreccio C, Acevedo J, Yuan Y, Liaw J, Duran V, et al. Increased lung and bladder cancer incidence in adults after in utero and early-life arsenic exposure. Cancer Epidemiol Biomarkers Prev. 2014;23(8):1529-38.

98. Hauser R, Calafat AM. Phthalates and human health. Occup Environ Med. 2005;62(11):806-18.

99. European Commission (EC). Commission implementing decision (EU) 2017/1210 of 4 July 2017 on the identification of bis(2-ethylhexyl) phthalate (DEHP), dibutyl phthalate (DBP), benzyl butyl phthalate (BBP) and diisobutyl phthalate (DIBP) as substances of very high concert according to Article 57 (f) of Regulation (EC) No 1907/2006 of the European Parliament and of the Council. 2017.

100. Schutze A, Kolossa-Gehring M, Apel P, Bruning T, Koch HM. Entering markets and bodies: increasing levels of the novel plasticizer Hexamoll(R) DINCH(R) in 24 h urine samples from the German Environmental Specimen Bank. Int J Hyg Environ Health. 2014;217(2-3):421-6.

101. Bui TT, Giovanoulis G, Cousins AP, Magner J, Cousins IT, de Wit CA. Human exposure, hazard and risk of alternative plasticizers to phthalate esters. Sci Total Environ. 2016;541:451-67.

102. Fromme H, Schutze A, Lahrz T, Kraft M, Fembacher L, Siewering S, et al. Non-phthalate plasticizers in German daycare centers and human biomonitoring of DINCH metabolites in children attending the centers (LUPE 3). Int J Hyg Environ Health. 2016;219(1):33-9.

103. Larsson K, Lindh CH, Jonsson BA, Giovanoulis G, Bibi M, Bottai M, et al. Phthalates, non-phthalate plasticizers and bisphenols in Swedish preschool dust in relation to children's exposure. Environ Int. 2017;102:114-24.

104. Diamanti-Kandarakis E, Bourguignon JP, Giudice LC, Hauser R, Prins GS, Soto AM, et al. Endocrine-disrupting chemicals: an Endocrine Society scientific statement. Endocr Rev. 2009;30(4):293-342.

105. Radke EG, Braun JM, Meeker JD, Cooper GS. Phthalate exposure and male reproductive outcomes: A systematic review of the human epidemiological evidence. Environ Int. 2018;121(Pt 1):764-93.

106. Radke EG, Glenn BS, Braun JM, Cooper GS. Phthalate exposure and female reproductive and developmental outcomes: a systematic review of the human epidemiological evidence. Environ Int. 2019;130:104580.

107. van der Veen I, de Boer J. Phosphorus flame retardants: properties, production, environmental occurrence, toxicity and analysis. Chemosphere. 2012;88(10):1119-53.

108. Poma G, Glynn A, Malarvannan G, Covaci A, Darnerud PO. Dietary intake of phosphorus flame retardants (PFRs) using Swedish food market basket estimations. Food Chem Toxicol. 2017;100:1-7.

109. Vandenberg LN, Hauser R, Marcus M, Olea N, Welshons WV. Human exposure to bisphenol A (BPA). Reprod Toxicol. 2007;24(2):139-77.

110. Rubin BS. Bisphenol A: an endocrine disruptor with widespread exposure and multiple effects. J Steroid Biochem Mol Biol. 2011;127(1-2):27-34.

111. EFSA. CEF Panel (EFSA Panel on Food Contact Materials, Enzymes, Flavourings and Processing Aids); Scientific Opinion on the risks to public health related to the presence of bisphenol A (BPA) in foodstuffs: Executive summary. The EFSA Journal. 2015;13(1):3978.

112. EFSA. Scientific Opinion of the Panel on Contaminants in the Food Chain on a request from the European Commission on Polycyclic Aromatic Hydrocarbons in Food. The EFSA Journal. 2008;724:1-114.

113. Widenfalk A, Fohgelberg P. Resthalter av bekämpningsmedel (OP1) som indikator i livsmedel samt uppskattat intag och risk bland svenska konsumenter (inklusive barn). Uppsala, Sweden: Swedish Food Agency; 2012.

114. Mårtenson A. Resthalter av pyretroider/pyretriner som indikator i livsmedel samt uppskattat intag för svenska konsumenter (inklusive barn). 2014.

115. EFSA. Statement on the available outcomes of the human health assessment in the context of the pesticides peer review of the active substance chlorpyrifos. . The EFSA Journal. 2019;17:5809.

116. Axelstad M, Boberg J, Vinggaard AM, Christiansen S, Hass U. Triclosan exposure reduces thyroxine levels in pregnant and lactating rat dams and in directly exposed offspring. Food Chem Toxicol. 2013;59:534-40.

117. Ham J, Lim W, You S, Song G. Butylated hydroxyanisole induces testicular dysfunction in mouse testis cells by dysregulating calcium homeostasis and stimulating endoplasmic reticulum stress. Sci Total Environ. 2020;702:134775.

118. Yueh MF, Tukey RH. Triclosan: A Widespread Environmental Toxicant with Many Biological Effects. Annu Rev Pharmacol Toxicol. 2016;56:251-72.

119. Moraeus L, Lemming EW, Hursti UK, Arnemo M, Sipinen JP, Lindroos AK. Riksmaten Adolescents 2016-17: A national dietary survey in Sweden - design, methods, and participation. Food Nutr Res. 2018;62.

120. United Nations (UN). Country classification. Data sources, country classifications and aggregation methology, World Economic Situation and Prospects (WESP). http://www.un.org/en/development/desa/policy/wesp/wesp_current/2014wesp_country_classification.pdf (access date 4 Dec 2019); 2014.

121. Sveriges Kommuner och Regioner (SKR). Swedish Association of Local Authorities and Regions, Classification of Swedish municipalities 2017. https://webbutik.skr.se/sv/artiklar/kommungruppsindelning-2017.html (access date 10 Feb 2020); 2016.

122. Cole TJ, Lobstein T. Extended international (IOTF) body mass index cut-offs for thinness, overweight and obesity. Pediatr Obes. 2012;7(4):284-94.

123. Gasull M, Pumarega J, Kiviranta H, Rantakokko P, Raaschou-Nielsen O, Bergdahl IA, et al. Methodological issues in a prospective study on plasma concentrations of persistent organic pollutants and pancreatic cancer risk within the EPIC cohort. Environ Res. 2019;169:417-33.

124. Powley CR, George SW, Ryan TW, Buck RC. Matrix effect-free analytical methods for determination of perfluorinated carboxylic acids in environmental matrixes. Anal Chem. 2005;77(19):6353-8.

125. Bárány E, Bergdahl IA, Schütz A, Skerfving S, Oskarsson A. Inductively coupled plasma mass spectrometry for direct multi-element analysis of diluted human blood and serum. Journal of Analytical Atomic Spectrometry. 1997;12:1005-9.

126. 16802:2016 S-E. Foodstuffs. Determination of elements and their species. Determination of inorganic arsenic in foodstuffs of marine and plant origin by anion-exchange HPLC-ICP-MS. In: standard S, ed; 2016.

127. Sloth JJ, Larsen EH, Julshamn K. Determination of organoarsenic species in marine samples using gradient elution cation exchange HPLC-ICP-MS. Journal of Analytical Atomic Spectrometry. 2003;18:452–9.

128. 15763:2009 S-E. Foodstuffs - Determination of trace elements - Determination of arsenic, cadmium, mercury and lead in foodstuffs by inductively coupled plasma mass spectrometry (ICP-MS) after pressure digestion. In: standard S, ed; 2009.

129. Gyllenhammar I, Glynn A, Jonsson BA, Lindh CH, Darnerud PO, Svensson K, et al. Diverging temporal trends of human exposure to bisphenols and plastizisers, such as phthalates, caused by substitution of legacy EDCs? Environ Res. 2017;153:48-54.

130. Carnerup MA, Spanne M, Jonsson BA. Levels of N-methyl-2-pyrrolidone (NMP) and its metabolites in plasma and urine from volunteers after experimental exposure to NMP in dry and humid air. Toxicol Lett. 2006;162(2-3):139-45.

131. *Bjermo H, Ax E, Cantillana T, Glynn A, Darnerud PO, Lindroos AK. Miljöföroreningar i blod och urin och kopplingar till rapporterat matintag i Riksmaten 2010–11 – resultatsammanställning (in Swedish). Uppsala, Sweden: Swedish Food Agency; 2013.*

132. *Glynn A, Aune M, Darnerud PO, Cnattingius S, Bjerselius R, Becker W, et al. Determinants of serum concentrations of organochlorine compounds in Swedish pregnant women: a cross-sectional study. Environ Health. 2007;6:2.*

133. *Glynn AW, Wolk A, Aune M, Atuma S, Zettermark S, Maehle-Schmid M, et al. Serum concentrations of organochlorines in men: a search for markers of exposure. Sci Total Environ. 2000;263(1-3):197-208.*

134. *Glynn AW, Granath F, Aune M, Atuma S, Darnerud PO, Bjerselius R, et al. Organochlorines in Swedish women: determinants of serum concentrations. Environ Health Perspect. 2003;111(3):349-55.*

135. *Sahlstrom LM, Sellstrom U, de Wit CA, Lignell S, Darnerud PO. Brominated flame retardants in matched serum samples from Swedish first-time mothers and their toddlers. Environ Sci Technol. 2014;48(13):7584-92.*

136. *Darnerud PO, Lignell S, Aune M, Isaksson M, Cantillana T, Redeby J, et al. Time trends of polybrominated diphenylether (PBDE) congeners in serum of Swedish mothers and comparisons to breast milk data. Environ Res. 2015;138:352-60.*

137. *Bjermo H, Aune M, Cantillana T, Glynn A, Lind PM, Ridefelt P, et al. Serum levels of brominated flame retardants (BFRs: PBDE, HBCD) and influence of dietary factors in a population-based study on Swedish adults. Chemosphere. 2017;167:485-91.*

138. *Glynn A, Lignell S, Darnerud PO, Aune M, Halldin Ankarberg E, Bergdahl IA, et al. Regional differences in levels of chlorinated and brominated pollutants in mother's milk from primiparous women in Sweden. Environ Int. 2011;37(1):71-9.*

139. *Warensjö Lemming EW, Moraeus L, Petrelius Sipinen J, Lindroos AK. Riksmaten ungdom 2016-17. Livsmedelskonsumtion bland ungdomar i Sverige. Uppsala, Sweden: Swedish Food Agency; 2018.*

140. *Hardell E, Carlberg M, Nordstrom M, van Bavel B. Time trends of persistent organic pollutants in Sweden during 1993-2007 and relation to age, gender, body mass index, breast-feeding and parity. Sci Total Environ. 2010;408(20):4412-9.*

141. *Gallo MV, Deane GD, DeCaprio AP, Schell LM. Changes in persistent organic pollutant levels from adolescence to young adulthood. Environ Res. 2015;140:214-24.*

142. *Benskin JP, De Silva AO, Martin LJ, Arsenault G, McCrindle R, Riddell N, et al. Disposition of perfluorinated acid isomers in Sprague-Dawley rats; part 1: single dose. Environ Toxicol Chem. 2009;28(3):542-54.*

143. *Benskin JP, De Silva AO, Martin JW. Isomer profiling of perfluorinated substances as a tool for source tracking: a review of early findings and future applications. Rev Environ Contam Toxicol. 2010;208:111-60.*

144. *Norén E, Lindh C, Larsson E. Urin- och serumhalter av organiska miljöföroreningar hos ungdomar i Skåne Resultat från den femte delstudien 2017. . Lund, Sweden: Avdelningen för Arbets- och miljömedicin, Lunds universitet 2019.*

145. *Centers for Disease Control and Prevention (CDC). Forth National report on Human Exposure to Environmental Chemicals, Updated Tables, (January 2017). Atlanta, U.S.: U.S. Department of Health and Human Services; 2017.*

146. *Averina M, Brox J, Huber S, Furberg AS. Perfluoroalkyl substances in adolescents in northern Norway: Lifestyle and dietary predictors. The Tromso study, Fit Futures 1. Environ Int. 2018;114:123-30.*

