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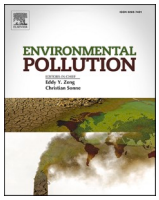

Microplastics in feed affect the toxicokinetics of persistent halogenated pollutants in Atlantic salmon☆

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ABSTRACT

Microplastics (MPs) are carriers of persistent organic pollutants (POPs). The influence of MPs on the toxicokinetics of POPs was investigated in a feeding experiment on Atlantic salmon (*Salmo salar*), in which fish were fed similar contaminant concentrations in feed with contaminants sorbed to MPs (*Cont. MPs*); feed with virgin MPs and contaminated feed (1:1), and feed with contaminants without MPs (*Cont.*). The results showed that the salmon fillets accumulated more POPs when fed with a diet where contaminants were sorbed to the MPs, despite the 125–250 μm size MPs themselves passing the intestines without absorption. Furthermore, depuration was significantly slower for several contaminants in fish fed the diet with POPs sorbed to the MPs. Modelled elimination coefficients and assimilation efficiencies of lipophilic chlorinated and brominated contaminants correlated with contaminant hydrophobicity (log K_{ow}) within the diets and halogen classes. The more lipophilic the contaminant was, the higher was the transfer from feed to salmon fillet. The assimilation efficiency for the diet without MPs was 50–71% compared to 54–89% for the contaminated MPs diet. In addition, MPs caused a greater proportional uptake of higher molecular weight brominated congeners. In the present study, higher assimilation efficiencies and a significantly higher slope of assimilation efficiencies *vs* log K_{ow} were found for the *Cont. MPs* diet ($p = 0.029$), indicating a proportionally higher uptake of higher-brominated congeners compared to the *Cont.* diet. Multiple variance analyses of elimination coefficients and assimilation efficiencies showed highly significant differences between the three diets for the chlorinated ($p = 2E\text{-}06$; 6E-04) and brominated ($p = 5E\text{-}04$; 4E-03) congeners and within their congeners. The perfluorinated POPs showed low assimilation efficiencies of <12%, which can be explained by faster eliminations corresponding to half-lives of 11–39 days, as well as a lower proportional distribution to the fillet, compared to e.g. the liver.

1. Introduction

Microplastics (MPs) are present in the marine environment in concentrations estimated to exceed safe concentrations for several aquatic species soon (Egger et al., 2022). About 11.6 to 21.1 million tons of plastic litter of size class 32–651 μm, primarily polyethylene (PE), polypropylene (PP), and polystyrene (PS), were estimated to be suspended in the top 200 m of the Atlantic Ocean. The particles <100 μm constituted 2-3 orders higher mass concentration than the larger MPs (Pabortsava and Lampitt, 2020). MPs can be divided into two groups depending on their source. Primary MP is defined as MPs produced from various products, cosmetics, detergents, and polymers, from which they can be released into the environment (ter Halle et al., 2017). Secondary MP is the degradation of plastics into MPs (size <5 mm-1 μm) which can further degrade into nanoplastics (NPs, size <1000 nm) through physical, chemical, or biological processes (Hartmann et al., 2019).

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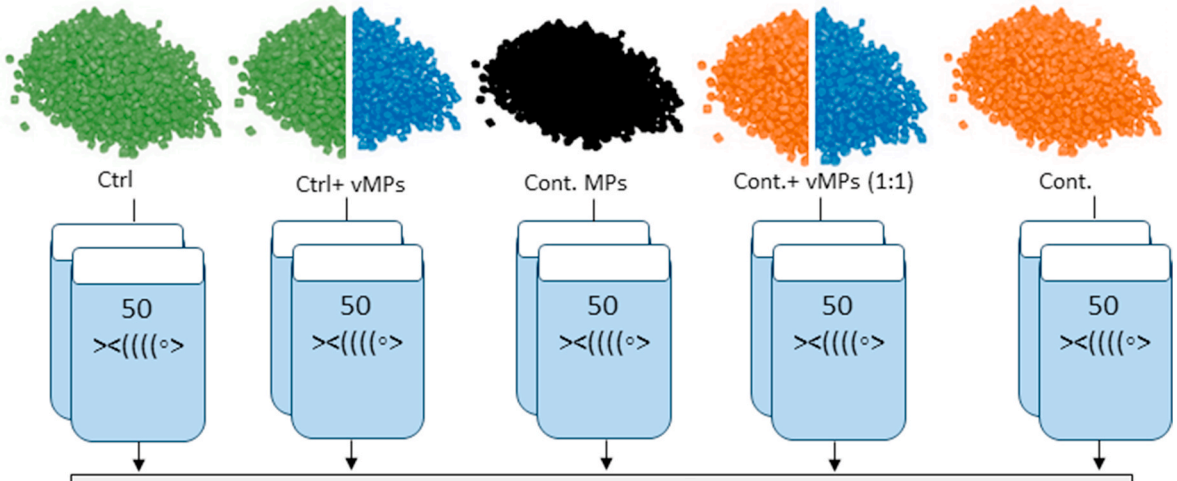


			Accumulation						Elimination					
Diet		Days	0	19	38	57	76	98	101	105	112	119	126	154
B1	Ctrl		2					2x4						2x4
B2	Ctrl + vMPs		2					2x4						2x4
1)	Cont. MPs		2	2x4	2x4	2x4	2x4	2x4	2x3	2x3	2x3	2x3	2x3	2x4
2)	Cont.+ vMPs		2	2x4	2x4	2x4	2x4	2x4	2x3	2x3	2x3	2x3	2x3	2x4
3)	Cont.		2	2x4	2x4	2x4	2x4	2x4	2x3	2x3	2x3	2x3	2x3	2x4
			10	32	32	32	32	40	18	18	18	18	18	40

Fig. 1. Overview of the salmon feeding experiment, and scheme for sampling in the accumulation and elimination phase. *Ctrl*: regular feed without MPs, *Ctrl + vMPs*: regular feed with MPs, *Cont. MPs*: feed with contaminants sorbed to MPs, *Cont.+ vMPs*: feed with contaminants mixed with virgin MPs (1:1) at the time of feeding, *Cont.*: feed with contaminants sorbed to feed pellets.

The presence of MPs in aquatic environments is of serious concern to marine life because the animals can ingest MPs directly or indirectly through the food chain. The particle size of ingested MPs affects marine life. Larger MPs can affect feed passage, but MPs of 100–300 μm size range could pass through the digestive system of fish, whereas 5 μm MP can be translocated across the intestinal epithelium into the circulatory system and thereby accumulate in the liver, as shown in zebrafish (*Danio rerio)* (Lu et al., 2016). The smaller MPs can even be absorbed at the cellular level. Immune cells in Atlantic salmon's intestine, blood, and kidney could phagocytize PE-MPs of 1–5 μm at low concentrations (Abihssira-García et al., 2020). The presence of MPs in the marine environment may have both human health and environmental impacts. People often consume seafood as part of their diet, exposing them to pollutants.

Along with MPs, persistent organic pollutants (POPs), such as polychlorinated biphenyls (PCBs), brominated flame retardants (BfR) polybrominated diphenyl ethers (PBDEs), hexabromocyclododecanes (HBCDs), and per- and polyfluorinated alkyl substances (PFAS) bioaccumulate in wild fish when they ingest contaminated prey. POPs are also present in aquaculture through marine feed ingredients such as fishmeal and fish oil (Berntssen et al., 2011a,b). The co-occurrence of MPs and POPs raises additional concerns. Some POPs have a high affinity for plastic, with concentration factors of up to 10^6 (Mato et al., 2001). PE and PP are rubbery amorphous polymers that favor the absorption of chlorinated and brominated POPs, involving partitioning into the material of molecules by relatively weak van-der-Waals forces (Hartmann et al., 2017). The more hydrophobic the compounds are, the higher the affinity for the polymer, which generally correlates with the octanol-water partition coefficient (O'Connor et al., 2016). PFAS are surfactants that have a non-polar part (hydrophobic perfluorinated carbon part that can bind to MPs by van-der-Waals forces) and a polar ionized part (carboxylic or sulfonic acid) that MPs cannot absorb, thus potentially being adsorbed to MPs (Bakir et al., 2014; Cormier et al., 2022). After absorption or adsorption, the MPs can act as chemical vectors. Abihssira-García et al. (2022) demonstrated that MPs placed next to salmon cages were capable of sorbing POPs leaching from salmon feed and concluded that MPs may play a role in transporting POPs from marine aquaculture. While MPs are known to sorb POPs and can transport and distribute them in fish, their exact role in this transfer remains to be elucidated. The dietary assimilation efficiency, *i.e.*, the percentage of contaminants from the feed ending up in the fish fillet, depends on bioaccessibility and bioavailability from the gastrointestinal system, metabolism, excretion, distribution, and bioaccumulation potential. Teuten et al. (2007) demonstrated that the desorption rate of phenanthrene from PE was much higher when a digestive surfactant (a cholesterol-derived bile salt) was added. Bakir et al. (2014) also demonstrated enhanced desorption of POPs from PE-MPs using digestive simulation studies, where the DDT (Dichlorodiphenyltrichloroethane) desorption rate was 30 times higher than in seawater. Bakir et al. (2016) included enhanced bioavailability from intestinal surfactants in modelling the influence of MPs on the transfer of hydrophobic contaminants from food and water into fish fillets; they found a negligible effect of the presence of MPs. However, they did not rule out that there might be scenarios where the presence of MPs makes a more important contribution. In an experiment with rainbow fish (*Melanotaenia fluviatilis*) fed PBDEs absorbed to PE-MPs (10–700 μm), Wardrop et al. (2016) showed that PBDEs desorbed from the MPs crossed the GI system and were distributed to the fillets. In an experiment with European seabass (*Dicentrarchus labrax*), Granby et al. (2018) found that the presence of MPs in feed affected the toxicokinetics of halogenated contaminants and the gene expression of the liver. In zebrafish (*Danio rerio*) exposed to contaminated feed with MPs, the gene expression results indicated that the co-occurrence of MPs potentiated the toxic effects of some halogenated contaminants (Rainieri et al., 2018).

