Environmental Science & Technology

pubs.acs.org/est

Article

Exposure to Legacy Per- and Polyfluoroalkyl Substances from Diet and Drinking Water in California Adults, 2018−2020

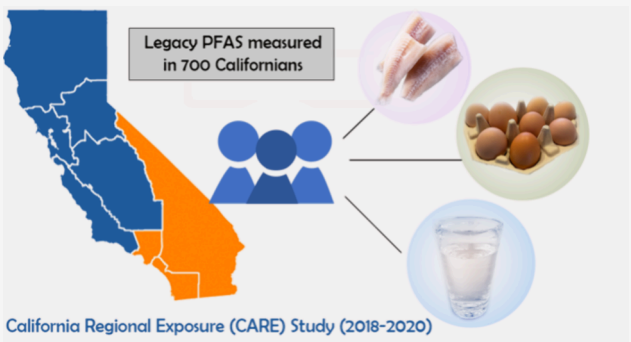
Emily H. Pennoyer,* Toki Fillman, Wendy Heiger-Bernays, Kathleen R. Attfield, Hyoung-Gee Baek, Songmei Gao, Sabrina Smith, June-Soo Park, Nerissa Wu, and Thomas F. Webster

Cite This: *Environ. Sci. Technol.* 2025, 59, 9896−9906

Read Online

ACCESS | Metrics & More | Article Recommendations | Supporting Information

ABSTRACT: People are exposed to per- and polyfluoroalkyl substances (PFAS) through multiple sources, with diet historically considered a major source in general populations. This study characterized legacy PFAS in serum from 700 California adults and examined contributions from diet and drinking water. We applied robust regression to estimate associations between nontransformed serum PFAS concentrations, self-reported food consumption, and drinking water PFAS concentrations measured under the USEPA's third Unregulated Contaminant Monitoring Rule (2013−2015). Detectable drinking water concentrations were associated with increased serum perfluorooctanoic acid (PFOA) (0.26 ng/mL; 95% CI: 0.077, 0.43), perfluorohexanesulfonic acid (PFHxS) (0.64 ng/mL; 95% CI: 0.058, 1.23), and perfluorooctanesulfonic acid (PFOS) (0.39 ng/mL; 95% CI: −0.76, 0.86). Seafood consumption was associated with increased perfluorononanoic acid (PFNA) (0.013 ng/mL; 95% CI: 0.0058, 0.021), perfluorodecanoic acid (PFDeA) (0.0059 ng/mL; 95% CI: 0.0026, 0.0092), and perfluoroundecanoic acid (PFUnDA) (0.010 ng/mL; 95% CI: 0.0054, 0.015), while eggs were associated with increased PFDeA (0.0035 ng/mL; 95% CI: 0.00010, 0.0069) and PFNA (0.0073 ng/mL; 95% CI: 0.00017, 0.014). Findings could indicate that dietary contributions may be less than those in earlier studies conducted in other populations, possibly due to shifts in PFAS production over the past 20 years, and that drinking water remains an important source of exposure to PFOA and PFHxS in this population.



KEYWORDS: *PFAS, diet, drinking water, biomonitoring, exposure*

INTRODUCTION

Per- and polyfluoroalkyl substances (PFAS) refer to a wide class of synthetic fluorinated chemicals developed for a range of uses. Since various definitions classify the PFAS chemical class differently,[1] here, we refer to "legacy" PFAS, defined as longer-chain perfluoroalkyl acids (carboxylates with ≥7 perfluorinated carbon atoms and sulfonates with ≥6 perfluorinated carbon atoms) and their precursors historically used for commercial and industrial uses, including aqueous film-forming foam (AFFF), processing aids in manufacturing, consumer products, and food contact materials.[2−4] Many legacy PFAS, including perfluorooctanesulfonic acid (PFOS) and perfluorooctanoic acid (PFOA), have been phased out of production in the U.S. since the early 2000s, but the persistent chemicals remain ubiquitous in the environment and are still detected in humans decades later.[5] Countries in Europe have similarly phased out production of some PFAS, but there is ongoing manufacturing of PFOA in other parts of the world, such as India and China,[6,7] and newer replacement compounds with modified chemical structures (i.e., novel PFAS) continue to be produced.[8]

Characterizing sources of exposure is important for understanding and mitigating human health risk. Studies of legacy PFAS have linked exposure to serious health effects including disrupted immune function, increased cholesterol, hepatic and developmental effects, and increased risk for some cancers.[9−14] Although there are a number of possible exposure sources,[15] diet is often considered the most important exposure pathway for general populations, except in areas with highly contaminated drinking water.[16,17] The extent to which different sources currently contribute to exposure in the general U.S. population is not well-understood.