147. *Bjermo H, Darnerud PO, Pearson M, Barbieri HE, Lindroos AK, Nalsen C, et al. Serum concentrations of perfluorinated alkyl acids and their associations with diet and personal characteristics among Swedish adults. Mol Nutr Food Res. 2013;57(12):2206-15.*

148. *Martin JW, Asher BJ, Beesoon S, Benskin JP, Ross MS. PFOS or PreFOS? Are perfluorooctane sulfonate precursors (PreFOS) important determinants of human and environmental perfluorooctane sulfonate (PFOS) exposure? J Environ Monit. 2010;12(11):1979-2004.*

149. *Karrman A, Langlois I, van Bavel B, Lindstrom G, Oehme M. Identification and pattern of perfluorooctane sulfonate (PFOS) isomers in human serum and plasma. Environ Int. 2007;33(6):782-8.*

150. *Gyllenhammar I, Benskin JP, Sandblom O, Berger U, Ahrens L, Lignell S, et al. Perfluoroalkyl Acids (PFAAs) in Children's Serum and Contribution from PFAA-Contaminated Drinking Water. Environ Sci Technol. 2019;53(19):11447-57.*

151. *Zhang Y, Beesoon S, Zhu L, Martin JW. Biomonitoring of perfluoroalkyl acids in human urine and estimates of biological half-life. Environ Sci Technol. 2013;47(18):10619-27.*

152. *Stubleski J, Salihovic S, Lind PM, Lind L, Dunder L, McCleaf P, et al. The effect of drinking water contaminated with perfluoroalkyl substances on a 10-year longitudinal trend of plasma levels in an elderly Uppsala cohort. Environ Res. 2017;159:95-102.*

153. *Ullah S, Alsberg T, Berger U. Simultaneous determination of perfluoroalkyl phosphonates, carboxylates, and sulfonates in drinking water. J Chromatogr A. 2011;1218(37):6388-95.*

154. *Filipovic M. PFOA och PFOS nya miljögifter i dricksvatten. LMNT-Nytt, Riksföreningen för lärarna i matematik, naturvetenskap och teknik. 2014SSN 1402-00412014:1 april. 2014.*

155. *Andersson EM, Scott K, Xu Y, Li Y, Olsson DS, Fletcher T, et al. High exposure to perfluorinated compounds in drinking water and thyroid disease. A cohort study from Ronneby, Sweden. Environ Res. 2019;176:108540.*

156. *Lewis RC, Johns LE, Meeker JD. Serum Biomarkers of Exposure to Perfluoroalkyl Substances in Relation to Serum Testosterone and Measures of Thyroid Function among Adults and Adolescents from NHANES 2011-2012. Int J Environ Res Public Health. 2015;12(6):6098-114.*

157. *Ji K, Kim S, Kho Y, Paek D, Sakong J, Ha J, et al. Serum concentrations of major perfluorinated compounds among the general population in Korea: dietary sources and potential impact on thyroid hormones. Environ Int. 2012;45:78-85.*

158. *Karrman A, Mueller JF, van Bavel B, Harden F, Toms LM, Lindstrom G. Levels of 12 perfluorinated chemicals in pooled australian serum, collected 2002-2003, in relation to age, gender, and region. Environ Sci Technol. 2006;40(12):3742-8.*

159. *Wong F, MacLeod M, Mueller JF, Cousins IT. Enhanced elimination of perfluorooctane sulfonic acid by menstruating women: evidence from population-based pharmacokinetic modeling. Environ Sci Technol. 2014;48(15):8807-14.*

160. *Han X, Nabb DL, Russell MH, Kennedy GL, Rickard RW. Renal elimination of perfluorocarboxylates (PFCAs). Chem Res Toxicol. 2012;25(1):35-46.*

161. *Koponen J, Winkens K, Airaksinen R, Berger U, Vestergren R, Cousins IT, et al. Longitudinal trends of per- and polyfluoroalkyl substances in children's serum. Environ Int. 2018;121(Pt 1):591-9.*

162. *Toms LM, Calafat AM, Kato K, Thompson J, Harden F, Hobson P, et al. Polyfluoroalkyl chemicals in pooled blood serum from infants, children, and adults in Australia. Environ Sci Technol. 2009;43(11):4194-9.*

163. *Nost TH, Vestergren R, Berg V, Nieboer E, Odland JO, Sandanger TM. Repeated measurements of per- and polyfluoroalkyl substances (PFASs) from 1979 to 2007 in males from Northern Norway: assessing time trends, compound correlations and relations to age/birth cohort. Environ Int. 2014;67:43-53.*

164. *Mattisson K, Tekavec E, Lundh T, Olsson A, Stroh E. Miljömedicinsk hälsoövervakning av barn i områden med förorenad mark från glasbruk – Lessebo kommun. Rapport nr 19/2018 Lund, Sweden: Avdelningen för Arbets- och miljömedicin, Lunds universitet; 2018.*

165. *Vrijens J, Leermakers M, Stalpaert M, Schoeters G, Den Hond E, Bruckers L, et al. Trace metal concentrations measured in blood and urine of adolescents in Flanders, Belgium: reference population and case studies Genk-Zuid and Menen. Int J Hyg Environ Health. 2014;217(4-5):515-27.*

166. *Rollin H, Mathee A, Levin J, Theodorou P, Wewers F. Blood manganese concentrations among first-grade schoolchildren in two South African cities. Environ Res. 2005;97(1):93-9.*

167. *Laohaudomchok W, Lin X, Herrick RF, Fang SC, Cavallari JM, Christiani DC, et al. Toenail, blood, and urine as biomarkers of manganese exposure. J Occup Environ Med. 2011;53(5):506-10.*

168. *Smith D, Gwiazda R, Bowler R, Roels H, Park R, Taicher C, et al. Biomarkers of Mn exposure in humans. Am J Ind Med. 2007;50(11):801-11.*

169. *Heitland P, Koster HD. Biomonitoring of 37 trace elements in blood samples from inhabitants of northern Germany by ICP-MS. J Trace Elem Med Biol. 2006;20(4):253-62.*

170. *Nisse C, Tagne-Fotso R, Howsam M, Richeval C, Labat L, Leroyer A. Blood and urinary levels of metals and metalloids in the general adult population of Northern France: The IMEPOGE study, 2008-2010. Int J Hyg Environ Health. 2017;220(2 Pt B):341-63.*

171. *Barany E, Bergdahl IA, Bratteby LE, Lundh T, Samuelson G, Schutz A, et al. Trace elements in blood and serum of Swedish adolescents: relation to gender, age, residential area, and socioeconomic status. Environ Res. 2002;89(1):72-84.*

172. *Baecklund M, Pedersen NL, Bjorkman L, Vahter M. Variation in blood concentrations of cadmium and lead in the elderly. Environ Res. 1999;80(3):222-30.*

173. Olsson IM, Bensryd I, Lundh T, Ottosson H, Skerfving S, Oskarsson A. Cadmium in blood and urine--impact of sex, age, dietary intake, iron status, and former smoking--association of renal effects. Environ Health Perspect. 2002;110(12):1185-90.

174. Vahter M, Berglund M, Akesson A, Liden C. Metals and women's health. Environ Res. 2002;88(3):145-55.

175. Clark NA, Teschke K, Rideout K, Copes R. Trace element levels in adults from the west coast of Canada and associations with age, gender, diet, activities, and levels of other trace elements. Chemosphere. 2007;70(1):155-64.

176. Jung E, Hyun W, Ro Y, Lee H, Song K. A study on blood lipid profiles, aluminum and mercury levels in college students. Nutr Res Pract. 2016;10(4):442-7.

177. Benes B, Spevackova V, Smid J, Cejchanova M, Cerna M, Subrt P, et al. The concentration levels of Cd, Pb, Hg, Cu, Zn and Se in blood of the population in the Czech Republic. Cent Eur J Public Health. 2000;8(2):117-9.

178. Rukgauer M, Klein J, Kruse-Jarres JD. Reference values for the trace elements copper, manganese, selenium, and zinc in the serum/plasma of children, adolescents, and adults. J Trace Elem Med Biol. 1997;11(2):92-8.

179. Bocca B, Madeddu R, Asara Y, Tolu P, Marchal JA, Forte G. Assessment of reference ranges for blood Cu, Mn, Se and Zn in a selected Italian population. J Trace Elem Med Biol. 2011;25(1):19-26.

180. Bonberg N, Pesch B, Ulrich N, Moebus S, Eisele L, Marr A, et al. The distribution of blood concentrations of lead (Pb), cadmium (Cd), chromium (Cr) and manganese (Mn) in residents of the German Ruhr area and its potential association with occupational exposure in metal industry and/or other risk factors. Int J Hyg Environ Health. 2017;220(6):998-1005.

181. Oulhote Y, Mergler D, Bouchard MF. Sex- and age-differences in blood manganese levels in the U.S. general population: national health and nutrition examination survey 2011-2012. Environ Health. 2014;13:87.

182. Kim Y, Lobdell DT, Wright CW, Gocheva VV, Hudgens E, Bowler RM. Blood metal concentrations of manganese, lead, and cadmium in relation to serum ferritin levels in Ohio residents. Biol Trace Elem Res. 2015;165(1):1-9.

183. Meltzer HM, Brantsaeter AL, Borch-Iohnsen B, Ellingsen DG, Alexander J, Thomassen Y, et al. Low iron stores are related to higher blood concentrations of manganese, cobalt and cadmium in non-smoking, Norwegian women in the HUNT 2 study. Environ Res. 2010;110(5):497-504.

184. Vahter M. Genetic polymorphism in the biotransformation of inorganic arsenic and its role in toxicity. Toxicol Lett. 2000;112-113:209-17.

185. Skroder Loveborn H, Kippler M, Lu Y, Ahmed S, Kuehnelt D, Raqib R, et al. Arsenic Metabolism in Children Differs From That in Adults. Toxicol Sci. 2016;152(1):29-39.

186. Nermell B, Lindberg AL, Rahman M, Berglund M, Persson LA, El Arifeen S, et al. Urinary arsenic concentration adjustment factors and malnutrition. Environ Res. 2008;106(2):212-8.

187. Hinwood AL, Sim MR, de Klerk N, Drummer O, Gerostamoulos J, Bastone EB. Are 24-hour urine samples and creatinine adjustment required for analysis of inorganic arsenic in urine in population studies? Environ Res. 2002;88(3):219-24.

188. Baeyens W, Vrijens J, Gao Y, Croes K, Schoeters G, Den Hond E, et al. Trace metals in blood and urine of newborn/mother pairs, adolescents and adults of the Flemish population (2007-2011). Int J Hyg Environ Health. 2014;217(8):878-90.

189. Bae HS, Ryu DY, Choi BS, Park JD. Urinary Arsenic Concentrations and their Associated Factors in Korean Adults. Toxicol Res. 2013;29(2):137-42.

190. Kalman DA, Dills RL, Steinmaus C, Yunus M, Khan AF, Prodhan MM, et al. Occurrence of trivalent monomethyl arsenic and other urinary arsenic species in a highly exposed juvenile population in Bangladesh. J Expo Sci Environ Epidemiol. 2014;24(2):113-20.

191. Lindberg AL, Ekstrom EC, Nermell B, Rahman M, Lonnerdal B, Persson LA, et al. Gender and age differences in the metabolism of inorganic arsenic in a highly exposed population in Bangladesh. Environ Res. 2008;106(1):110-20.

192. Silva MJ, Reidy JA, Preau JL, Samandar E, Needham LL, Calafat AM. Measurement of eight urinary metabolites of di(2-ethylhexyl) phthalate as biomarkers for human exposure assessment. Biomarkers. 2006;11(1):1-13.

193. Kasper-Sonnenberg M, Koch HM, Apel P, Ruther M, Palmke C, Bruning T, et al. Time trend of exposure to the phthalate plasticizer substitute DINCH in Germany from 1999 to 2017: Biomonitoring data on young adults from the Environmental Specimen Bank (ESB). Int J Hyg Environ Health. 2019;222(8):1084-92.