There is a lack of long-term feeding studies with MPs and fish, especially for aquaculture-relevant species such as Atlantic salmon (*Salmo salar*). Therefore, it is valuable to widen our knowledge about MPs as contaminant carriers and understand their impact on accumulation and elimination in this widely farmed and consumed marine species. In this study, we measured the toxicokinetic parameters of ubiquitous POPs (PCBs, PBDEs, HBCD, and PFAS) in Atlantic salmon in the presence and absence of MPs.

2. Materials and methods

2.1. Feeding experiment

Fish tanks: The experiment was conducted at the North Sea Science Centre, Hirtshals, Denmark. The Animal Experiments Inspectorate of the Ministry of Food, Agriculture and Fisheries of Denmark has granted the approval of the experiment. Salmon specimens were obtained from Biomar A/S, Hirtshals, Denmark. The experiment was carried out in a flow-through system of 10 fiberglass tanks (1.18 m × 1.18 m x 0.65 m; 900 L seawater) with rounded corners and equipped with a whirl separator, enabling the collection of uneaten pellets. Fish were distributed evenly in 10 tanks. Seawater (salinity ~ 28–30‰) was pumped from the North Sea off Hirtshals (57.61N, 9.98E) to a storage tank and split into two lines; one line was oxygenated with pure oxygen in an oxygen cone and afterward led to the tanks together with the non-oxygenated line assuring oxygen saturation >80% throughout the trial. The water temperature was only adjusted by regulating room temperature and water flow and decreased from 14 °C at the start of accumulation to 10-7 °C during the elimination. The fish were kept at 24-h light, the standard procedure for salmon, because they need an extended feeding period, and on/off light cycles cause them stress. Feeding was done by belt feeders for 16–18 h per day. During the acclimatization, they were fed a commercial diet (Biomar A/S), but two weeks before the trial, they were fed the experimental plastic-free control diet. At the onset of the trial, two tanks per diet were randomly selected between the ten tanks. The feeding ratio was calculated from estimated biomass and expected feed conversion ratio (FCR = 1). The 23-week experiment included a 98-day accumulation period followed by a 56-day elimination period during which all groups were fed the plastic-free control diet (scheme in Fig. 1).

Fish sampling: Two weeks before the experiment, all fish were anaesthetised with benzokain (Benzoak® Vet 200 mg/L(ACD Pharmaceuticals AS, Leknes, Norway), 15–20 mL/100 L water and passive integrator transponder(pit)-tagged in the dorsal muscle (RFID tags, Loligo Systems, Denmark). They were not fed the day before and on the day of weighing. They were allowed to recover from this procedure for two weeks before they were anaesthetised again and weighed individually. Growth rates could then be obtained for each fish sampled for analysis. The total biomass per tank was recorded to estimate the feed ration and calculate the feed conversion ratio (FCR: g eaten feed/g biomass gained) for each growth period and the feeding rate per diet (F: g feed/g fish/day). The specific growth rate (SGR) for individual fish was calculated (Eq. (1)). From that, the growth factor GF is calculated (Eq. (2)). It expresses dilution due to fish growth exemplified in the graphical abstract by its rate constant k_G. Three to four fish were sampled per tank (6–8 per diet) and weighed, and one fillet per fish was frozen (-20 °C) in pools per tank.

$$SGR = (Ln\, w_2 – Ln\, w_1) \bullet t^{-1} \quad (1)$$

$$GF = (1 + SGR \cdot t) \quad (2)$$

2.2. Feed preparation

The experimental feeds (4.5 mm extruded pellets) were produced by the Danish Technological Institute (Kolding, Denmark). The feed formulation consisted of a nutritionally optimal mix of fishmeal, fish oil, wheat binder, vitamins, and minerals (Biomar A/S) (Table S1). The oil was spray-coated onto the extruded pellets in a rotating mixer. Five experimental diets were included in the trial (Fig. 1); two control diets with low chemical contamination, one of them with added 2% MPs to the pellets; and three contaminated diets: [1)]*contaminated (cont.) MPs* with the contaminants being sorbed to the MPs before incorporation into pellets (see below); [2)] *cont.+virgin (v)MPs*, the contaminants were coated on feed pellets, which were fed (1:1) together with the control feed with pellets containing MP, *i.e.*, the plastic particles did not 'encounter' the contaminants until they reached the fish's digestive tract, and [3)] *contaminants (cont.)* without MPs. Overnight mixing of a solution of contaminants with the LD-PE MP allowed the contaminants to be sorbed to the MP for which they have a high affinity before being added to the feed. The contaminants were expected to be further sorbed to the MPs during the four months of storage before the experiment. Coating of contaminants on the feed pellets was performed by adding contaminants to the fish oil, which was then spray-coated onto the extruded pellets. Coating of MPs with contaminants was performed by solubilising the contaminants in ethanol and then mixing with the particles overnight at ambient temperature, allowing the ethanol to evaporate and the MPs to sorb the contaminants. The five individual experimental diet groups were established by either one of the diets or diet mixes. It was assumed that, unless there were some noticeable nutritional differences between the diets, the experimental groups that were fed one of the diets mixes with chemical contaminants ingested even amounts of contaminated and non-contaminated pellets, and thus the same amount of contaminants. Following the same setup, diet groups fed with MP-containing feed ingested 2% MPs. The MPs were made from low-density polyethylene (LD-PE) (RIBLON 30) milled to a particle size <400 μm, which was sieved into the size range of 125–250 μm. A "pre-wash" with octane and pentane was performed to extract soluble plastic monomers and remove very fine attached plastic particles. The MP size range was selected so that they would pass the gastrointestinal (GI) system without blockage and without affecting the epithelium, e.g., at < 20–30 μm, by inducing histological changes or inflammatory responses and possibly crossing the digestive tract. The three contaminant diets/diets mixes contained 7–150 ng g^{-1} lipid in feed of α-HBCD, PBDE-47, -99, −153, −154 (congener composition of DE71, La Guardia et al., 2006) and PCB congeners 28, 52, 101, 138, 153, 180. In addition, the feed contained ~ 7–12 ng g^{-1} of perfluorooctane sulphonic acid (PFOS), perfluorooctanoic acid (PFOA), and perfluorononanoic acid (PFNA).

2.3. Chemical analysis

Sample preparation: The pooled samples of 3–4 fish fillets without skin were homogenized in a blender made of stainless steel (Braun, Germany) and stored in sealed PE bags at −20 °C until chemical analyses.

Chemicals: Acetone and methanol were from Rathburn (Walkerburn, UK); hexane, ethanol, acetic acid, and concentrated sulphuric acid were from Merck (Darmstadt, Germany); anhydrous sodium sulphate from VWR (Radnor, PA, USA); the water was from a Milli-Q Gradient A10 system (Millipore, Bedford, MA, USA). α-, β-, γ-Hexabromocyclododecane solutions (50 ± 2.5 μg mL^{-1}) as well as $^{13}C_{12}$-α-HBCD, PFAS mixture (2 μg mL^{-1} in methanol) and corresponding isotopically labelled standard mixture were from Wellington laboratories (Ontario, Canada). PCB and PBDE congeners were from Cambridge Isotopes Laboratories Inc. (Andover, MA, USA).

Analyses:, PCBs, PBDEs, α-HBCD: Four g fish sample was mixed with 60 g of anhydrous sodium sulphate and Soxhlet extracted for 6 h with 150 mL of acetone: hexane (1:1). The extract was vacuum evaporated to 10 mL, of which aliquots of 2.5 mL were taken for lipid determination and PCB, PBDE analyses. The remaining extract was evaporated and dissolved in 2.0 mL ^{13}C-labelled α-HBCD solution in hexane and cleaned with 10 mL concentrated sulphuric acid by rotating the tube x 20. The

hexane phase was rinsed with Milli-Q water, evaporated using nitrogen, and 200 μL methanol was added. The PCB, PBDE extract was also vacuum evaporated, dissolved in 2 mL hexane, added ^{13}C-labelled internal PCB and PBDE standards, and cleaned up with concentrated sulphuric acid and Milli-Q water as for the HBCD-analyses. The clear solvent layer was evaporated to 200 μL using nitrogen and transferred to GC vials. The lipid content was determined by weighing after drying the 2.5 mL aliquots at 70 °C to constant weight. *PFAS:* A 2 g sample was extracted with 20 mL methanol, vacuum evaporated, and the residue digested using 18 mL 0.1 M KOH; after neutralization with formic acid, samples were cleaned up using 6 cc/150 mg Oasis WAX SPE columns (Waters Corporation, Milford, MA, USA). The PFAS were eluted with 2% ammonium hydroxide in methanol.