The European Food Safety Authority (EFSA) and others have estimated that diet is the major source of exposure in the general European population.[18−21] While similar assessments

Received: November 26, 2024
Revised: April 15, 2025
Accepted: April 16, 2025
Published: May 14, 2025

https://doi.org/10.1021/acs.est.4c11872
Environ. Sci. Technol. 2025, 59, 9896−9906

for the general U.S. population are lacking, studies in the U.S. have indicated diet as an important source of exposure in places without PFAS water contamination.[22,23] In high freshwater seafood-consuming populations in Europe and the U.S., ingestion of fish has been identified as a major pathway of exposure.[24−26] Fish tissue data, while limited in the U.S., suggest that concentrations in marine fish are generally lower than freshwater fish.[27−29] Some PFAS can transfer from soil to plants,[30−32] and food crops and livestock feed can be contaminated when grown in agricultural areas where sewage sludge has been applied as a soil amendment (i.e., biosolids) or irrigated with contaminated water.[33−38] Legacy PFAS may come into contact with food during processing or migrate into food from packaging.[39−41]

The conventional approach to estimate human exposure based on measured concentrations in food is hampered by data limitations, as this requires timely and reliable chemical concentrations from a large representative sample of foods, as well as assumptions about intake and exposure factors (e.g., body weight and ingestion rate).[15,19] Much of the available data on PFAS in foods come from Europe but cannot be readily extrapolated to the U.S. due to variability in dietary exposures associated with geographic differences in industries, food production, and consumption patterns.[34,42] Limited data on PFAS concentrations in commercially available foods in the U.S. are available; much of the existing data were collected over 15 years ago,[43−46] but PFAS production and use in the U.S. have changed since then. The U.S. Food and Drug Administration (FDA) began analyzing PFAS in a subset of foods from the FDA Total Diet Study (TDS) in 2019 and has since found detectable levels of PFAS in a limited number of foods, though sample sizes for each food type are small.[47,48] Without a large representative sample of PFAS concentrations in food in the U.S., drawing conclusions about dietary exposure as done in Europe remains a challenge.

Alternatively, human biomonitoring data that measure PFAS in serum coupled with questionnaires can provide insights into dietary consumption patterns associated with increased levels of serum PFAS. Cross-sectional studies in the U.S. have demonstrated a relationship between levels of PFAS in serum and reported consumption of fast food or packaged foods, seafood, dairy, and meat.[49−54] However, many of these types of studies have relied on older biomonitoring data and do not account for contributions from drinking water or potential confounding from other sources.

Ingestion of contaminated drinking water has been identified as a major route of exposure to legacy PFAS in communities with heavily contaminated drinking water.[16,17,55−57] Less is known about the contribution of drinking water in general populations. In the nationwide Nurses' Health Study, water concentrations of PFOA and PFNA collected in 1989−1990 were significant predictors of serum levels for participants who drank eight or more cups of tap water per day.[58] More recently, concentrations of PFOA and PFOS measured in serum from California teachers in 2011−2013 were 30−40% higher among female teachers with detectable levels of these compounds in public water supplies compared to those without.[59] National surveillance data from the U.S. Environmental Protection Agency (USEPA) third Unregulated Contaminant Monitoring Rule (UCMR 3) (2013−2015) suggest that roughly 4% of large public water systems contain detectable levels of PFAS, estimated to affect millions of people living in the U.S.[60,61] Monitoring of public and private water supplies from 2016 to 2021 using lower detection limits suggests that the number of water systems impacted across the U.S. could be much greater.[62,63]

Comprehensive exposure studies are needed to assess contributions from multiple exposure pathways and account for potential confounding from other sources. To our knowledge, no existing biomonitoring studies simultaneously assess the effects of diet and drinking water on PFAS body burden using causal modeling rather than predictive modeling. Modeling serum concentrations with respect to multiple sources requires understanding exposures and pharmacokinetics to ensure appropriate model inputs. While empirical studies demonstrate a linear association between serum and water,[16,64] many studies log-transform serum concentrations to address assumptions regarding the distribution of model error terms. However, log transformation imposes an exponential dose−response relationship, which may not accurately explain the relationship between water, diet, and serum, necessitating alternative approaches. To address these issues, our study aimed to characterize PFAS body burden in a general California population and estimate the relationship between reported serum concentrations with diet and drinking water.