194. *Centers for Disease Control and Prevention (CDC). Fourth National Report on Human Exposure to Environmental Chemicals. Updated Tables (January 2019). Atlanta, U.S.: U.S. Department of Health and Human Services; 2019.*

195. *Bjermo H, Gyllenhammar I, Lindh CH, Lignell S. Concentrations of phthalate metabolites and phenolic substances in urine from first-time mothers in Uppsala, Sweden: temporal trends 2009-2018. Uppsala, Sweden: Swedish Food Agency; 2019.*

196. *Littorin M, Amilon Å, Maxe M, Axmon A, Jönsson BA, Lindh C. Exponering för kemiska bekämpningsmedel hos landsbygdsbefolkning i Skåne 2010-11. Lund, Sweden: Division of Occupational and Environmental Medicine, Lund University 2011.*

197. *Littorin M, Maxe M, Amilon Å, Jönsson BA, Lindh C. Analyser av pesticider i urin hos skånska kvinnor 2010. . Rapport 2013 till Naturvårdsverket: Arbets- och miljömedicin, Labmedicin Skåne, Lund och Avd för Arbets- och miljömedicin, Inst för Laboratoriemedicin, Lund; 2013.*

198. *Barr DB, Olsson AO, Wong LY, Udunka S, Baker SE, Whitehead RD, et al. Urinary concentrations of metabolites of pyrethroid insecticides in the general U.S. population: National Health and Nutrition Examination Survey 1999-2002. Environ Health Perspect. 2010;118(6):742-8.*

199. *Becker K, Seiwert M, Angerer J, Kolossa-Gehring M, Hoppe HW, Ball M, et al. GerES IV pilot study: assessment of the exposure of German children to organophosphorus and pyrethroid pesticides. Int J Hyg Environ Health. 2006;209(3):221-33.*

200. *Dewalque L, Pirard C, Charlier C. Measurement of urinary biomarkers of parabens, benzophenone-3, and phthalates in a Belgian population. Biomed Res Int. 2014;2014:649314.*

201. *Zota AR, Calafat AM, Woodruff TJ. Temporal trends in phthalate exposures: findings from the National Health and Nutrition Examination Survey, 2001-2010. Environ Health Perspect. 2014;122(3):235-41.*

202. *Frederiksen H, Jensen TK, Jorgensen N, Kyhl HB, Husby S, Skakkebaek NE, et al. Human urinary excretion of non-persistent environmental chemicals: an overview of Danish data collected between 2006 and 2012. Reproduction. 2014;147(4):555-65.*

203. *Preston EV, McClean MD, Claus Henn B, Stapleton HM, Braverman LE, Pearce EN, et al. Associations between urinary diphenyl phosphate and thyroid function. Environ Int. 2017;101:158-64.*

204. *Huang W, Caudill SP, Grainger J, Needham LL, Patterson DG, Jr. Levels of 1-hydroxypyrene and other monohydroxy polycyclic aromatic hydrocarbons in children: a study based on U.S. reference range values. Toxicol Lett. 2006;163(1):10-9.*

205. *Juric A, Singh K, Hu XF, Chan HM. Exposure to triclosan among the Canadian population: Results of the Canadian Health Measures Survey (2009-2013). Environ Int. 2019;123:29-38.*

206. *Vughs D, Baken KA, Dingemans MML, de Voogt P. The determination of two emerging perfluoroalkyl substances and related halogenated sulfonic acids and their significance for the drinking water supply chain. Environ Sci Process Impacts. 2019;21(11):1899-907.*

207. *Gebbink WA, Berger U, Cousins IT. Estimating human exposure to PFOS isomers and PFCA homologues: the relative importance of direct and indirect (precursor) exposure. Environ Int. 2015;74:160-9.*

Appendices

Appendix 1. Details of analytical methods used in *Riksmaten Adolescents 2016–17*

A1.1. Chlorinated and brominated persistent organic pollutants in individual serum samples

The analyses of chlorinated and brominated POPs (Table 1) in individual blood serum samples were performed by the Finnish National Institute for Health and Welfare, Department of Health Security. The method used has been described previously [123]. The method used for the analysis of 22 POPs in serum samples has been previously described in detail. The POP compounds measured were ten PCBs 74, 99, 118, 138, 153, 156, 170, 180, 183, 187; pesticides pentachlorobenzene (PeCB), hexachlorobenzene (HCB), alpha-hexachlorocyclohexane (α-HCH), beta-hexachlorocyclohexane (β-HCH), gamma-hexachlorocyclohexane (γ-HCH), oxychlordane, trans-nonachlor, p, p'-DDT, p, p′-DDE. Brominated diphenyl ethers: BDE-47, BDE-99, BDE-153 were also were measured.

Pre-treatment of the samples was as follows: ethanol and 13C-labelled internal standards of each compound in toluene were added to samples (200 µL) in test tubes and mixed to precipitate the proteins and equilibrate internal standards. Dichloromethane-hexane (1:4) was added for extraction followed by activated silica to bind the sample water, ethanol, and precipitate. Samples were mixed, and layers were allowed to separate. The upper dichloromethane-hexane layer was poured to a solid phase extraction cartridge (SPE cartridge) containing from bottom to top 10% AgNO3 impregnated silica and a mixture of Na2SO4 and silica. Elution of SPE cartridges was performed using dichloromethane-hexane, and the eluate was concentrated to 15–20 µL for gas chromatography-triple quadrupole mass spectrometry (GC-MS/MS) analysis. The instrument used was an Agilent 7010 GC-MS/MS system (Wilmington, DE, USA), GC column DB-5MS UI (J&W Scientific, 20m, ID 0.18 mm, 0.18 µm). Limits of quantification ranged from 5 pg/mL for PCB congeners and trans-nonachlor to 40 pg/mL for p,p'-DDE. All concentrations of POPs are in pg/mL serum. Two blank samples and two control samples (NIST SRM1958) were included in each batch of samples. Measured concentrations of POPs in SRM1958 were 80–105% of the certified/reference concentrations. The coefficient of variation (CV%) from SRM 1958 (n = 18) was <3.6% for all compounds. The THL laboratory participates three times a year in AMAP inter-laboratory comparisons (Ring Test for Persistent Organic Pollutants in Human Serum, National Institute of Public Health, Quebec, Canada). In AMAP ring tests for POPs, laboratory's results varied from 83% to 132% of the assigned values depending on the compound.

A1.2 Brominated flame retardants in pooled serum

Brominated flame retardants in pooled blood serum samples were analysed at the Swedish Food Agency according to a previously described method [18]. After extraction and clean-up, the analytes were quantified using capillary gas chromatography and mass selective detection in electron capture negative ionization and selected ion monitoring modes (GC/LRMS/ECNI-SIM). The system used for quantification consisted of an Agilent 6890N GC equipped with an Agilent 5973N MS. The sample (2 x3 µl), was injected (pulsed splitless) using a programmable temperature vaporizing (PTV) injector

with an initial temperature of 70°C followed by rapid heating to 300°C. The analytes were separated on a DB-5MS capillary column (15m x 0.25 mm id, 0.1 µm, J&W Scientific) using a ramped carrier gas flow and the oven temperature was programmed from 60°C to 325°C. Methane was used as reaction gas and the ion source, quadrupole and transfer line temperatures were kept at 210°C, 110°C and 310°C, respectively.

13C-BDE-155 was used as internal surrogate standard for the quantification of BDE-28, -47, -66, -100, -99, -154, -153 and -183 as well as HBB, PBEB, BTBPE and HBCD. 13C-BDE-209 was used for the quantification of BDE-209 (isotope dilution technique) and DBDPE.

Calibration standard solutions corresponding to a level range in serum of 0.625–125 ng/kg fresh weight for PBDEs, HBB, PBEB and BTBPE, 1.25–250 ng/kg for BDE-47, BDE-209 and HBCD and 12.5–625 ng/kg for DBDPE were included in the run. The various analytes were identified by their retention times relative to the internal standards. The samples were quantified using calibration curves created from the calibration standards analysed in the same run. Quadratic regression with the inverse square of concentration was used for the calibration curves.

All solvents used were tested for trace amounts of analytes. The glassware was either rinsed with acetone or heated in an oven at 450°C for at least 3 hours before use. Silica and alumina gel was heated at 450°C overnight to eliminate PBDE residuals and to lower the background levels of the blanks. Silica gel was deactivated with 3% MilliQ water and both silica and alumina gel were washed with n-hexane before use.

Due to possible UV induced degradation of the analytes, particularly for BDE-209, all sample extracts and standard solutions were stored in amber glassware and all steps were performed in a UV-free environment.

N-hexane was injected between the sample and calibration standard series to ensure there were no memory effects. A chemical blank was included in each extraction series to monitor background levels. A spiked in-house control sample was also included in each extraction series. For each batch of samples, the corresponding blank sample levels were subtracted from the sample levels. The LOD is derived from the lowest standard level injected giving a S/N of at least 6. The LOQ was determined as LOD plus five times the standard deviation of the blanks (LOD + 5 SDblank) and ranged between 0.6 and 20 ng/kg serum depending on the analyte. Measurement uncertainty was ≤50%.

A1.3 Per- and polyfluoroalkyl substances (PFAS) in blood serum

PFAS analyses in blood serum samples were performed at the Department of Environmental Science and Analytical Chemistry (ACES), Stockholm University. Sample extraction was carried out using a method adapted from Powley et al. [124]. Briefly, individual samples (~ 0.5 g serum in a polypropylene (PP) centrifuge tubes) were spiked with 0.5 ng each of a suite of isotopically labelled internal standards (Table A1.3.1). The samples were extracted twice with 4 mL of acetonitrile in an ultrasonic bath. Following centrifugation, the supernatant extract was removed, and the combined acetonitrile phases were concentrated to 1 mL under a stream of nitrogen. The concentrated extract underwent dispersive clean-up on graphitised carbon and acetic acid. A volume of 0.5 mL of the cleaned-up extract was added to 0.2 mL of 4 mM aqueous ammonium acetate, and volumetric standards M8PFOA and M8PFOS (50 µL of a 10 pg/µL solution) were added. The extract was stored at -20˚C prior to analysis. On the day of analysis, the extracts were centrifuged and transferred to an

autosampler vial for instrumental analysis. 5 µL aliquots of the final extracts were injected automatically on a Waters Acquity Ultra Performance Liquid Chromatograph (UPLC) which was coupled to a Waters Xevo TQS triple quadrupole mass spectrometer. Chromatographic separation of target analytes was achieved on a BEH C18 guard column (1.7 µm, 5 × 2.1 mm; Waters) coupled to a BEH C18 analytical column (1.7 µm particles, 50 × 2.1 mm). The mobile phase consisted of ammonium acetate in water (phase A) and ammonium acetate in acetonitrile (phase B), the proportions of which were adjusted over the course of the run. The mass spectrometer was operated in negative electrospray ionisation, multiple reaction monitoring (MRM) mode. Quantification was based on isotope dilution (see Table A1.3.1 for targets and their corresponding isotopically labelled internal standards).

Samples were run in batches consisting of 30 samples plus 2 blanks and 3-4 QC samples. The QC samples consisted of pooled human serum, which were analysed both with and without a spike of authentic PFAS. Additionally, the NIST Standard Reference Material 1057 was run with every third batch. LOQs were determined using the lowest calibration point. If a signal occurred in the method blanks within a batch, the LOQ was based on the mean blank + 3x standard deviation. LOQs differ between the batches as not all samples were analysed on the instrument at the same time. LOQs varied from 0.02 to 0.56 ng/g for different analytes and batches.