Detection. The LC-MS/MS detection of α-HBCD was performed on an Agilent HPLC (Agilent Technologies, Santa Clara, CA, USA) coupled to a Micromass Quatro Ultima tandem mass spectrometer (Waters, Hertfordshire, UK). The separation of 15 μL of the analytes was made on a Kinetex C18-HPLC column 2.6 μm, 100 × 2.1 mm (Phenomenex, Copenhagen, Danmark) using a gradient of 0.01% acetic acid (A) and methanol (B) at a flow of 0.2 mL min^{-1}: at 0 min 50% B, 10–23 min 95% B, 33 min 50% B. The MS source temperature was 120 °C, and the desolvation temperature was 380 °C. The collision energy was 35 eV, and the argon collision gas pressure was 2.3e^{-3} mbar. The LC-MS/MS detection of PFAS was performed using Dionex Ultimate 3000/Bruker EVOQ Elite UPLC-MS/MS (cone 3000V, cone temp. 350 °C) (Dionex Corp., Sunnyvale, CA, USA/Bruker Corp., Billerica, MA, USA) and an Acquity UPLC CSH C18 (130 Å, 1.7 μm, 100 × 2.1 mm) column (Waters, Denmark). The injection volume was 10 μL, the column oven temperature was 50 °C, and the autosampler temperature was 10 °C. The eluents used were A) 2 mM ammonium acetate in Milli-Q water: methanol (90:10), adjusted with 25 % ammonia solution to pH 8.5, and B) methanol. Eluent flow initially 0.15 mL min^{-1}; Eluent program; 0–0.5 min 10% B; 5.5 min 60% B; 12 min 95% B 0.2 mL min^{-1}; 16 min 95% B at 0.2 mLmin^{-1}; 16.1 min 10% B at 0.15 mL min^{-1} until 23 min. The GC-high resolution MS detection was performed on a Waters Auto Spec Premier™ instrument in electron impact (EI) mode, monitoring in selected ion recording (SIR) at resolution 10,000. Two ion fragments were monitored for each of the analysed PCB and PBDE congeners from the chlorine and bromine isotope clusters, respectively. The MS conditions were ion source temperature 260 °C, electron energy 40 eV, trap 650 μA and GC/MS- interface temperature 270 °C–280 °C. The GC column was a DB-5MS (Agilent Technologies), 60 m, i.d. 0.25 mm, film thickness 0.25 μm. The temperature programme was 90 °C for 2 min; 20 °C min^{-1} to 180 °C; 2 °C min^{-1} to 260 °C; 5 °C min^{-1} to 320 °C; 320 °C kept for 10 min.

Quality assurance: The laboratory is accredited by the Danish accreditation body DANAK and participates in proficiency tests arranged by the European Reference Laboratory for halogenated POPs. The recoveries of PCB and BfR spiked at 4 ng g^{-1} ~40 ng g^{-1} lipid weight (lw) to salmon fillet were PCBs 93%–101%, PBDEs 91%–95%, and α-HBCD 98–106%. The recoveries of 1 ng g^{-1} spiked PFAS to salmon fillet were PFOA 95%, PFNA 77%, and PFOA 83%.

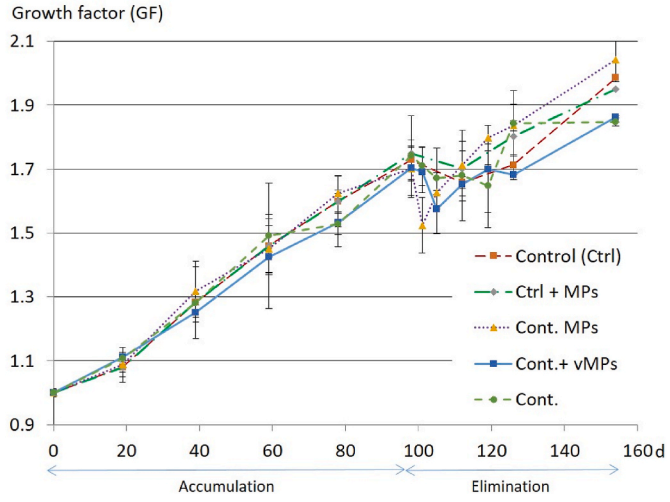


Fig. 2. The growth factor of duplicate fish samples of 3–4 fish relative to initial weight (mean ± SD per diet) during the accumulation and elimination phase. The GF was calculated as 1+ SGR˙*t* where SGR is the specific growth rate and t is the duration of growth.

2.4. Modelling the toxicokinetics

The model for determining the toxicokinetic parameters is based on the assumption that the uptake of contaminants in fish occurs via feed consumption. The salmons ate most of the pellets as they entered the water, giving little time for POPs to diffuse into the water. In addition, the POPs have high PE-water partitioning coefficients (Lohmann, 2012) and will remain in the fish oil of the feed or sorbed to the MPs instead of dissolving in the water. Regarding perfluorinated acids (PFAA), Sun et al. (2022) found that ingestion is the dominant uptake route for fish.

The contaminant concentration in the fish is a function of time (Eq. (3)), where α is the assimilation efficiency, which describes the carry-over of the contaminant from the feed to the fish; F is the feeding rate; C_{feed} the feed contaminant concentration; C_{fish} the contaminant concentration in the fish multiplied by a growth factor (GF); and k_{el} is the elimination coefficient. GFs specific to individual fish samples are used to compensate for dilution due to fish growth. For the C_{fish} based on lw, the feeding rates (F) per diet used for modelling are also per lw. The elimination coefficient k_{el} is the slope of the elimination regression of the natural logarithm of the contaminant concentration *vs* time (Eq. (4)). The half-life ($t_{½}$) is estimated from k_{el} (Eq. (5)).

From the calculated k_{el} and measured C_{feed}, the assimilation efficiency (α) is modelled from the C_{fish} results of the accumulation. By mathematical integration of Eq. (3), the concentration in the salmon fillet C_{fish} is derived, and rearranging this equation yields the assimilation efficiency α. (Eq. (6)). Using an iterative model on Eq (6), α is obtained when the measured C_{fish} of the accumulation curve is best fitted (lowest sum of variances in deviations between measured and modelled concentrations, Fig. S6). The control feed given during the elimination period contained background contamination which, for the chlorinated and brominated compounds with long half-lives, slightly affected k_{el} and α, and are therefore adjusted for by calculating a new $k_{el,\ adj}$ and α_{adj} from Eqs. (7) and (8).

$$dC_{fish}/dt = \alpha \bullet F \bullet C_{feed} - k_{el} \bullet C_{fish} \quad (3)$$

$$Ln\ C_{fish} = constant - k_{el} \bullet t \quad (4)$$

$$t_{½} = ln\ 2/k_{el} \quad (5)$$

$$C_{fish} = (\alpha \bullet F \bullet C_{feed} / k_{el}) \bullet (1 - exp^{-k_{el} \bullet t}) => \quad \alpha = \frac{C_{fish} \bullet k_{el}}{F \bullet C_{feed} \bullet (1 - exp^{-k_{el} \bullet t})} \quad (6)$$

$$Ln\ (C_{fish} - \alpha \bullet F \bullet C_{control\ feed}) = constant - k_{el\ adj} \bullet t \quad (7)$$

$$C_{fish} = (\alpha \bullet F \bullet C_{feed} / k_{el\ adj}) \bullet (1 - exp^{-k_{el\ adj} \bullet t}) => \quad \alpha = \frac{C_{fish}\ k_{el\ adj}}{F \bullet C_{feed} \bullet (1 - exp^{-k_{el\ adj} \bullet t})} \quad (8)$$

2.5. Statistics

The slopes of the elimination (k_{el}) and accumulation, as well as the

Table 1
Mean concentration in the feed, log K_{ow}a)/log D_{ow} (anion) b), assimilation efficiencies, elimination coefficients, and half-lives for PCBs HBCD, PBDEs (lw based), and PFAS (ww based) in fillets of salmon fed the contaminated diets (Standard Errors in Table S2).

POPs/mean C_{feed}	log D_{ow} a'b)	Assimilation efficiency (ɑ) (%)			Elimination coefficient (k_{el}) (10^{-2} d^{-1})			Half-live ($t½$) (d)		
		Cont. MPs	Cont. + vMPs	Cont.	Cont. MPs	Cont. + vMPs	Cont.	Cont. MPs	Cont. + vMPs	Cont.
CB28/34 ng g^{-1} lw	5.67	55	50	55	−0.01	0.26	0.42	–	264	164
PCB52/36 ng g^{-1} lw	5.84	59	46	54	0.01**	0.17*	0.64	–	411	108
PCB101/44 ng g^{-1} lw	6.38	63	58	60	0.14**	0.42	0.82	509	164	85
PCB138/147 ng g^{-1} lw	6.83	66	57	69	0.19**	0.34**	0.92	358	206	75
PCB153/75 ng g^{-1} lw	6.92	69	59	68	0.25*	0.29*	0.79	278	239	88
PCB180/7.8 ng g^{-1} lw	7.36	81	64	71	0.29*	0.28	0.74	239	244	94
ɑ-HBCD/28 ng g^{-1} lw	5.63	54	41	50	0.15**	0.27	0.68	456	255	97
PBDE47/81 ng g^{-1} lw	6.76	57	52	51	0.13	0.32	0.37	527	215	185
PBDE99/134 ng g^{-1} lw	7.27	71	59	59	0.25	0.49	0.56	278	141	125
PBDE153 /15 ng g^{-1} lw	7.58	77	65	70	0.21*	0.41	0.57	324	171	121
PBDE154 /11 ng g^{-1} lw	7.89	89	72	71	0.34	0.54	0.51	203	129	137
PFOA/9.6 ng g^{-1}	1.58	2.7	4.5	3.4	2.0	3.7	6.1	35	19	11.4
PFNA/8.3 ng g^{-1}	2.30	11	6.1	4.3	4.8	3.9	5.5	15	18	12.6
PFOS 10.4 ng g^{-1}	3.05	11	(18)	8.8	1.8	(1.9)	2.0	39	(37)	35

*) significant different slopes between the diet and the cont. diet without MPs at 5 % significance, **) 1 % significance.
[a] reference for log K_{ow}: HBCD (Braekevelt et al., 2003); PCBs (Hawker and Connell, 1988); PBDEs (Li et al., 2008).
[b] reference for log D_{ow} (anion) used for PFAS (Park et al., 2020).

slopes of assimilation efficiencies of contaminants from the different diets, are compared using a statistical program at www.real-statistics.com (Zaiontz C, 2023)(theory: Paternoster et al., 1998). The accumulation lines are normalised relative to the diet without MPs to compensate for minor differences in feed contaminant concentrations and feeding rates. Differences in k_{el} and ɑ between the three contaminated diets within PCBs, PBDEs, and PFAS are tested by paired t-tests and by two-factor ANOVA without replication (Excel Analysis ToolPak).

3. Results and discussion

3.1. Salmon growth, growth factors (GF) and feeding rates (F)

The individual salmons weighed before the feeding experiment 214 ± 33 g (n = 361), and those sampled at the end of the elimination period weighed 566 ± 83 g (n = 40). In comparison, the mean weight between diets was 214 ± 4 g at the start and 565 ± 32 g at the end of the experiment. During the accumulation period up to day 98, the GF of the salmon in the five diets was not significantly different (Fig. 2). At the beginning of the elimination period, a decrease in growth was observed, which can be explained by the decline in fish feeding during handling of fish by weighing. After a week, they became less stressed and continued to gain weight. As the lipophilic POPs are calculated per lipid weight, changes in % lipid in fillets may indirectly affect C_{fish}. The mean lipid contents of the two pooled fillet samples per diet ranged from 5 to 7% *(w/w)* (Fig. S1). A general trend was observed of a decrease in fillet lipid contents during the first 39 days of the accumulation period and the first half of the elimination period, after which fish lipids increased until the end of the periods. There was no difference in the lipid content of the salmon fillets between the diets. The feeding rates (F) during accumulation were 0.0231; 0.0236; 0.0223 g feed lipid/g fish lipid/day corresponding to 6.67; 6.67; 6.54 mg feed/g fish/day for the diets with *Cont. MPs, Cont.+vMPs,* and *Cont.*

3.2. Feed contaminant concentrations

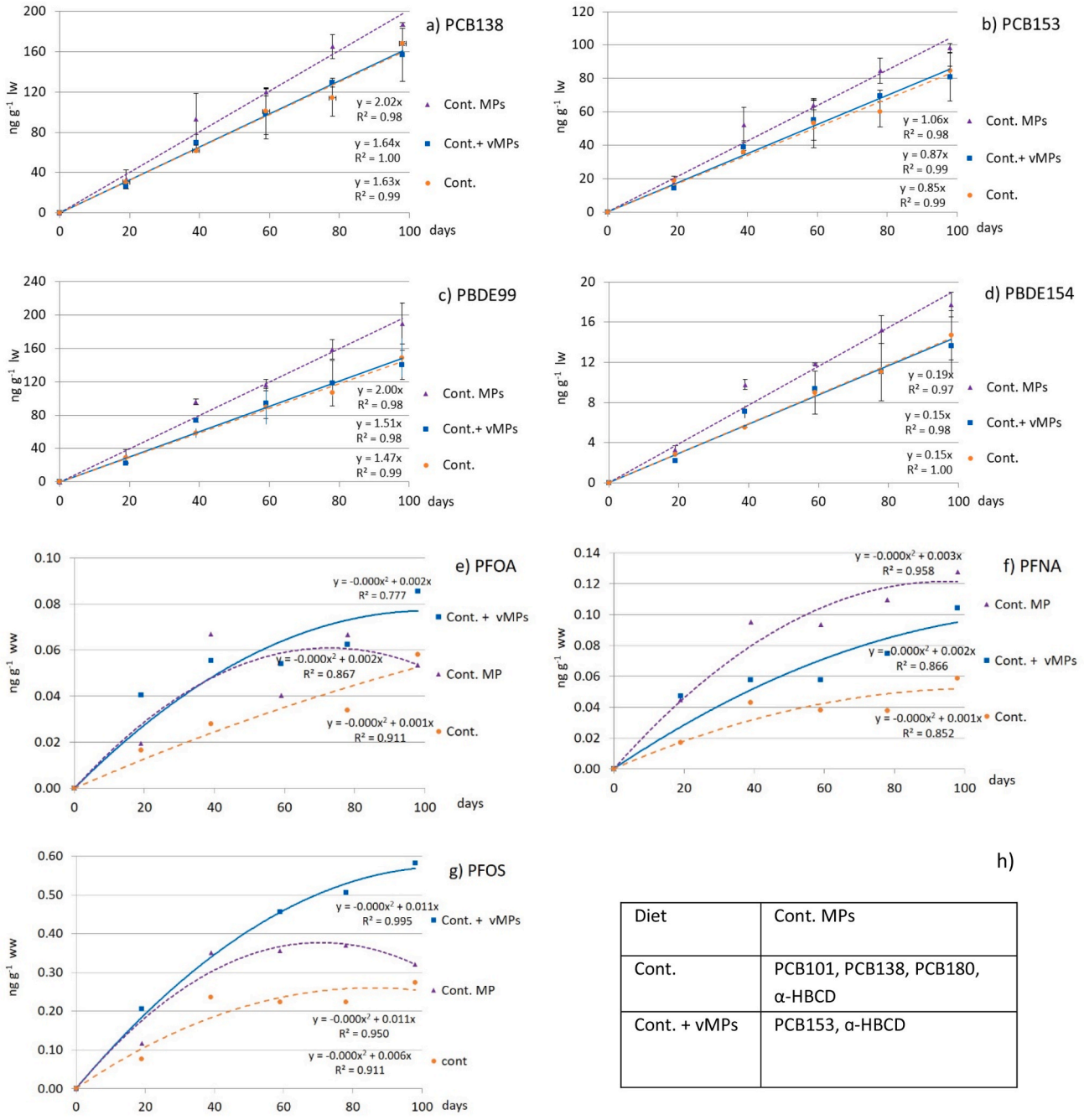
The contaminated feed concentrations of ɑ-HBCD, PBDE congeners, and PCB congeners ranged from 8 to 148 ng g^{-1} lipid weight (lw) and PFAS ranged from 7 to 11 ng g^{-1} wet weight (ww)(Fig. S2). Average concentrations in contaminated feeds are shown in Table 1, and the concentrations in the two control and three spiked feeds are in Fig. S2. The control feeds had background contamination on ~10% of the contaminated feeds (except for PCB28, PCB180 ~ 20%, PCB138 ~ 3%, and PFAS ~ 1%). PFAS binds to proteins and is therefore calculated on a wet-weight basis. The PFAS concentrations in the *Cont. MPs*' diets were lower compared to the two other diets, relatively for PFOA 95%, PFNA 85%, and PFOS 65%, which may be due to differences in the sorption efficiency of MPs. PFOS is not as strongly sorbed due to the more hydrophilic and larger sulfonic acid group (Bakir et al., 2014; Cormier et al., 2022).

3.3. PCB, PBDE, and HBCD accumulations, eliminations, and estimated k_{el}, $t½$, and α

The 96 days accumulation regression lines of C_{fish} (ng g^{-1} lw) reveal the same overall pattern for the 11 lipophilic contaminants (Fig. 3a–d, Fig. S3a-k). The accumulation regression lines for the *Cont. MPs* diets are higher than those of the *Cont.+vMPs* and *Cont.* diets. Furthermore, the correlation of the accumulations of the *Cont. MPs* diet improved by quadratic regression (see Fig. S3) and showed an increasing difference to the other diets throughout most of the accumulation period, although decreasing towards the end. The correlation of the accumulation regressions is strong, with R^2 of 0.96–1.00 (mean 0.99, n = 33). However, linear regressions are shown in Fig. 3a–d and used to test for statistical differences in accumulation slopes between the three diets. They demonstrate significantly higher accumulation (increased slope) for several lipophilic POPs of the *Cont. MPs* diet where the contaminants were sorbed to MPs compared to the *Cont.* diet and the *Cont.+vMPs* diet (Fig. 3, Fig. S3).

Contaminant accumulation depends on the elimination coefficient (k_{el}). This value was calculated by determining the slope of the natural logarithm of C_{fish} (ln $C_{fish\ adj}$) *vs* time for the elimination period as per Eq. (4). The results are displayed in Fig. 4a–d, Fig. S4a-k. The k_{el} regression lines for the salmon fed the *Cont. MPs* diet were significantly lower than those of the *Cont.* diet for 7 of 11 lipophilic POPs. In addition, for 3 PCBs k_{el} regression lines of the *Cont.+vMPs* diet was significantly lower compared to that of the *Cont.* diet (Table 1).