Blanks were mostly non-detects with only a few very low detects in some batches. Therefore, no blank correction was conducted on the final data. Data for the 45 individual PFAS measured in the present work were classified as either ‘quantitative’, ‘semi-quantitative’, ‘qualitative’, or ‘not-reported’, based on either availability of authentic standards and/or performance of QC samples (Table A1.3.1). In all cases, we assumed that data quality for branched (br-) and linear (lin-) isomers was equivalent. ‘Quantitative’ indicated that an exactly matched authentic standard was available and that QC samples displayed reasonable accuracy and precision (25 targets). Among these targets, average recoveries ranged from 96% to 133% (RSDs 12–35%), with the best performance observed for C6–C14 PFCAs (105–120%; RSD: 12–25%) and C4, C6, C8, and C10 PFSAs (109–119% RSDs: 20–28%). ‘Semi-quantitative’ targets were those quantified with exactly matched authentic standards but displaying sub-optimal accuracy and/or precision for QC samples. For these targets (i.e. 6:2 and 8:2 diPAP, EtFOSAA, 7:3 FTCA, and 8:2 FTSA), average recoveries ranged from 67–52% (RSD: 30–63%). ‘Qualitative’ targets included PFHxDA, PFOcDA, 6:2/8:2 diPAP, FOSAA, 3:3 and 5:3 FTCA, and 6:2 FTS, for which very poor accuracy and precision were observed, as well as PFPeS, PFHpS, PFNS, PFUnDS, and PFPeDA, for which authentic standards were unavailable and therefore method performance could not be assessed. We note that most targets displaying poor accuracy and/or precision did not have exactly matched isotopically labelled internal standards. It is expected that as both authentic and isotopically labelled standards become available, data quality for all of these ‘qualitative’ targets will improve considerably. Finally, one target (HFPO-DA) was not reported due to very poor recoveries and reproducibility in QC samples across all batches. The reason for this is unclear but may be related to in-course fragmentation, a phenomenon that has been previously reported for this particular target [206].

Finally, concentrations determined in NIST values reference material were in good agreement with both the certified values and those previously reported by others [207].

Table A1.3.1. Target analytes with their quantification and qualification ions as well as the internal standard used for quantification. lin indicates linear isomers, and br indicates branched isomer.

Target Analyte	Precursor ion	Quant. product ion	Qual. product ion	Standard	IS	IS transition	Data Quality*
PFHxA	313	269	119	lin-PFHxA	$^{13}C_2$-PFHxA	315/270	Quant
PFHpA	363	319	169	lin-PFHpA	$^{13}C_4$-PFOA	367/322	Quant
lin-PFOA	413	369	169	lin-PFOA	$^{13}C_4$-PFOA	417/372	Quant
br-PFOA	413	369	169	lin-PFOA	$^{13}C_4$-PFOA	417/372	Quant
PFNA	463	419	219	lin-PFNA	$^{13}C_5$-PFNA	468/423	Quant
PFDA	513	469	269	lin-PFDA	$^{13}C_2$-PFDA	515/470	Quant
PFUnDA	563	519	269	lin-PFUnDA	$^{13}C_2$-PFUnDA	565/520	Quant
PFDoDA	613	569	169	lin-PFDoDA	$^{13}C_2$-PFDoDA	615/570	Quant
PFTrDA	662.9	619	169	lin-PFTrDA	$^{13}C_2$-PFDoDA	615/570	Quant
PFTeDA	712.9	669	169	lin-PFTeDA	$^{13}C_2$-PFDoDA	615/570	Quant
PFPeDA	762.9	719	169	lin-PFTeDA	$^{13}C_2$-PFDoDA	615/570	Qual
PFHxDA	813	769	169	lin-PFHxDA	$^{13}C_2$-PFDoDA	615/570	Qual
PFOcDA	913	869	169	lin-PFOcDA	$^{13}C_2$-PFDoDA	615/570	Qual
PFBS	298.9	80	99	lin-PFBS	$^{18}O_2$-PFHxS	403/84	Quant
PFPeS	348.9	80	99	lin-PFHxS	$^{18}O_2$-PFHxS	403/84	Qual
lin-PFHxS	398.9	80	99	lin-PFHxS	$^{18}O_2$-PFHxS	403/84	Quant
br-PFHxS	399	80	99	lin-PFHxS	$^{18}O_2$-PFHxS	403/84	Quant
PFHpS	448.9	80	99	lin-PFHxS	$^{18}O_2$-PFHxS	403/84	Qual
lin-PFOS	498.9	80, 99		lin-PFOS	$^{13}C_4$-PFOS	503/80	Quant
br-PFOS	498.9	80, 99		lin-PFOS	$^{13}C_4$-PFOS	503/80	Quant
PFNS	548.9	80	99	lin-PFOS	$^{13}C_4$-PFOS	503/80	Qual
lin-PFDS	598.9	80	99	lin-PFDS	$^{13}C_4$-PFOS	503/80	Quant
br-PFDS	598.9	80	99	lin-PFDS	$^{13}C_4$-PFOS	503/80	Quant
PFUnDS	648.9	80	99	lin-PFDS	$^{13}C_4$-PFOS	503/80	Qual
lin-FOSA	497.9	78	169	lin-FOSA	$^{13}C_8$-FOSA	506/78	Quant
br-FOSA	497.9	78	169	lin-FOSA	$^{13}C_8$-FOSA	506/78	Quant
lin-FOSAA	555.9	498	419	lin-FOSAA	D_3-MeFOSAA	573/419	Qual
br-FOSAA	555.9	498	419	lin-FOSAA	D_3-MeFOSAA	573/419	Qual
lin-MeFOSAA	570	419	483	lin-MeFOSAA	D_3-MeFOSAA	573/419	Quant
br-MeFOSAA	570	419	483	lin-MeFOSAA	D_3-MeFOSAA	573/419	Quant
lin-EtFOSAA	584	419	526	lin-EtFOSAA	D_5-EtFOSAA	589/419	Semi-Q
br-EtFOSAA	584	419	526	lin-EtFOSAA	D_5-EtFOSAA	589/419	Semi-Q
9Cl-PF3ONS	531	351	83	lin-9Cl-PF3ONS	$^{13}C_4$-PFOS	503/80	Quant
11Cl-PF3OUdS	631	451	83	lin-11Cl-PF3OUdS	$^{13}C_4$-PFOS	503/80	Quant
ADONA	377	251	85	ADONA	$^{13}C_4$-PFOS	503/80	Quant
HFPO-DA	329	169	185	HFPO-DA	$^{13}C_4$-PFOA	417/372	NR
3:3 FTCA	241	117	177	lin-3:3 FTCA	$^{13}C_4$-PFOA	417/372	Qual
5:3 FTCA	341	237	217	lin-5:3 FTCA	$^{13}C_4$-PFOA	417/372	Qual

Target Analyte	Precursor ion	Quant. product ion	Qual. product ion	Standard	IS	IS transition	Data Quality*
7:3 FTCA	441	337	148	lin-7:3 FTCA	$^{13}C_4$-PFOA	417/372	Semi-Q
4:2 FTSA	327	307	80.6	lin-4:2 FTSA	$^{13}C_2$-6:2 FTSA	429/409	Quant
6:2 FTSA	427	407	80.6	lin-6:2 FTSA	$^{13}C_2$-6:2 FTSA	429/409	Qual
8:2 FTSA	527	507	80.6	lin-8:2 FTSA	$^{13}C_2$-6:2 FTSA	429/409	Semi-Q
6:2 diPAP	789	443	97	lin-6:2 diPAP	$^{13}C_4$-6:2 diPAP	793/445	Semi-Q
8:2 diPAP	989	543	97	lin-8:2 diPAP	$^{13}C_4$-8:2 diPAP	993/545	Semi-Q
6:2/8:2 diPAP	889	443	543	lin-6:2/8:2 diPAP	$^{13}C_4$-8:2 diPAP	993/545	Qual
$^{13}C_8$-PFOA**	421	376					
$^{13}C_8$-PFOS**	507	80					

*Quant = quantitative; indicates that an authentic standard is available and QC samples displayed reasonable accuracy and precision across all batches. Note that the data quality for branched isomers is assumed to be the same as their linear counterpart.
Semi-Q =semi-quantitative; indicates authentic standard was used but sub-optimal accuracy and/or precision was obtained across all batches. Note that the data quality for branched isomers is assumed to be the same as their linear counterpart.
Qual = qualitative; indicates that an authentic standard was not available in which case quantification was performed with a structurally similar substance, OR that poor accuracy and/or precision was observed for QC samples. Note that the data quality for branched isomers is assumed to be the same as their linear counterpart.
NR = Not Reported. Spikes were not recovered in most QC samples.
** $^{13}C_8$-PFOA and $^{13}C_8$-PFOS were used as recovery internal standards.

A1.4. Metals in whole blood and serum, and metalloid arsenic in urine

Analysis of Pb, Cd, Hg, Cr, Mn, Co, Ni and Al were performed at the Division of Occupational and Environmental Medicine, Lund University. For the determination of Pb, Cd, Hg, Cr, Mn, Co and Ni in whole blood, venous blood was collected in a 4 ml lithium heparin tube. For the determination of Al in serum, venous blood was collected into a 10 ml tube with coagulation activator. The blood was then allowed to clot for 30 min, and the clot was removed by centrifugation (1500 g). Blood and serum samples were stored at −80°C until analysis. Prior to analysis, the samples were diluted 20-fold with an alkaline solution as described by Bárány et al. [125]. All analyses were performed in duplicate. The concentrations of metals were determined by inductively coupled plasma mass spectrometry (ICP-MS; iCAP Q, Thermo Fisher Scientific, Bremen, GmbH) equipped with collision cell with kinetic energy discrimination and helium as collision gas. All analyses were performed in duplicate. The LOD values were estimated as signal-to-noise ratio (S/N) of three and varied from 0.05 to 5.0 µg/L. Precision of the method varied from 2.8% to 15% depending on analyte. Valid certified reference materials (serum samples with one for Al and two for other metals known concentrations of analytes) were routinely analysed to control analytical quality of the results. The reference materials were obtained from Seronorm Trace Elements Serum L-1, Lot 0903106 (SERO AS, Billingstad, Norway), Seronorm Trace Elements Whole Blood L-1, Lot 1103128 (SERO AS, Billingstad, Norway) and G-EQUAS R59, Material 1A (The German External Quality Assessment Scheme, Erlangen Germany). There was a good agreement between provided and measured concentrations in the reference materials.

Levels of total arsenic and arsenic species were determined in individual and pooled urine samples at the Swedish Food Agency. The As species determined were: inorganic As (sum of arsenite and arsenate); the organic As species dimethyl arsenate (DMA); monomethyl arsonate (MMA); and arsenobetain (AB). Levels of inorganic As and MMA were determined according to the European

standard for inorganic As based on anion-exchange HPLC-ICP-MS [126], and levels of DMA and AB according to a method based on cation–exchange HPLC-ICP-MS [127]. Total arsenic was determined by ICP-MS according to the European Standard for total As [128]. The methods for total As, inorganic As and MMA are accredited by SWEDAC (Sweden's national accreditation body). An Agilent 7700x ICP-MS was used in all the three methods. The LOD values were 0.2 µg/kg for inorganic As and MMA, 0.4 µg/kg for DMA and AB, and 2.2 µg/kg for total As. The expanded uncertainty was ±34% for inorganic As and MMA, and ±20% for total As, DMA and AB. Certified reference materials were analysed with the samples to monitor both short- and long-term analytical quality of the results. Relative standard deviations (RSD) for analysis of the urine certified reference materials (NIST 2669 Arsenic species in frozen human urine Level 1 and 2, and Seronorm Trace Elements Urine L-1) varied from 2.4% to 8.1% depending on the As species analysed (Table A1.4.1). The relative bias (the differences between the concentration of the reference material and the concentration found using the analytical method) varied from -12% to +19% (Table A1.4.1).

Table A1.4.1. Results and certified concentrations for the urine certified reference materials analysed with the respective method.