The toxicokinetic parameters of lipophilic contaminants relate to their octanol/water partition coefficient (log K_{ow}) and concentration level (Fisk et al., 1998). The contaminant's log K_{ow} and the toxicokinetic behaviour were correlated (Fig. 5 a-d), and distinct toxicokinetic changes were observed when the salmon were exposed to MPs in



Diet	Cont. MPs
Cont.	PCB101, PCB138, PCB180, α-HBCD
Cont. + vMPs	PCB153, α-HBCD

Fig. 3. a–g: Growth factor (GF) adjusted concentrations (C_{fish}, ng g^{-1} lw) of a)PCB138, b)PCB153, c)PBDE99, d)PBDE154, e)PFOA, f)PFNA, g)PFOS in salmon fillets during accumulation, and h) List of POPs with significantly different linear slopes between diets. Error bars show SD, Figs. S3a–k: show all PCBs, PBDEs, HBCD.

combination with contaminants. Multiple comparisons by two-way ANOVA tests of k_{el} for individual PCBs and PBDEs, respectively, showed highly significant differences between diets and between contaminants (Fig. 5a and b).

The assessment of the toxicokinetic behaviour of PCBs without the presence of MPs demonstrated that k_{el} increased with log K_{ow} up to log K_{ow} around seven (PCB138), then decreased (Fig. 5a). Fisk et al. (1998) studied the accumulation and elimination of PCBs in rainbow trout and found relationships between log K_{ow} and contaminant bioaccumulation and elimination. However, they also found that k_{el} was concentration-dependent and decreased more than twofold within a 5-fold concentration reduction. In the present study, the feed contaminant concentrations were maintained constant between treatments but varied between contaminants, e.g. PCB congeners reflecting environmental concentration ranges. The concentration increased from PCB28 at 34 ng g^{-1} lw to PCB138 at 147 ng g^{-1} lw and then decreased to PCB180 at 7.8 ng g^{-1} lw. Therefore, the concentration decrease may explain the k_{el} decline for log $K_{ow} > 7$. Since half-life is conversely correlated to k_{el}, PCB138 with the highest k_{el} had the lowest $t½$ at 75 days (Table 1). The PCB k_{el} range from 0.42 to $0.92{\cdot}10^{-2}$ d^{-1} is comparable to Fisk et al.'s (1998). For the PBDEs, k_{el} increased with log K_{ow} from PBDE47 at $0.37{\cdot}10^{-2}$ d^{-1} to PBDE153 at $0.57{\cdot}10^{-2}$ d^{-1} and then deceased to BDE154 at $0.51{\cdot}10^{-2}$ d^{-1}.

The PCB elimination (k_{el}) from fillets of salmon fed with the *Cont.* diet was significantly faster than those from the two contaminated diets with MPs. The k_{el} from fillets of fish fed the *Cont. MPs* diet with contaminants absorbed to MP increased linearly with log K_{ow}, and the results were lower than those of the *Cont.+vMPs* diet except for PCB180

(Fig. 5a).

The assimilation efficiency (α), *i.e.*, the amount from the feed that is transferred to the salmon fillets, depends on the absorption, distribution, metabolism, and excretion of the contaminant. The plots of α *vs* the log K_{ow} for PCBs and PBDEs (Fig. 5c and d) demonstrated linear increases in α with higher log K_{ow} within the contaminant classes and diets with R^2 of 0.86–0.99 (mean 0.94, n = 6). For differences in α, two-way ANOVA tests were also highly significant between diets and even higher between PCB and PBDE congeners, respectively. The higher the halogenation or log K_{ow}, the more contaminant is transferred from feed to the salmon fillets. Regarding PBDE transfer, Isosaari et al. (2005) reported that Atlantic salmon fillets accumulated 42–59% of ∑PBDEs compared to 51–71% of individual congeners for this study's *Cont.* diet. Berntssen et al., 2011a studied PCB and PBDE carryover in Atlantic salmon and found at 38 ng g^{-1} ww ∑PCB6 in feed 42–50% carryover of congeners, while in this study we found 54–71%. Their carryover of four PBDE congeners ranged from 34 to 49%. In the presence of 3.1 ng g^{-1} ww α-HBCD in feed, they found 35% carryover to the fillet (Berntssen et al., 2011b), while in this study, at a feed concentration of ~5.6 ng g^{-1} w/w 50% was transferred.

PBDE k_{el}'s for the fish fed the *Cont. MP* diet were lower; however, in fish fed the *Cont.* or *Cont.+vMPs* diets, elimination rates were comparable (Fig. 5b). The assimilation efficiency (α) as a function of log K_{ow} is higher for all PCBs and PBDEs for salmon fed the *Cont. MPs* diet compared to the other two diets (Fig. 5 c,d). A previous study addressing the role of MPs in the transfer of PBDE contaminants reported that MPs addition reduced PBDE assimilation in the tissue of a marine amphipod (*Allorchestes compressa*). In addition, MPs caused a greater proportional uptake of higher-brominated congeners such as PBDE-154 and -153 compared to PBDE 47 (Chua et al., 2014). In the present study, higher assimilation efficiencies and a significantly higher slope of assimilation efficiencies *vs* log K_{ow} were also found for the *Cont. MPs* diet ($p = 0.029$), indicating a proportionally higher uptake of higher-brominated congeners compared to the *Cont.* diet. This is explained by decreased PBDE diffusion coefficients with increasing molecular weight (MW) (Narváez Valderrama et al., 2016). The lower MW PBDEs diffuse further into the MPs while the higher MW PBDEs diffuse less; and for the *Cont. MPs* diet with PBDEs absorbed into MPs, this would relatively increase the desorption and uptake of the higher MW PBDEs. In this study, diffusion/absorption into the MPs occurs from the time of feed preparation to experiment initiation (~4 months), while the diffusion/desorption out of MPs into the intestinal fluid occurs within hours.

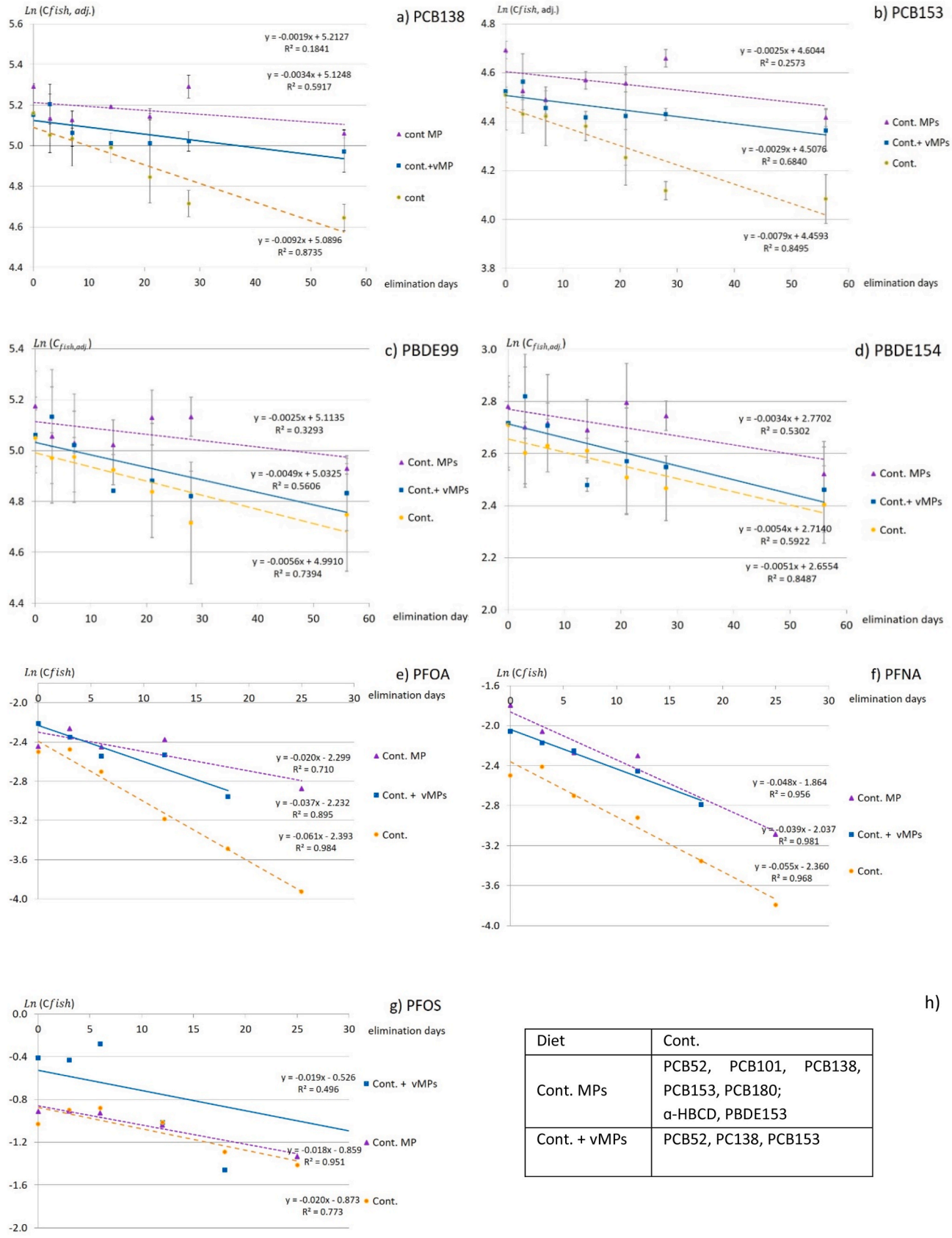
3.4. PFAS accumulations, eliminations, and estimated k_{el}, $t_{½}$, and α

The accumulation regression lines for PFAS are fitted with quadratic curves and showed markedly lower bioaccumulation and different shapes compared to those of the lipophilic contaminants (Fig. 3). The PFAA concentrations measured in the salmon fillets were low compared to the PFAA concentrations in the contaminated diets of 7–12 ng g^{-1} ww (Fig. S2). The maximum C_{fish} in fillets were PFOA ~0.08 ng g^{-1}, PFNA ~0.12 ng g^{-1}, and PFOS ~0.6 ng g^{-1}ww. The distribution of PFAA to muscle proteins and neutral lipids is low because it binds to other proteins such as serum albumin and liver fatty acid binding protein [L-FABP]) resulting in accumulation in tissues other than the fillet (Vidal et al., 2019a). However, the chemical partitioning of PFAS increased with PFAA chain length and was greater for PFOS, which contains the sulphonic acid group (C8-PFSA) compared to PFOA, which contains the carboxylic acid group (C8-PFCA). During most of the accumulation phase, the PFAS concentrations were lowest in the fish fed the *Cont.* diet. After day 80 of accumulation, PFOA and PFOS levels in fish fed with the *Cont. MPs* diet decreased rather than continued to increase, resulting in end accumulation in levels comparable to those fed with the *Cont.* diet. For PFNA, the quadratic curve for the *Cont. MPs* diet increased less towards the end of accumulation, although it was still higher than for the *Cont.* and *Cont.+vMPs* diets. The elimination regressions of the carboxylic acids PFOA and PFNA from the fillets of the salmon fed the *Cont.* diets were 0.061 d^{-1} ($R^2 = 0.98$) and 0.055 d^{-1} ($R^2 = 0.97$), and $t_{½}$ were 11.4 d and 12.6 d, respectively (Table 1). Furthermore, k_{el} of PFOA (0.037 d^{-1}, $R^2 = 0.90$) and PFNA (0.039 d^{-1}, $R^2 = 0.95$) from fillets of salmon fed the *Cont.+vMPs* diet were comparable, as were the longer $t_{½}$ at 19 d and 18 d. In the *Cont. MP* diet, the k_{el} of PFNA lies between the other two diets (0.048 d^{-1}, $R^2 = 0.96$), while the elimination of PFOA was slower but with a weaker correlation (k_{el} 0.020 d^{-1}, $R^2 = 0.71$). Regarding PFOS, the *Cont.* and *Cont. MPs* diets have comparable elimination slopes (k_{el} 0.018–0.020 d^{-1}). The *Cont.+vMPs* diet has the same k_{el}, but the value is uncertain due to a weak correlation. The PFOS half-lives were 35–39 d. Bile/fecal elimination is the most important route for PFOS, while PFCAs up to and including PFOA also have a renal elimination driven by an active protein-binding mechanism (Sun et al., 2022). The mentioned differences in toxicokinetics explain the longer $t_{½}$ of PFOS, which is reflected in the higher concentrations in the studied salmon and generally more PFOS than PFCA in wild and farmed fish for human consumption (Domingo and Nadal, 2017). Falk et al. (2015) found $t_{½}$ of PFOA 7.0 d, PFNA 9.5 d, and PFOS 8.4 d in rainbow trout fillets, and PFOA 9.0 d, PFNA 11.6 d, and PFOS 20.4 d in rainbow trout liver. Their feed concentrations were 172–345 ng g^{-1}, >20-fold higher than in this study, which, in addition to the species difference, may affect toxicokinetics at lower concentrations, increasing $t_{½}$. Lower temperatures also slow down toxicokinetics, increasing $t_{½}$ (Vidal et al., 2019b). A lower flow-through seawater temperature at 8–10 °C in this study during the PFAS elimination phase compared to 14–16 °C in Falk et al. (2015) may also have contributed to the higher $t_{½}$. Assimilation efficiencies to salmon fillets were expectedly low for PFCA 2–11% and PFOS 9–11% due to higher bioaccumulation in the liver, blood, kidney, and brain than in muscle tissue (Vidal et al., 2019a). They found liver/muscle ratios of ~14 for PFOS in rainbow trout, and Rainieri et al. (2018) found liver/muscle ratios of 14 for PFOS, 8 for PFNA, and 6 for PFOA in zebrafish.

3.5. Influence of MPs on the toxicokinetic behaviour of halogenated POPs

The distribution of the halogenated contaminants in organisms differs from that of MPs. MPs of 125–250 μm are expected to pass through the digestive system. In contrast, the proportion of bioaccessible contaminants passes across the intestinal epithelium to the liver, where it is distributed in the salmon. However, PFAS themselves may damage the phospholipid bilayer membranes of the epithelium (Fitzgerald et al., 2018; Liu et al., 2022), promoting the entry of some smaller-size MPs into the circulatory system, causing hepatotoxicity. Conversely, MPs can also damage the intestine, enabling more access to contaminants. Pedà et al. (2016) found histological alterations, especially in the distal intestine of European sea bass exposed to feed with 0.1% MPs. If, in this study, the contaminants that were sorbed to MPs were desorbed more distally, where the intestinal epithelium could be fragile, it may have increased the uptake of contaminants from the *Cont. MPs* diet.

For the contaminants to be bioaccessible from the *Cont. MPs*, they must be desorbed from the PE-MPs. The dual hydrophobic and hydrophilic properties of PFAS imply that they are mainly adsorbed to MPs (Wang et al., 2015), while the lipophilic contaminants are absorbed. Thus, the supply of surfactants in the GI system is crucial for the effective desorption of contaminants from MPs (Bakir et al., 2014; Teuten et al., 2007), e.g., *in vitro*, studies have shown 70–80% desorption of PFOS from PE-MP into an artificial gut fluid (Cormier et al., 2022). In support of an *in vivo* influence of MPs on physiological conditions, Lu et al. (2018) showed that 50 μm MP given to mice in environmentally high but realistic concentrations affected the gut microbiome, induced hepatic lipid disorder, reduced liver triglycerides, and released cholesterol and related lipid surfactants into the GI system. Such an increase in gut surfactants could have enhanced the desorption of contaminants from the *Cont. MPs* diet and hypothetical disturbances in liver function could have affected sorption interactions and delayed the elimination of



h)

Diet	Cont.
Cont. MPs	PCB52, PCB101, PCB138, PCB153, PCB180; α-HBCD, PBDE153
Cont. + vMPs	PCB52, PC138, PCB153

Fig. 4. a–g: The fitted regression lines from ln($C_{fish,\ adjusted}$) *vs* elimination time for contaminants in fillets of salmon fed the three diets and h) List of POPs with significantly different linear slopes between diets (Figs. S4a–k show all lipophilic contaminants).

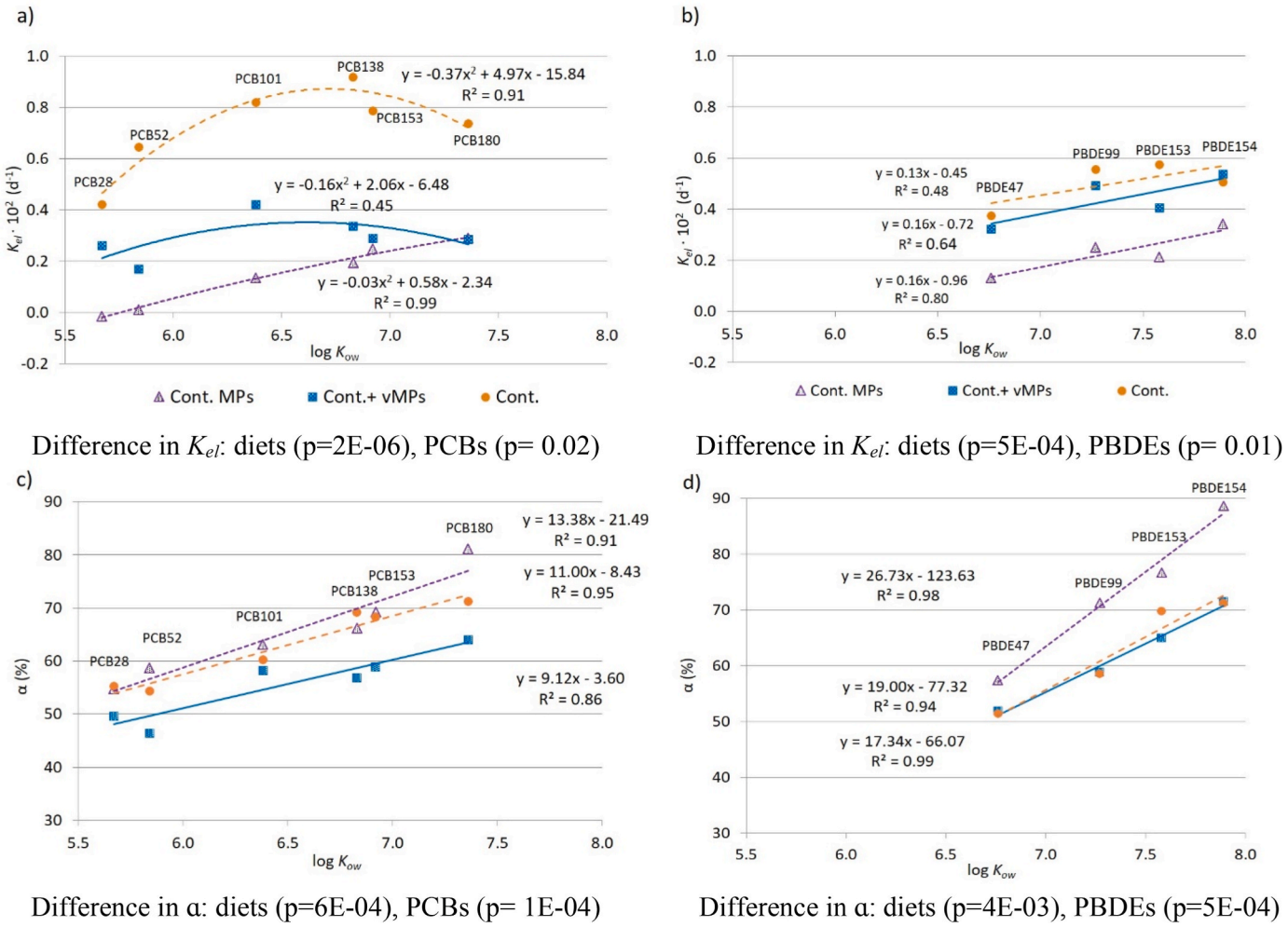


Fig. 5. Elimination coefficients and assimilation efficiencies α of a,c) PCBs and b,d) PBDEs in relation to log K_{ow} in fillets of salmon fed the three contaminated diets. Significant differences between diets and contaminants tested by two-way ANOVA are shown below the figures (PFAS results are shown in Fig. S5).

contaminants.