Anion-HPLC-ICP-MS	**Inorganic As (sum of AsIII and AsV)**				
	Result, mg/kg	RSD, %	No replicates	Relative bias, %	Certified conc. +/- MO*, mg/kg
Urine, NIST 2669 Level I	0.0046	2.4	9	19	0.00388 +/- (0.0004)
Urine, NIST 2669 Level II	0.0128	6	9	15	0.01119 +/- (0.0013)
	Monomethyl arsenic acid, MMA				
Urine, NIST 2669 Level I	0.00191	2.7	9	2	0.00188 +/- 0.00039
Urine, NIST 2669 Level II	0.00758	6.5	9	4	0.00718 +/- 0.00056
Cation-HPLC-ICP-MS	**Dimethyl arsenic acid, DMA**				
Urine, NIST 2669 Level I	0.00323	6.2	13	-7	0.00347 +/- 0.00041
Urine, NIST 2669 Level II	0.0221	8	12	-12	0.0253 +/- 0.0007
	Arsenobetain, AB				
Urine, NIST 2669 Level I	0.0117	5.8	13	-6	0.0124 +/- 0.0019
Urine, NIST 2669 Level II	0.00135	8.1	12	-6	0.00143 +/- 0.00008
ICP-MS	**Total arsenic**				
Urine, Seromnorm Level I	0.150	6.2	8	-5	0.158 +/- 0.32

*MO = measurement uncertainties (MO). Figures in brackets for iAs are pooled from uncertainties for AsIII and AsV in the certificate.

A1.5. Phthalate metabolites and phenolic compounds in urine

Phthalate metabolites and phenolic compounds were analysed in urine by the Division of Occupational and Environmental Medicine, Lund University as previously described [129] with some modifications. Briefly, urine was added to ammonium acetate (pH 6.5) and glucoronidase (E-coli), and incubated at 37°C for 30 min. Thereafter, labelled (3H or 13C) internal standards (IS) of all analysed compounds in a 50:50 (v:v) water and acetonitrile solution were added, with the exception of BHA and DBP. A C18 column was used prior to the injector to reduce interference of contaminants in the mobile phase. The compounds were separated on different C18 columns. The mobile phases were buffered water and acetonitrile or methanol with 0.08% formic acid, 0.1% ammonia or 5 mM ammonium format. The samples were analysed on a Shimadzu UFLC system (Shimadzu Corporation, Kyoto, Japan) coupled to a QTRAP5500 triple quadrupole linear ion trap mass spectrometer equipped with a TurboIon Spray source (LC-MS/MS; AB Sciex, Foster City, CA, USA). All samples were analysed randomly. For quality control of the analyses, chemical blanks and in-house prepared quality control samples were analysed in all sample batches. LOD was defined as the concentration corresponding to a peak area ratio of three times the standard deviation of the chemical blanks. LODs varied from 0.03 to 1.6 ng/mL for different analytes. The imprecision of the method, determined as the coefficient of variation (CV) of the quality control sample, did not exceed 20% for all compounds with the exception of cx-MiDP (27% at concentration of 0.6 ng/mL), BPS (25% at concentration of 0.8 ng/mL), DBP (41% at concentration of 0.1 ng/mL) and BHA (21% at concentration of 0.8 ng/mL).

Lund University is a reference laboratory for analyses of urinary phthalate metabolites and BPA in European biomonitoring projects (http://www.eu-hbm.info/cophes and https://www.hbm4eu.eu/). The laboratory participates in the ICI/EQUAS exercises for the analysis of BPA, BPS, 4,4-BPF, 1-HP, MBzP, MEHP , 5OH MEHP, 5oxo MEHP, 5cx MEPP, OH MiNP, cx- MiNP , OH MiDP, cx-MiDP, OH-MINCH and cx-MINCH, and is approved for these compounds in the HBM4EU project. Moreover, the laboratory participates in the Erlangen Inter-Laboratory Comparison Program for several phthalate metabolites, TCP, and 3-PBA.

Appendix 2. Summary statistics of contaminants per region, gender and grade

Table A2.1. Serum concentrations of chlorinated and brominated POPs (pg/mL) in Swedish adolescents per region (n=1,096). Compounds that were detected in less than 50% of the samples are excluded from the table.

Region	Compound	Mean	SD	5th percentile	Median	95th percentile
Gothenburg	HCB	44	14	24	42	72
(n=221)	p,p'-DDE	139	238	<LOQ	84	393
	PCB-118	7	4	<LOQ	6	16
	PCB-138	33	20	10	29	66
	PCB-153	54	35	14	46	113
	PCB-170	16	12	<LOQ	13	42
	PCB-180	33	25	7	26	82
	PCB-187	7	5	<LOQ	6	17
	Total PCB	170	106	57	144	344
Linköping	HCB	90	591	22	41	77
(n=156)	p,p'-DDE	222	471	43	99	723
	PCB-118	7	6	<LOQ	6	17
	PCB-138	30	27	8	23	72
	PCB-153	48	39	12	35	124
	PCB-170	14	14	<LOQ	10	37
	PCB-180	28	27	5	20	75
	PCB-187	7	7	<LOQ	<LOQ	20
	Total PCB	155	126	51	115	378
Lund	HCB	44	16	23	43	72
(n=210)	p,p'-DDE	137	171	<LOQ	98	314
	PCB-118	7	4	<LOQ	6	15
	PCB-138	31	21	8	26	75
	PCB-153	52	37	12	43	135
	PCB-170	16	13	<LOQ	11	44
	PCB-180	32	27	<LOQ	23	88
	PCB-187	7	5	<LOQ	5	18
	Total PCB	164	111	50	131	394
Stockholm	HCB	54	18	32	50	88
(n=164)	p,p'-DDE	200	227	56	132	621
	PCB-118	9	6	<LOQ	8	18
	PCB-138	41	27	12	34	100
	PCB-153	70	49	17	54	172
	PCB-170	20	18	<LOQ	14	48
	PCB-180	44	44	7	29	111
	PCB-187	10	13	<LOQ	7	26

Region	Compound	Mean	SD	5th percentile	Median	95th percentile
	Total PCB	220	160	67	177	501
Umeå	HCB	42	35	20	37	67
(n=131)	p,p'-DDE	249	740	<LOQ	72	1708
	PCB-118	9	8	<LOQ	7	22
	PCB-138	31	22	9	24	66
	PCB-153	47	32	13	40	106
	PCB-170	13	11	<LOQ	11	36
	PCB-180	26	21	6	21	68
	PCB-187	6	4	<LOQ	<LOQ	15
	Total PCB	151	99	54	123	348
Uppsala	HCB	46	17	19	43	76
(n=84)	p,p'-DDE	189	349	41	108	505
	PCB-118	7	4	<LOQ	7	14
	PCB-138	31	19	7	28	66
	PCB-153	52	34	11	47	108
	PCB-170	17	15	<LOQ	14	43
	PCB-180	33	29	5	27	94
	PCB-187	7	5	<LOQ	6	15
	Total PCB	167	108	48	147	357
Örebro	HCB	43	20	19	40	70
(n=130)	p,p'-DDE	260	594	<LOQ	88	1043
	PCB-118	8	10	<LOQ	6	20
	PCB-138	28	23	7	22	77
	PCB-153	45	39	10	33	129
	PCB-170	13	11	<LOQ	9	45
	PCB-180	25	23	<LOQ	17	87
	PCB-187	7	5	<LOQ	<LOQ	17
	Total PCB	153	164	46	109	385

Table A2.2. Serum concentrations of chlorinated and brominated POPs (pg/mL) in Swedish adolescents per grade (grade 5, n=332; grade 8, n=409; grade 11, n=355; total n=1,096). Compounds that were detected in less than 50% of the samples are excluded from the table.

Compound	Grade	Mean	SD	5th percentile	Median	95th percentile
HCB	5	47	26	21	43	75
	8	64	365	22	44	77
	11	43	16	22	40	72
p,p'-DDE	5	186	396	<LOQ	89	549
	8	156	208	<LOQ	98	442
	11	236	582	<LOQ	94	680
PCB-118	5	8	6	<LOQ	6	17
	8	8	5	<LOQ	6	16
	11	8	8	<LOQ	7	18
PCB-138	5	31	23	8	25	79
	8	34	24	9	28	72
	11	32	22	9	27	84
PCB-153	5	51	41	11	40	133
	8	56	40	14	48	129
	11	52	36	12	43	132
PCB-170	5	15	14	<LOQ	10	41
	8	17	15	<LOQ	13	42
	11	15	12	<LOQ	12	45
PCB-180	5	29	28	<LOQ	20	80
	8	35	33	6	26	89
	11	31	25	6	24	90
PCB-187	5	7	6	<LOQ	4	18
	8	8	9	<LOQ	6	19
	11	7	5	<LOQ	6	19
Total PCB	5	161	122	50	123	391
	8	177	128	54	145	388
	11	170	132	53	138	420

Table A2.3. Serum concentrations of chlorinated and brominated POPs (pg/mL) in Swedish adolescents per gender (boys, n=482; girls, n=614; total n=1,096). Compounds that were detected in less than 50% of the samples are excluded from the table.

Compound	Gender	Mean	SD	5th percentile	Median	95th percentile
HCB	Boys	51	24	26	47	82
	Girls	53	298	21	39	65
p,p'-DDE	Boys	209	409	41	106	639
	Girls	177	423	<LOQ	85	537
PCB-118	Boys	8	6	<LOQ	7	19
	Girls	8	6	<LOQ	6	16
PCB-138	Boys	37	25	9	31	82
	Girls	29	21	8	24	65
PCB-153	Boys	61	42	15	51	143
	Girls	47	36	12	38	108
PCB-170	Boys	19	15	<LOQ	14	47
	Girls	14	12	<LOQ	10	34
PCB-180	Boys	38	31	7	28	97
	Girls	27	27	<LOQ	20	71
PCB-187	Boys	9	7	<LOQ	7	21
	Girls	7	8	<LOQ	<LOQ	16
Total PCB	Boys	193	130	57	160	422
	Girls	152	123	50	116	344

Table A2.4. Serum concentrations of PFASs (ng/g) in Swedish adolescents per region (n=1,096-1,098). Compounds detected in more than 50% of the samples are included in the table.

Region	Compound	Mean	SD	5th percentile	Median	95th percentile
Gothenburg	lin-PFOA	1.40	0.63	0.73	1.25	2.49
(n=220-221)	PFNA	0.42	0.21	0.16	0.37	0.79
	PFDA	<LOQ	0.11	<LOQ	0.18	0.42
	lin-PFHxS	0.43	0.38	<LOQ	0.36	0.89
	total PFHxS	0.51	0.39	0.18	0.44	0.95
	lin-PFOS	2.28	1.42	0.91	1.93	5.06
	br-PFOS	0.94	0.46	<LOQ	0.87	1.85
	total PFOS	3.23	1.80	1.32	2.75	6.75
Linköping	lin-PFOA	1.28	0.63	0.51	1.12	2.49
(n=156)	PFNA	0.42	0.23	<LOQ	0.37	0.87
	PFDA	0.19	0.15	<LOQ	0.14	0.45
	lin-PFHxS	0.63	1.36	<LOQ	0.37	1.03
	total PFHxS	0.69	1.39	0.22	0.45	1.08
	lin-PFOS	2.19	1.79	0.73	1.67	5.02
	br-PFOS	0.96	0.89	<LOQ	0.81	1.98
	total PFOS	3.15	2.60	1.10	2.54	6.96
Lund	lin-PFOA	1.40	1.09	0.58	1.14	3.10
(n=209)	PFNA	0.38	0.19	<LOQ	0.35	0.78
	PFDA	0.19	0.13	<LOQ	0.16	0.39
	lin-PFHxS	5.58	23.86	<LOQ	0.37	29.87
	total PFHxS	5.77	24.30	0.22	0.43	30.79
	lin-PFOS	5.57	12.52	0.82	2.11	23.58
	br-PFOS	3.51	10.43	<LOQ	0.93	18.33
	total PFOS	9.07	22.89	1.27	3.04	42.29
Stockholm	lin-PFOA	1.44	0.59	0.64	1.34	2.44
(n=166-167)	PFNA	0.51	0.33	0.17	0.46	1.02
	PFDA	0.21	0.15	<LOQ	0.18	0.44
	lin-PFHxS	0.95	1.52	<LOQ	0.72	1.82
	total PFHxS	1.03	1.54	0.32	0.78	1.88
	lin-PFOS	3.70	2.63	1.19	2.89	8.99
	br-PFOS	1.62	1.13	<LOQ	1.30	3.81
	total PFOS	5.32	3.63	1.74	4.40	12.80
Umeå	lin-PFOA	1.24	0.68	0.47	1.09	2.72
(n=131)	PFNA	0.39	0.23	<LOQ	0.35	0.82
	PFDA	0.21	0.17	<LOQ	0.14	0.56
	lin-PFHxS	0.37	0.42	<LOQ	0.29	0.81
	total PFHxS	0.42	0.42	0.11	0.35	0.85