The following interpretations of the impact of MPs on contaminant toxicokinetics are based on parallels drawn to other studies where fish were fed similar dietary formulations of MPs in combinations with halogenated POPs. A sea bass study showed down-regulation of liver gene expression of *cyp1a1* and *gstα* midway through the bioaccumulation phase of a *Cont. MPs* diet with contaminants sorbed to MPs, indicating an inhibition of the detoxification process (Granby et al., 2018). Genes *cyp1a1* and *gstα* encode phase I (cytochrome-P450-1A1) and phase II (glutathione S transferase α) detoxification enzymes. These inhibitions may explain the higher bioaccumulation of the *Cont. MPs* diet compared to the *Cont.+vMPs* diet and the *Cont.* diet without MPs seen in both the seabass study and this salmon study. Neither the sea bass *Cont.+vMPs* diet nor the *Cont.* diet showed down-regulation of the same liver enzymes.

On the contrary, at the end of the elimination phase, these diets showed an upregulation of the gene expression of detoxification enzymes to adapt to a higher level of contamination. Towards the end of the elimination phase, sea bass from the two diets with combinations of contaminants and MPs also had significantly reduced hepato-somatic indices (HSI) compared to diets without MPs (Granby et al., 2018). Reduction in liver weight and a likely accompanying disturbance in the liver metabolism may have distributed lipid compounds and lipophilic contaminants from the liver to the fish fillets, thereby delaying contaminant eliminations for the MP-containing diets. Pedà et al. (2022) also observed a significant reduction in HSI in sea bass after 90 days of exposure to 0.1% PVC-MP, and Rochman et al., 2013 found severe glycogen depletion and histological effects on the liver of Japanese medaka (*Oryzias latipes*) exposed to either virgin or contaminated PE-MP.

Although PFAS are not themselves subject to metabolism, PFAS can alter the regulation of genes related to the xenobiotic metabolism of other compounds (Lee et al., 2020). Both up-and down-regulation of genes can occur. This cannot explain the PFAS accumulation and elimination curves, e.g., the increased accumulations for the *Cont.+ vMPs* and *Cont. MPs* diets (Figs. 3 and 4 e-g). However, the increased accumulations may be partially caused by an increased release of gastric surfactants in the presence of MPs as shown in a study by Lu et al. (2018), which could facilitate PFAS uptake.

4. Conclusions and perspectives

A comprehensive salmon trial was performed to gain insight into the toxicokinetics of 14 halogenated POPs when co-occurring with PE-MPs. The lipid-soluble PCBs and PBDEs showed high assimilation efficiencies (α). Together with the elimination coefficients (k_{el}), these toxicokinetic parameters are correlated with contaminant hydrophobicity (log K_{ow}) within the diets and halogen classes. Moreover, multiple comparisons of α or k_{el} obtained for three contaminated diets demonstrated highly significant differences between diets and between congeners within the halogen classes. The salmons fed the *Cont. MPs* diet with contaminants sorbed to MPs showed higher fillet accumulation and lower fillet

elimination compared to the contaminated diets without or with vMPs; thus, binding to MP did not inhibit but enhanced the contaminant bioavailability and reduced its elimination. For the *Cont. MPs* diet the assimilation efficiency (α) was proportionally greater for the higher MW PBDEs compared to the other diets ($p = 0.029$), which to our knowledge has not previously been demonstrated in fish species. PFAS was also affected by the co-occurrence of MPs; the MP-containing diets increased PFOA and PFNA accumulation and decreased their elimination. PFOA and PFNA generally had short half-lives and low assimilation efficiencies, i.e., transfer from feed to fillet, compared to PFOS, but especially compared to the lipophilic halogenated POPs.

The reasons for the changes in the toxicokinetic behaviour of POPs when co-occurring with MPs are not evident. The causes may be increased transfer from *Cont. MPs* due to fragile intestinal epithelium; disturbed liver homeostasis, releasing fatty acids and cholesterol, acting as gut surfactants, increasing absorption of contaminants from the intestine. Furthermore, the liver impact may affect contaminant metabolism or distribute more contaminants from the liver to the salmon fillet.

Although MPs and POPs are ubiquitous in the environment, there are few toxicokinetic studies on the co-exposure to MPs and POPs. Studies such as the present one widen our knowledge about MPs as carriers of contaminants in fish, which is highly relevant to revisit our seafood safety evaluations. Aquatic wildlife is affected by MPs, but MPs have also been found in, e.g., human liver and cardiovascular plaque, which were related to severe health disorders (Horvatits et al., 2022; Marfella et al., 2024). Studies on other vertebrates, such as fish, may contribute to understanding the mechanisms of potential interactions and implications of coexisting MPs and chemical hazards.

CRediT authorship contribution statement

Kit Granby: Writing – original draft, Investigation, Formal analysis, Conceptualization. **Bina Bhattarai:** Writing – review & editing, Writing – original draft. **Ninna Johannsen:** Writing – review & editing, Investigation, Formal analysis. **Michiel J.J. Kotterman:** Writing – review & editing, Investigation, Conceptualization. **Jens J. Sloth:** Writing – review & editing, Investigation. **Tommy Licht Cederberg:** Writing – review & editing, Formal analysis. **António Marques:** Writing – review & editing, Funding acquisition, Conceptualization. **Bodil Katrine Larsen:** Writing – original draft, Investigation, Formal analysis, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envpol.2024.124421.

References

Abihssira-García, I.S., Kögel, T., Gomiero, A., Kristensen, T., Krogstad, M., Olsvik, P.A., 2022. Distinct polymer-dependent sorption of persistent pollutants associated with Atlantic salmon farming to microplastics. Mar. Pollut. Bull. 180 https://doi.org/10.1016/j.marpolbul.2022.113794.

Abihssira-García, I.S., Park, Y., Kiron, V., Olsvik, P.A., 2020. Fluorescent microplastic uptake by immune cells of atlantic salmon (Salmo salar L.). Front. Environ. Sci. 8 https://doi.org/10.3389/fenvs.2020.560206.

Bakir, A., O'Connor, I.A., Rowland, S.J., Hendriks, A.J., Thompson, R.C., 2016. Relative importance of microplastics as a pathway for the transfer of hydrophobic organic chemicals to marine life. Environ. Pollut. 219, 56–65. https://doi.org/10.1016/j.envpol.2016.09.046.

Bakir, A., Rowland, S.J., Thompson, R.C., 2014. Enhanced desorption of persistent organic pollutants from microplastics under simulated physiological conditions. Environ. Pollut. 185, 16–23. https://doi.org/10.1016/j.envpol.2013.10.007.

Berntssen Marc, H.G., Maage, A., Julshamn, K., Oeye, B.E., Lundebye, A.K., 2011a. Carry-over of dietary organochlorine pesticides, PCDD/Fs, PCBs, and brominated flame retardants to Atlantic salmon (Salmo salar L.) fillets. Chemosphere 83, 95–103. https://doi.org/10.1016/j.chemosphere.2011.01.017.

Berntssen, M.H.G., Valdersnes, S., Rosenlund, G., Torstensen, B.E., Zeilmaker, M.J., van Eijkeren, J.C.H., 2011b. Toxicokinetics and carry-over model of α-hexabromocyclododecane (HBCD) from feed to consumption-sized atlantic salmon (Salmo salar). Food Addit. Contam. 28, 1274–1286. https://doi.org/10.1080/19440049.2011.587029.

Braekevelt, E., Tittlemier, S.A., Tomy, G.T., 2003. Direct measurement of octanol-water partition coefficients of some environmentally relevant brominated diphenyl ether congeners. Chemosphere 51, 563–567. https://doi.org/10.1016/S0045-6535(02)00841-X.

Chua, E.M., Shimeta, J., Nugegoda, D., Morrison, P.D., Clarke, B.O., 2014. Assimilation of polybrominated diphenyl ethers from microplastics by the marine amphipod, allorchestes compressa. Environ. Sci. Technol. 48, 8127–8134. https://doi.org/10.1021/es405717z.

Cormier B, Borchet F, Kärrman A, Szot M, Yeung LWY, Keiter SH. Sorption and desorption kinetics of PFOS to pristine microplastic n.d. https://doi.org/10.1007/s11356-021-15923-x/Published.

Domingo, J.L., Nadal, M., 2017. Per- and polyfluoroalkyl substances (PFASs) in food and human dietary intake: a review of the recent scientific literature. J. Agric. Food Chem. 65, 533–543. https://doi.org/10.1021/acs.jafc.6b04683.