Region	Compound	Mean	SD	5th percentile	Median	95th percentile
	lin-PFOS	1.88	0.92	0.71	1.71	3.51
	br-PFOS	0.88	0.47	<LOQ	0.77	1.92
	total PFOS	2.76	1.32	1.21	2.57	5.44
Uppsala	lin-PFOA	1.42	0.95	0.57	1.19	3.16
(n=84)	PFNA	0.48	0.37	<LOQ	0.40	1.21
	PFDA	0.23	0.22	<LOQ	0.16	0.70
	lin-PFHxS	2.46	4.54	<LOQ	0.36	11.92
	total PFHxS	2.61	4.72	0.18	0.44	12.46
	lin-PFOS	2.17	1.44	0.63	1.76	4.74
	br-PFOS	1.07	0.73	<LOQ	0.83	2.20
	total PFOS	3.23	2.10	1.00	2.66	6.58
Örebro	lin-PFOA	1.23	0.42	0.55	1.21	2.15
(n=130)	PFNA	0.42	0.21	<LOQ	0.40	0.81
	PFDA	0.19	0.12	<LOQ	0.16	0.40
	lin-PFHxS	0.90	1.16	<LOQ	0.48	2.50
	total PFHxS	0.96	1.19	0.24	0.54	2.59
	lin-PFOS	2.29	1.36	0.79	1.92	4.93
	br-PFOS	1.06	0.54	<LOQ	0.96	2.04
	total PFOS	3.35	1.80	1.18	2.77	6.80

Table A2.5. Serum concentrations of PFASs (ng/g) in Swedish adolescents per grade (grade 5, n=332; grade 8, n=410; grade 11, n=354–356; total n=1,096–1,098). Compounds that were detected in less than 50% of the samples are excluded from the table.

Compound	Grade	Mean	SD	5th percentile	Median	95th percentile
lin-PFOA	5	1.51	0.93	0.61	1.31	3.02
	8	1.21	0.54	0.62	1.14	2.19
	11	1.36	0.75	0.55	1.20	2.67
PFNA	5	0.43	0.25	<LOQ	0.40	0.81
	8	0.40	0.22	<LOQ	0.36	0.79
	11	0.46	0.28	<LOQ	0.39	1.05
PFDA	5	0.19	0.13	<LOQ	0.17	0.41
	8	0.18	0.11	<LOQ	0.15	0.38
	11	0.22	0.19	<LOQ	0.17	0.60
lin-PFHxS	5	4.51	19.11	<LOQ	0.50	15.85
	8	0.53	0.94	<LOQ	0.37	1.03
	11	0.50	0.66	<LOQ	0.35	1.10
total PFHxS	5	4.67	19.47	0.21	0.58	16.32
	8	0.60	0.96	0.20	0.45	1.14
	11	0.57	0.66	0.18	0.42	1.15
lin-PFOS	5	4.54	10.10	0.82	2.23	14.34
	8	2.36	1.78	0.79	1.83	5.51
	11	2.46	1.73	0.76	1.95	6.03
br-PFOS	5	2.73	8.36	<LOQ	1.05	9.81
	8	1.02	0.76	<LOQ	0.83	2.28
	11	1.07	0.69	<LOQ	0.91	2.41
total PFOS	5	7.27	18.39	1.25	3.33	24.00
	8	3.38	2.46	1.25	2.63	7.51
	11	3.53	2.34	1.10	2.85	8.10

Table A2.6. Serum concentrations of PFASs (ng/g) in Swedish adolescents per gender (boys, n=482; girls, n=614–616; total n=1,096–1,098). Compounds that were detected in less than 50% of the samples are excluded from the table.

Compound	Grade	Mean	SD	5th percentile	Median	95th percentile
lin-PFOA	Boys	1.39	0.75	0.65	1.28	2.30
	Girls	1.32	0.75	0.57	1.14	2.67
PFNA	Boys	0.46	0.26	<LOQ	0.41	0.87
	Girls	0.40	0.24	<LOQ	0.36	0.85
PFDA	Boys	0.20	0.16	<LOQ	0.15	0.45
	Girls	0.20	0.14	<LOQ	0.18	0.41
lin-PFHxS	Boys	2.23	14.25	<LOQ	0.46	4.96
	Girls	1.32	6.66	<LOQ	0.36	1.48
total PFHxS	Boys	2.33	14.49	0.21	0.53	5.09
	Girls	1.42	6.84	0.18	0.42	1.54
lin-PFOS	Boys	3.52	7.29	0.93	2.22	7.76
	Girls	2.69	4.32	0.72	1.82	6.23
br-PFOS	Boys	1.84	6.10	<LOQ	1.06	3.09
	Girls	1.33	3.18	<LOQ	0.82	2.51
total PFOS	Boys	5.35	13.33	1.37	3.34	10.82
	Girls	4.02	7.42	1.10	2.65	8.45

Table A2.7. Concentrations of metals (µg/L) in whole blood in Swedish adolescents per region (n=1,099). Metals that were detected in less than 50% of the samples are excluded from the table.

Region	Metal	Mean	SD	5th percentile	Median	95th percentile
Gothenburg	Cd	0.14	0.09	0.06	0.12	0.25
(n=222)	Hg	0.89	0.86	0.08	0.72	1.97
	Pb	8.29	6.37	3.79	6.95	15.71
	Cr	0.61	0.20	0.40	0.60	0.79
	Mn	10.93	3.10	7.02	10.67	16.33
	Co	0.15	0.10	0.07	0.12	0.35
	Ni	0.69	0.27	0.47	0.64	1.01
Linköping	Cd	0.18	0.22	0.06	0.13	0.42
(n=157)	Hg	0.86	0.59	0.19	0.77	2.06
	Pb	8.21	5.49	3.39	6.80	18.26
	Cr	0.65	0.15	0.47	0.61	0.93
	Mn	10.94	3.28	6.96	10.34	16.89
	Co	0.15	0.10	0.06	0.13	0.38
	Ni	0.59	0.16	0.36	0.57	0.86
Lund	Cd	0.15	0.17	0.07	0.12	0.33
(n=210)	Hg	0.95	0.85	0.13	0.72	2.72
	Pb	8.06	3.93	3.82	7.22	14.14
	Cr	0.54	0.13	0.36	0.53	0.77
	Mn	10.99	3.05	7.11	10.47	16.60
	Co	0.15	0.09	0.06	0.11	0.31
	Ni	0.64	0.16	0.40	0.63	0.83
Stockholm	Cd	0.16	0.21	0.06	0.12	0.35
(n=166)	Hg	1.04	1.28	0.21	0.79	2.08
	Pb	8.51	5.64	4.09	7.08	16.69
	Cr	0.45	0.10	0.33	0.43	0.63
	Mn	11.48	3.35	7.23	10.87	16.96
	Co	0.16	0.10	0.06	0.13	0.36
	Ni	0.70	0.14	0.47	0.70	0.91
Umeå	Cd	0.21	0.45	<LOD	0.11	0.45
(n=130)	Hg	0.73	0.63	0.14	0.59	1.94
	Pb	8.09	4.15	3.61	7.49	15.88
	Cr	0.55	0.23	0.25	0.59	0.92
	Mn	11.0	4.10	6.74	9.84	18.27
	Co	0.15	0.11	0.05	0.10	0.39
	Ni	0.67	0.26	0.48	0.60	0.39

Region	Metal	Mean	SD	5th percentile	Median	95th percentile
Uppsala	Cd	0.16	0.23	<LOD	0.11	0.33
(n=84)	Hg	0.71	0.46	0.17	0.64	1.49
	Pb	10.16	14.80	3.94	7.83	18.14
	Cr	0.67	0.21	0.49	0.64	0.94
	Mn	10.99	3.68	6.73	10.44	17.55
	Co	0.13	0.07	0.05	0.10	0.29
	Ni	0.57	0.10	0.43	0.58	0.72
Örebro	Cd	0.13	0.08	0.05	0.12	0.26
(n=130)	Hg	1.02	0.96	0.22	0.77	2.30
	Pb	7.98	3.83	3.24	7.38	15.77
	Cr	0.77	0.24	0.53	0.67	1.20
	Mn	11.07	3.58	5.92	10.86	17.19
	Co	0.15	0.11	0.06	0.12	0.40
	Ni	0.60	0.20	0.35	0.59	0.90

Table A2.8. Concentrations of metals (µg/L) in whole blood in Swedish adolescents per grade (grade 5, n=331; grade 8, n=410; grade 11, n=358; total n=1,099). Metals that were detected in less than 50% of the samples are excluded from the table.

Metal	Grade	Mean	SD	5th percentile	Median	95th percentile
Cd	5	0.10	0.05	<LOD	0.10	0.20
	8	0.15	0.11	0.06	0.13	0.30
	11	0.23	0.36	0.06	0.14	0.60
Hg	5	0.87	0.59	0.17	0.75	1.93
	8	0.94	1.03	0.14	0.69	2.26
	11	0.89	0.90	0.14	0.72	2.06
Pb	5	8.35	4.45	3.86	7.33	16.43
	8	8.43	7.77	3.56	7.06	16.30
	11	8.27	6.15	3.61	7.11	16.19
Cr	5	0.57	0.19	0.36	0.55	0.77
	8	0.58	0.15	0.36	0.57	0.86
	11	0.64	0.25	0.27	0.63	1.09
Mn	5	10.83	2.95	6.74	10.48	16.18
	8	11.75	3.53	7.33	11.20	17.62
	11	10.48	3.46	6.34	9.80	17.01
Co	5	0.13	0.07	0.06	0.11	0.26
	8	0.18	0.12	0.07	0.15	0.43
	11	0.13	0.09	0.05	0.10	0.33
Ni	5	0.64	0.25	0.42	0.61	0.88
	8	0.66	0.17	0.43	0.64	0.90
	11	0.64	0.19	0.37	0.60	0.98

Table A2.9. Concentrations of metals (µg/L) in whole blood in Swedish adolescents per gender (boys, n=480; girls, n=619; total n=1,099). Metals that were detected in less than 50% of the samples are excluded from the table.

Metal	Gender	Mean	SD	5th percentile	Median	95th percentile
Cd	Boys	0.14	0.16	0.06	0.11	0.27
	Girls	0.18	0.26	0.06	0.13	0.39
Hg	Boys	1.05	1.12	0.19	0.78	2.59
	Girls	0.79	0.59	0.13	0.67	1.87
Pb	Boys	9.45	8.51	4.39	7.82	17.51
	Girls	7.47	3.80	3.44	6.61	16.04
Cr	Boys	0.61	0.23	0.36	0.59	1.0
	Girls	0.58	0.18	0.31	0.58	0.88
Mn	Boys	10.60	3.21	6.49	10.08	16.37
	Girls	11.41	3.47	6.95	10.88	17.51
Co	Boys	0.12	0.08	0.05	0.10	0.30
	Girls	0.17	0.11	0.06	0.14	0.40
Ni	Boys	0.64	0.22	0.42	0.61	0.90
	Girls	0.65	0.19	0.41	0.63	0.94

Table A2.10. Density-adjusted urine concentrations of phthalate metabolites and phenolic compounds (ng/mL) in Swedish adolescents per region (n=1,095–1,104). Compounds that were detected in less than 50% of the samples are excluded from the table.

Region	Compound	Mean	SD	5th percentile	Median	95th percentile
Gothenburg	MEP	61.3	90.1	9.7	30.3	213.4
(n=220-222)	MnBP	55.3	49.9	17.4	40.9	161.3
	MBzP	10.1	11.9	1.2	5.6	37.6
	MEHP	1.9	1.4	0.7	1.6	3.9
	5-OH-MEHP	8.7	7.8	3.0	7.0	19.8
	5-oxo-MEHP	6.9	5.2	2.0	5.5	16.6
	2-cx-MEHP	2.4	1.9	0.7	1.8	5.9
	5-cx-MEPP	8.0	7.2	2.6	6.4	20.1
	OH-MiNP	11.1	45.8	1.2	3.2	34.9
	Oxo-MiNP	4.6	16.8	0.6	1.6	13.2
	cx-MiNP	15.8	66.2	2.0	5.4	46.9
	cx-MiDP	0.5	0.9	0.1	0.4	1.4
	OH-MPHP	1.8	5.1	0.4	0.9	4.0
	Oxo-MiNCH	4.8	16.4	0.3	0.9	22.6
	cx-MiNCH	3.2	9.0	0.2	0.9	15.0
	OH-MiNCH	3.9	12.5	0.1	0.8	17.1
	DPP	2.2	1.7	0.6	1.8	5.5
	DBP	0.2	0.2	<LOD	0.1	0.6
	BPA	1.2	1.2	0.2	0.8	3.5
	BPS	0.2	0.4	0.04	0.1	0.7
	4,4-BPF	0.6	2.2	<LOD	0.1	2.4
	2-OH-PH	0.4	0.7	<LOD	0.2	1.2
	TCP	1.8	2.2	0.6	1.3	4.4
	3PBA	0.4	0.6	0.1	0.3	1.4
	TCS	3.7	28.7	<LOD	0.2	1.8
	BHA	11.7	42.8	<LOD	0.6	53.2
	BP-3	11.0	87.4	<LOD	0.7	16.3
Linköping	MEP	141.2	346.1	10.3	36.9	727.9
(n=155-157)	MnBP	50.1	41.2	16.9	38.9	111.2
	MBzP	14.0	22.8	1.1	6.4	47.3
	MEHP	2.6	3.6	0.6	1.8	6.9
	5-OH-MEHP	12.9	26.7	3.1	7.6	41.2
	5-oxo-MEHP	9.9	18.4	2.3	5.9	28.2
	2-cx-MEHP	3.3	5.3	0.8	2.0	8.8
	5-cx-MEPP	11.0	16.4	2.5	6.7	35.7
	OH-MiNP	10.0	49.2	0.9	3.4	28.3
	Oxo-MiNP	3.7	12.3	0.6	1.6	12.7
	cx-MiNP	16.0	84.8	1.8	5.1	28.4
	cx-MiDP	0.7	1.8	0.1	0.3	1.6

Region	Compound	Mean	SD	5th percentile	Median	95th percentile
	OH-MPHP	2.3	6.9	0.4	0.9	4.4
	Oxo-MiNCH	4.0	9.4	0.3	1.2	15.6
	cx-MiNCH	3.7	9.9	0.2	1.1	13.4
	OH-MiNCH	3.4	8.2	0.2	0.9	15.0
	DPP	2.9	4.5	0.6	2.1	6.2
	DBP	0.3	1.0	<LOD	0.2	0.8
	BPA	1.3	1.9	0.3	0.8	4.5
	BPS	0.5	3.9	<LOD	0.1	0.6
	4,4-BPF	2.0	16.6	<LOD	0.1	5.3
	2-OH-PH	0.2	0.3	<LOD	0.1	0.8
	TCP	1.9	2.6	0.5	1.2	6.0
	3PBA	0.4	0.5	0.1	0.3	1.1
	TCS	4.3	34.0	<LOD	0.3	6.2
	BHA	17.2	42.1	<LOD	1.1	87.8
	BP-3	14.8	81.0	<LOD	1.0	30.6
Lund	MEP	122.5	413.9	11.0	34.6	454.9
(n=208-210)	MnBP	50.3	35.6	12.2	40.1	112.1
	MBzP	11.7	19.1	1.4	6.1	41.6
	MEHP	2.3	2.5	0.5	1.6	7.0
	5-OH-MEHP	10.1	14.0	3.1	7.4	21.8
	5-oxo-MEHP	8.1	8.5	2.3	6.0	19.2
	2-cx-MEHP	2.8	3.0	0.7	2.1	7.2
	5-cx-MEPP	9.2	10.0	2.6	7.0	21.5
	OH-MiNP	7.2	12.4	1.2	3.4	28.7
	Oxo-MiNP	3.0	4.1	0.7	1.7	9.9
	cx-MiNP	11.0	18.1	2.2	5.5	41.4
	cx-MiDP	0.5	0.4	0.1	0.4	1.4
	OH-MPHP	1.4	1.3	0.4	1.0	3.8
	Oxo-MiNCH	2.5	5.6	0.3	0.9	8.7
	cx-MiNCH	2.0	4.6	0.1	0.6	8.6
	OH-MiNCH	2.0	4.6	0.1	0.6	7.3
	DPP	2.5	2.3	0.7	1.8	6.3
	DBP	0.2	0.2	<LOD	0.1	0.5
	BPA	1.4	3.5	0.2	0.8	2.7
	BPS	0.2	0.7	0.04	0.1	0.5
	4,4-BPF	0.2	0.4	<LOD	0.1	0.7
	2-OH-PH	0.2	0.2	<LOD	0.2	0.7
	TCP	1.5	1.3	0.5	1.1	4.2
	3PBA	0.4	0.4	0.1	0.3	1.2
	TCS	1.7	8.1	<LOD	0.3	2.4
	BHA	7.0	16.5	<LOD	0.8	41.5

Region	Compound	Mean	SD	5th percentile	Median	95th percentile
	BP-3	14.5	95.6	<LOD	0.8	22.2
Stockholm	MEP	77.6	187.5	8.7	29.0	320.8
(n=167-168)	MnBP	42.7	27.4	14.0	35.4	96.1
	MBzP	8.2	12.7	1.0	4.1	26.1
	MEHP	3.8	22.3	0.6	1.5	5.6
	5-OH-MEHP	16.7	87.5	3.3	7.7	25.2
	5-oxo-MEHP	14.2	81.5	2.3	5.8	20.7
	2-cx-MEHP	3.3	7.5	0.7	1.9	7.1
	5-cx-MEPP	14.7	70.2	2.6	6.8	25.3
	OH-MiNP	15.6	52.4	1.6	4.3	54.2
	Oxo-MiNP	6.3	20.2	0.9	2.0	20.8
	cx-MiNP	22.4	90.0	2.4	6.3	88.0
	cx-MiDP	0.7	1.3	0.2	0.4	1.6
	OH-MPHP	1.7	2.2	0.4	1.2	4.0
	Oxo-MiNCH	3.2	9.6	0.3	1.2	8.8
	cx-MiNCH	2.3	7.0	0.2	0.8	6.0
	OH-MiNCH	2.9	10.4	0.3	1.0	7.3
	DPP	3.0	4.0	0.8	1.9	8.5
	DBP	0.2	0.3	<LOD	0.1	0.7
	BPA	1.5	2.2	0.3	1.0	3.8
	BPS	0.3	0.5	0.04	0.1	0.9
	4,4-BPF	0.6	1.9	<LOD	0.1	2.4
	2-OH-PH	0.3	0.4	<LOD	0.2	1.0
	TCP	1.4	0.9	0.5	1.2	2.8
	3PBA	0.4	0.6	0.1	0.3	0.9
	TCS	4.7	50.9	<LOD	0.3	1.9
	BHA	14.1	50.9	<LOD	0.8	58.5
	BP-3	8.4	55.1	<LOD	0.6	19.6
Umeå	MEP	90.1	188.1	10.2	34.9	266.2
(n=130-131)	MnBP	65.2	49.5	18.8	51.4	144.7
	MBzP	22.1	25.3	3.1	14.7	65.1
	MEHP	2.5	1.9	0.7	1.9	6.5
	5-OH-MEHP	12.3	9.7	3.3	9.1	34.2
	5-oxo-MEHP	9.9	8.1	2.2	7.4	27.3
	2-cx-MEHP	3.3	5.3	0.8	2.2	8.3
	5-cx-MEPP	11.8	10.6	2.9	9.1	36.3
	OH-MiNP	13.6	38.9	1.4	4.6	43.7
	Oxo-MiNP	5.0	11.9	0.7	2.1	14.1
	cx-MiNP	18.6	33.0	2.7	8.7	56.5
	cx-MiDP	0.9	1.4	0.2	0.6	1.9

Region	Compound	Mean	SD	5th percentile	Median	95th percentile
	OH-MPHP	2.6	6.5	0.5	1.4	5.3
	Oxo-MiNCH	3.9	15.7	0.3	1.0	13.1
	cx-MiNCH	3.6	14.5	0.2	0.8	13.0
	OH-MiNCH	3.4	14.4	0.2	0.6	10.6
	DPP	2.7	3.2	0.8	1.8	5.8
	DBB	0.4	0.9	0.1	0.2	0.9
	BPA	1.7	3.0	0.3	0.9	4.3
	BPS	0.5	1.2	0.05	0.2	2.9
	4,4-BPF	0.4	0.8	<LOD	0.1	1.7
	2-OH-PH	0.2	0.1	<LOD	0.2	0.5
	TCP	3.2	6.9	0.6	1.3	9.6
	3PBA	0.4	0.4	0.1	0.3	1.0
	TCS	0.7	2.0	<LOD	0.3	2.0
	BHA	16.8	40.4	<LOD	1.8	94.5
	BP-3	9.9	37.6	<LOD	0.8	44.2
Uppsala	MEP	241.4	977.3	9.4	39.5	636.4
(n=84)	MnBP	61.7	100.3	15.5	44.6	142.0
	MBzP	16.3	23.9	1.9	8.7	72.2
	MEHP	2.6	4.6	0.6	1.6	7.4
	5-OH-MEHP	11.8	15.0	3.3	7.8	26.5
	5-oxo-MEHP	9.2	11.0	2.3	6.2	21.4
	2-cx-MEHP	3.1	4.1	0.8	2.0	6.3
	5-cx-MEPP	8.8	7.7	2.4	6.7	19.4
	OH-MiNP	11.1	32.4	1.4	4.0	27.7
	Oxo-MiNP	4.7	12.7	0.8	2.0	12.6
	cx-MiNP	11.7	18.8	2.1	5.4	36.8
	cx-MiDP	0.5	0.4	0.1	0.4	1.2
	OH-MPHP	1.6	1.8	0.4	1.0	5.6
	Oxo-MiNCH	3.1	5.0	0.3	1.0	16.4
	cx-MiNCH	2.1	4.9	0.2	0.8	7.6
	OH-MiNCH	2.7	4.9	0.2	0.8	15.6
	DPP	3.1	5.0	0.9	2.2	6.2
	DBP	0.3	0.5	0.1	0.2	0.6
	BPA	1.5	1.5	0.3	1.0	4.0
	BPS	0.2	0.2	0.04	0.1	0.7
	4,4-BPF	2.4	10.6	<LOD	0.1	12.0
	2-OH-PH	0.3	0.5	<LOD	0.2	1.1
	TCP	1.6	1.1	0.5	1.3	3.8
	3PBA	0.5	1.3	0.1	0.2	1.2
	TCS	15.1	123.5	<LOD	0.3	4.2
	BHA	10.0	21.3	<LOD	0.9	45.2

Region	Compound	Mean	SD	5th percentile	Median	95th percentile
	BP-3	7.2	17.3	<LOD	1.0	39.3
Örebro	MEP	131.2	430.6	8.4	31.3	412.5
(n=130-132)	MnBP	43.9	31.9	12.3	37.8	92.8
	MBzP	14.2	20.3	1.6	7.3	50.0
	MEHP	2.2	2.3	0.5	1.6	6.0
	5-OH-MEHP	11.0	11.9	2.9	7.7	27.0
	5-oxo-MEHP	8.7	8.5	2.5	6.2	20.0
	2-cx-MEHP	3.4	7.2	0.7	2.0	6.9
	5-cx-MEPP	10.5	13.2	2.7	7.4	25.4
	OH-MiNP	11.4	25.4	1.3	4.7	43.4
	Oxo-MiNP	4.4	10.4	0.7	2.1	15.7
	cx-MiNP	17.2	43.2	2.0	6.7	59.0
	cx-MiDP	0.5	0.5	0.1	0.4	1.7
	OH-MPHP	2.4	8.2	0.4	1.1	5.1
	Oxo-MiNCH	10.5	95.5	0.3	1.0	10.4
	cx-MiNCH	5.0	34.2	0.2	0.8	11.8
	OH-MiNCH	6.9	58.6	0.2	0.8	8.7
	DPP	2.6	3.1	0.7	1.8	6.8
	DBP	0.2	0.2	<LOD	0.1	0.5
	BPA	1.8	3.6	0.3	1.0	3.8
	BPS	0.3	0.4	0.04	0.1	0.9
	4,4-BPF	0.4	1.5	<LOD	0.1	1.4
	2-OH-PH	0.2	0.4	<LOD	0.1	0.9
	TCP	1.4	1.2	0.5	1.1	3.3
	3PBA	0.4	0.4	0.1	0.3	1.2
	TCS	1.9	14.4	0.1	0.3	2.7
	BHA	8.5	18.0	<LOD	0.8	41.7
	BP-3	12.6	73.5	<LOD	0.7	19.5

Table A2.11. Density-adjusted urine concentrations of phthalate metabolites and phenolic compounds (ng/mL) in Swedish adolescents per grade (grade 5, n=332–333; grade 8, n=410–412; grade 11, n=353–359; total n=1,095–1,104). Compounds that were detected in less than 50% of the samples are excluded from the table.