Falk, S., Failing, K., Georgii, S., Brunn, H., Stahl, T., 2015. Tissue specific uptake and elimination of perfluoroalkyl acids (PFAAs) in adult rainbow trout (Oncorhynchus mykiss) after dietary exposure. Chemosphere 129, 150–156. https://doi.org/10.1016/j.chemosphere.2014.06.061.

Fisk, A.T., Norstrom, R.J., Cymbalisty, C.D., Muir, D.C.G., 1998. Dietary accumulation and depuration of hydrophobic organochlorines: bioaccumulation parameters and their relationship with the octanol/water partition coefficient. Environ. Toxicol. Chem. 17, 951–961. https://doi.org/10.1002/etc.5620170526.

Fitzgerald, N.J.M., Wargenau, A., Sorenson, C., Pedersen, J., Tufenkji, N., Novak, P.J., et al., 2018. Partitioning and accumulation of perfluoroalkyl substances in model lipid bilayers and bacteria. Environ. Sci. Technol. 52, 10433–10440. https://doi.org/10.1021/acs.est.8b02912.

Granby, K., Rainieri, S., Rasmussen, R.R., Kotterman, M.J.J., Sloth, J.J., Cederberg, T.L., et al., 2018. The influence of microplastics and halogenated contaminants in feed on toxicokinetics and gene expression in European seabass (Dicentrarchus labrax). Environ. Res. 164, 430–443. https://doi.org/10.1016/j.envres.2018.02.035.

ter Halle, A., Ladirat, L., Martignac, M., Mingotaud, A.F., Boyron, O., Perez, E., 2017. To what extent are microplastics from the open ocean weathered? Environ. Pollut. 227, 167–443. https://doi.org/10.1016/j.envpol.2017.04.051.

Hartmann, N.B., Hüffer, T., Thompson, R.C., Hassellöv, M., Verschoor, A., Daugaard, A.E., et al., 2019. Are we speaking the same language? Recommendations for a definition and categorization Framework for plastic debris. Environ. Sci. Technol. 53, 1039–1047. https://doi.org/10.1021/acs.est.8b05297.

Hartmann, N.B., Rist, S., Bodin, J., Jensen, L.H.S., Schmidt, S.N., Mayer, P., et al., 2017. Microplastics as vectors for environmental contaminants: exploring sorption, desorption, and transfer to biota. Integrated Environ. Assess. Manag. 13, 488–493. https://doi.org/10.1002/ieam.1904.

Horvatits, T., Tamminga, M., Liu, B., Sebode, M., Carambia, A., Fischer, L., et al., 2022. Microplastics detected in cirrhotic liver tissue. EBioMedicine 82, 104147. https://doi.org/10.1016/j.

Isosaari, P., Lundebye, A.K., Ritchie, G., Lie, Kiviranta H., Vartiainen, T., 2005. Dietary accumulation efficiencies and biotransformation of polybrominated diphenyl ethers in farmed Atlantic salmon (Salmo salar). Food Addit. Contam. 22, 829–837. https://doi.org/10.1080/02652030500195205.

Liu, Y., Shi, Q., Liu, X., Wang, L., He, Y., Tang, J., 2022. Perfluorooctane sulfonate (PFOS) enhanced polystyrene particles uptake by human colon adenocarcinoma Caco-2 cells. Sci. Total Environ. 848 https://doi.org/10.1016/j.scitotenv.2022.157640.

Lohmann, R., 2012. Critical review of low-density polyethylene's partitioning and diffusion coefficients for trace organic contaminants and implications for its use as a passive sampler. Environ. Sci. Technol. https://doi.org/10.1021/es202702y.

Lu, L., Wan, Z., Luo, T., Fu, Z., Jin, Y., 2018. Polystyrene microplastics induce gut microbiota dysbiosis and hepatic lipid metabolism disorder in mice. Sci. Total Environ. 631–632, 449–458. https://doi.org/10.1016/j.scitotenv.2018.03.051.

Lu, Y., Zhang, Y., Deng, Y., Jiang, W., Zhao, Y., Geng, J., et al., 2016. Uptake and accumulation of polystyrene microplastics in zebrafish (Danio rerio) and toxic effects

in liver. Environ. Sci. Technol. 50, 4054–4060. https://doi.org/10.1021/acs.est.6b00183.
Marfella, R., Prattichizzo, F., Sardu, C., Fulgenzi, G., Graciotti, L., Spadoni, T., et al., 2024. Microplastics and nanoplastics in atheromas and cardiovascular events. N. Engl. J. Med. 390, 900–910. https://doi.org/10.1056/nejmoa2309822.
Mato, Y., Isobe, T., Takada, H., Kanehiro, H., Ohtake, C., Kaminuma, T., 2001. Plastic resin pellets as a transport medium for toxic chemicals in the marine environment. Environ. Sci. Technol. 35, 318–324. https://doi.org/10.1021/es0010498.
Narváez Valderrama, J.F., Baek, K., Molina, F.J., Allan, I.J., 2016. Implications of observed PBDE diffusion coefficients in low density polyethylene and silicone rubber. Environ. Sci. J. Integr. Environ. Res.: Process. Impacts 18, 87–94. https://doi.org/10.1039/c5em00507h.
O'Connor, I.A., Golsteijn, L., Hendriks, A.J., 2016. Review of the partitioning of chemicals into different plastics: consequences for the risk assessment of marine plastic debris. Mar. Pollut. Bull. 113, 17–24. https://doi.org/10.1016/j.marpolbul.2016.07.021.
Park, M., Wu, S., Lopez, I.J., Chang, J.Y., Karanfil, T., Snyder, S.A., 2020. Adsorption of perfluoroalkyl substances (PFAS) in groundwater by granular activated carbons: roles of hydrophobicity of PFAS and carbon characteristics. Water Res. 170 https://doi.org/10.1016/j.watres.2019.115364.
Paternoster, R., Brame, R., Mazerolle, P., Piquero, A., 1998. Using the correct statistical test for the equality of regression coefficients. Criminology 36, 859–866. https://doi.org/10.1111/j.1745-9125.1998.tb01268.x.
Pedà, C., Caccamo, L., Fossi, M.C., Gai, F., Andaloro, F., Genovese, L., et al., 2016. Intestinal alterations in European sea bass Dicentrarchus labrax (Linnaeus, 1758) exposed to microplastics: preliminary results. Environ. Pollut. 212, 251–256. https://doi.org/10.1016/j.envpol.2016.01.083.
Pedà, C., Romeo, T., Panti, C., Caliani, I., Casini, S., Marsili, L., et al., 2022. Integrated biomarker responses in European seabass Dicentrarchus labrax (Linnaeus, 1758) chronically exposed to PVC microplastics. J. Hazard Mater. 438 https://doi.org/10.1016/j.jhazmat.2022.129488.
Rainieri, S., Conlledo, N., Larsen, B.K., Granby, K., Barranco, A., 2018. Combined effects of microplastics and chemical contaminants on the organ toxicity of zebrafish (Danio rerio). Environ. Res. 162, 135–143. https://doi.org/10.1016/j.envres.2017.12.019.
Rochman, C.M., Hoh, E., Hentschel, B.T., Kaye, S., 2013. Long-term field measurement of sorption of organic contaminants to five types of plastic pellets: implications for plastic marine debris. Environ. Sci. Technol. https://doi.org/10.1021/es303700s.
Sun, J.M., Kelly, B.C., Gobas, F.A.P.C., Sunderland, E.M., 2022. A food web bioaccumulation model for the accumulation of per- and polyfluoroalkyl substances (PFAS) in fish: how important is renal elimination? Environ Sci Process Impacts 24, 1152–1164. https://doi.org/10.1039/d2em00047d.
Teuten, E.L., Rowland, S.J., Galloway, T.S., Thompson, R.C., 2007. Potential for plastics to transport hydrophobic contaminants. Environ. Sci. Technol. 41, 7759–7764. https://doi.org/10.1021/es071737s.
Vidal, A., Babut, M., Garric, J., Beaudouin, R., 2019a. Elucidating the fate of perfluorooctanoate sulfonate using a rainbow trout (Oncorhynchus mykiss) physiologically-based toxicokinetic model. Sci. Total Environ. 691, 1297–1309. https://doi.org/10.1016/j.scitotenv.2019.07.105.
Vidal, A., Lafay, F., Daniele, G., Vulliet, E., Rochard, E., Garric, J., et al., 2019b. Does water temperature influence the distribution and elimination of perfluorinated substances in rainbow trout (Oncorhynchus mykiss)? Environ. Sci. Pollut. Control Ser. 26, 16355–16365. https://doi.org/10.1007/s11356-019-05029-w.
Wang, F., Shih, K.M., Li, X.Y., 2015. The partition behavior of perfluorooctanesulfonate (PFOS) and perfluorooctanesulfonamide (FOSA) on microplastics. Chemosphere 119, 841–847. https://doi.org/10.1016/j.chemosphere.2014.08.047.
Wardrop, P., Shimeta, J., Nugegoda, D., Morrison, P.D., Miranda, A., Tang, M., et al., 2016. Chemical pollutants sorbed to ingested microbeads from personal care products accumulate in fish. Environ. Sci. Technol. 50, 4037–4044. https://doi.org/10.1021/acs.est.5b06280.
Zaiontz, C., 2023. Real statistics resource pack software, Release 8.9. www.real-statistics.com/regression/hypothesis-testing-significance-regression-line-slope/comparing-slopes-two-independent-samples/. (Accessed 21 May 2024).