Compound	Grade	Mean	SD	5th percentile	Median	95th percentile
MEP	5	82.5	333.1	7.5	22.2	243.2
	8	99.4	308.7	10.4	34.9	324.7
	11	154.8	517.6	12.4	45.1	583.0
MnBP	5	54.0	44.3	16.7	42.7	123.9
	8	49.5	38.5	16.9	39.3	112.2
	11	52.9	60.1	12.9	39.4	137.1
MBzP	5	13.9	21.6	1.8	6.9	50.9
	8	10.9	15.9	1.1	5.0	38.8
	11	14.7	20.9	1.3	8.0	50.4
MEHP	5	3.3	15.9	0.7	1.7	6.4
	8	2.0	2.6	0.5	1.5	4.6
	11	2.4	2.9	0.7	1.6	7.0
5-OH-MEHP	5	15.9	62.8	3.4	8.9	30.9
	8	10.0	16.9	3.1	7.2	24.3
	11	9.8	13.3	3.0	6.6	22.3
5-oxo-MEHP	5	13.0	58.2	2.9	7.4	25.5
	8	7.7	11.8	2.3	5.5	18.4
	11	8.0	8.9	2.0	5.4	20.6
2-cx-MEHP	5	3.6	6.4	0.9	2.3	7.9
	8	2.9	5.2	0.7	1.9	6.9
	11	2.6	3.2	0.7	1.8	6.0
5-cx-MEPP	5	14.3	50.4	3.4	8.6	33.7
	8	9.0	12.1	2.5	6.5	22.0
	11	8.7	9.6	2.5	6.0	23.3
OH-MiNP	5	9.4	36.4	1.4	3.7	29.0
	8	12.2	46.9	1.1	3.4	39.5
	11	11.8	31.8	1.3	4.2	39.6
Oxo-MiNP	5	3.9	13.3	0.7	1.8	10.5
	8	4.8	16.0	0.6	1.7	15.0
	11	4.7	10.7	0.7	2.0	15.0

Compound	Grade	Mean	SD	5th percentile	Median	95th percentile
cx-MiNP	5	13.7	54.2	2.1	5.7	35.0
	8	18.1	80.1	2.0	5.5	55.0
	11	16.0	30.0	2.2	6.4	56.5
cx-MiDP	5	0.6	0.8	0.1	0.4	1.5
	8	0.6	1.3	0.1	0.4	1.4
	11	0.6	1.0	0.1	0.4	1.6
OH-MPHP	5	1.6	2.8	0.4	1.0	4.1
	8	2.0	5.8	0.4	0.9	5.1
	11	2.2	5.9	0.4	1.1	4.7
oxo-MiNCH	5	2.7	10.0	0.3	0.9	8.1
	8	3.3	11.5	0.3	1.0	10.4
	11	7.4	58.9	0.3	1.2	18.3
cx-MiNCH	5	2.1	8.9	0.2	0.8	5.6
	8	2.4	6.7	0.2	0.8	9.1
	11	4.8	22.7	0.2	0.8	16.4
OH-MiNCH	5	2.3	9.1	0.2	0.7	6.5
	8	2.8	9.9	0.2	0.8	8.5
	11	5.5	36.7	0.2	0.8	15.5
DPP	5	2.7	3.4	0.7	1.8	6.6
	8	2.5	3.0	0.7	2.0	5.5
	11	2.8	3.6	0.7	1.9	7.7
DBP	5	0.3	0.8	<LOD	0.2	0.7
	8	0.2	0.3	<LOD	0.1	0.6
	11	0.2	0.4	<LOD	0.1	0.7
BPA	5	1.4	1.8	0.3	1.0	4.5
	8	1.2	1.9	0.2	0.8	3.2
	11	1.7	3.7	0.3	1.0	4.1
BPS	5	0.2	0.5	0.04	0.1	0.7
	8	0.3	2.5	0.04	0.1	0.6
	11	0.3	0.7	0.04	0.2	0.9
4,4-BPF	5	0.3	1.3	<LOD	0.1	1.3
	8	1.2	11.3	<LOD	0.1	1.4

Compound	Grade	Mean	SD	5th percentile	Median	95th percentile
	11	0.8	2.2	<LOD	0.1	4.0
2-OH-PH	5	0.2	0.3	<LOD	0.1	0.6
	8	0.3	0.5	<LOD	0.2	1.1
	11	0.3	0.3	<LOD	0.2	0.8
TCP	5	1.9	3.1	0.5	1.2	4.2
	8	1.6	1.5	0.5	1.2	4.4
	11	2.0	3.8	0.5	1.2	5.0
3PBA	5	0.5	0.8	0.1	0.3	1.3
	8	0.4	0.4	0.1	0.3	1.0
	11	0.4	0.6	0.1	0.2	1.2
TCS	5	0.7	3.7	<LOD	0.2	1.4
	8	5.6	64.8	<LOD	0.3	2.4
	11	4.6	32.4	<LOD	0.3	3.4
BHA	5	5.8	19.0	<LOD	0.5	25.8
	8	11.3	33.9	<LOD	0.9	62.2
	11	18.7	48.9	<LOD	1.6	94.5
BP-3	5	2.8	19.5	<LOD	0.5	7.9
	8	20.0	114.8	<LOD	0.8	26.5
	11	10.1	35.4	<LOD	1.3	50.8

Table A2.12. Density-adjusted urine concentrations of phthalate metabolites and phenolic compounds (ng/mL) in Swedish adolescents per gender (boys, n=479–482; girls, n=616–622; total n=1,095–1,104). Compounds that were detected in less than 50% of the samples are excluded from the table.

Compound	Gender	Mean	SD	5th percentile	Median	95th percentile
MEP	Boys	79.4	325.2	9.1	27.0	212.6
	Girls	137.8	441.3	9.9	39.4	550.2
MnBP	Boys	49.7	43.9	14.0	37.5	130.0
	Girls	53.7	51.1	16.9	41.9	117.2
MBzP	Boys	12.0	17.6	1.1	6.1	40.4
	Girls	13.9	20.8	1.4	7.1	50.0
MEHP	Boys	2.1	1.9	0.6	1.5	5.5
	Girls	2.9	11.9	0.6	1.7	6.4
5-OH-MEHP	Boys	10.1	10.2	3.1	7.6	24.5
	Girls	13.0	48.3	3.2	7.8	26.3
5-oxo-MEHP	Boys	7.8	7.5	2.1	5.7	20.5
	Girls	10.6	43.7	2.5	6.2	21.3
2-cx-MEHP	Boys	2.9	4.8	0.8	2.0	6.3
	Girls	3.1	5.2	0.8	2.1	7.2
5-cx-MEPP	Boys	8.9	9.6	2.5	6.7	22.4
	Girls	11.7	38.1	2.7	7.2	26.2
OH-MiNP	Boys	8.2	17.2	1.2	3.5	31.6
	Girls	13.6	50.1	1.3	3.9	39.6
Oxo-MiNP	Boys	3.3	6.5	0.6	1.6	11.0
	Girls	5.3	17.2	0.7	1.9	15.0
cx-MiNP	Boys	11.1	19.4	1.9	5.4	41.4
	Girls	19.9	77.6	2.2	6.3	59.0
cx-MiDP	Boys	0.5	0.4	0.1	0.4	1.3
	Girls	0.7	1.4	0.1	0.4	1.6
OH-MPHP	Boys	1.7	5.1	0.4	1.0	4.0
	Girls	2.1	5.2	0.4	1.0	4.9
Oxo-MiNCH	Boys	3.4	10.9	0.3	0.9	12.6

Compound	Gender	Mean	SD	5th percentile	Median	95th percentile
	Girls	5.3	45.2	0.3	1.1	11.9
cx-MiNCH	Boys	2.5	7.6	0.2	0.7	10.6
	Girls	3.5	18.0	0.2	0.8	12.1
OH-MiNCH	Boys	2.9	8.9	0.2	0.7	10.7
	Girls	4.1	28.7	0.2	0.8	11.9
DPP	Boys	2.3	2.6	0.6	1.8	5.8
	Girls	2.9	3.8	0.8	2.0	6.7
DBP	Boys	0.2	0.7	<LOD	0.1	0.6
	Girls	0.3	0.4	<LOD	0.2	0.7
BPA	Boys	1.5	3.0	0.2	0.9	3.8
	Girls	1.4	2.3	0.3	0.9	3.9
BPS	Boys	0.2	0.4	0.03	0.1	0.6
	Girls	0.4	2.1	0.04	0.1	0.9
4,4-BPF	Boys	1.2	10.5	<LOD	0.1	2.4
	Girls	0.5	1.7	<LOD	0.1	2.0
2-OH-PH	Boys	0.3	0.3	<LOD	0.2	0.9
	Girls	0.3	0.5	<LOD	0.2	0.9
TCP	Boys	1.7	2.3	0.5	1.1	4.5
	Girls	1.9	3.3	0.5	1.2	4.5
3PBA	Boys	0.4	0.7	0.1	0.2	1.0
	Girls	0.4	0.6	0.1	0.3	1.1
TCS	Boys	2.1	20.7	<LOD	0.3	1.7
	Girls	5.1	55.4	<LOD	0.3	2.7
BHA	Boys	6.9	30.9	<LOD	0.3	33.9
	Girls	16.0	40.0	<LOD	1.9	78.7
BP-3	Boys	3.3	17.8	0.1	0.5	11.3
	Girls	18.1	97.0	0.2	1.1	46.8

The present report summarises the results from analysis of contaminants in blood and urine samples from participants in the dietary survey *Riksmaten Adolescents 2016–17*. These biomonitoring data provide unique information on total exposure to contaminants from all sources, including food, in Swedish adolescents. The results will be used further in risk assessments of contaminants in food by the Swedish Food Agency (Livsmedelsverket). Data from the project is also part of the national health-related environmental monitoring at the Swedish Environmental Protection Agency (Naturvårdsverket). The aim of this monitoring is to estimate human exposure to hazardous substances, follow temporal trends in human exposure, and to link environmental exposure to effects on health. The results from this report may also be useful for experts working with risk assessment and risk management in other organizations at the national or regional level.

Livsmedelsverket är Sveriges expert- och centrala kontrollmyndighet på livsmedelsområdet. Vi arbetar för säker mat och bra dricksvatten, att ingen konsument ska bli lurad om vad maten innehåller och för bra matvanor. Det är vårt recept på matglädje.

Livsmedelsverket