METHODS

Study Population. Participants of the California Regional Exposure (CARE) Study were recruited in three phases from southern and eastern California during 2018−2020 by Biomonitoring California, a state-run public health surveillance program. CARE-LA included 430 adults from Los Angeles county (2018), CARE-2 included 359 adults from Riverside, San Bernardino, Imperial, Mono, and Inyo counties (2019), and CARE-3 collected samples from 90 adults in San Diego and Orange counties (2020), though full recruitment for CARE-3 was not completed due to the COVID-19 pandemic (see the Supporting Information (SI), Figure S1). Eligibility was limited to adults 18 years or older who had been living in the study area for 12 or more months. The CARE study was designed to sequentially reach different regions of California, utilizing a convenience sampling approach with applied quotas by subregion, gender, and race/ethnicity to improve representation of the underlying population.[65] Participants provided blood samples for chemical analyses and answered surveys. Comparisons provided between the overall CARE study populations and levels from the National Health and Nutrition Examination Survey (NHANES) used weighted CARE data to be more representative of the underlying population (SI Section S-1, Table S1).[65]

Demographic Data. Surveys collected demographic information, including the place of birth, length of time in current home, length of time in the U.S., race/ethnicity, age, sex, pregnancy history, educational attainment, and income. For CARE-LA, gender was used as a proxy variable for sex.

PFAS in Serum. Twelve PFAS were measured in serum using solid-phase extraction high-performance liquid chromatography by the Environmental Chemistry Laboratory at the California Department of Toxic Substances Control: *N*-ethyl-perfluorooctane sulfonamido acetic acid (Et-PFOSA-AcOH), *N*-methyl-perfluorooctane sulfonamido acetic acid (Me-PFOSA-AcOH), perfluorobutanesulfonic acid (PFBS), perfluorodecanoic acid (PFDeA), perfluorododecanoic acid (PFDoA), perfluoroheptanoic acid (PFHpA), perfluorohexanesulfonic acid (PFHxS), perfluorooctane sulfonamide (PFOSA), PFOS, PFOA, perfluorononanoic acid (PFNA),

https://doi.org/10.1021/acs.est.4c11872
Environ. Sci. Technol. 2025, 59, 9896−9906

and perfluoroundecanoic acid (PFUnDA). Additional information on chemical analysis and quality control is summarized in SI Section S-2. The IUPAC names, molecular formulas, and internal standards are provided in SI Section S-3. More details on the quantification of PFAS are available elsewhere.[66] For limits of detection (LOD), see SI Section S-4, Table S3. Values of PFAS in serum below the LOD were substituted with the LOD divided by $\sqrt{2}$.

Dietary Exposure. Participants completed food frequency questionnaires (FFQs) prior to serum collection in which they were asked how many times in a typical week they eat various foods. Survey responses were categorized as "less than once per week or never," "1−3 times per week," "4−6 times per week," or "every day". To model the relationship between dietary consumption and PFAS body burden, response categories were converted into numerical weekly consumption frequencies using the median value for each category and 0.5 for the lowest category.[67] Portion size was not reported.

We analyzed dietary consumption (meals/week) for individual foods or food groups, including red meat, poultry, seafood, dairy, eggs, potatoes, brown rice, takeout, and packaged heat-at-home foods. Food samples were not collected for PFAS analyses during the course of the study. Foods previously associated with PFAS exposures were included, regardless of whether they were based on direct measurements in food[19,44,68,69] or empirical studies on dietary exposures.[49,52,53,70,71] The FFQ did not ask about coffee or tea consumption, which has more recently been linked to increased PFAS blood concentrations,[49,54] but we expect contributions from these sources to be partially accounted for by controlling for water in our analysis, described below. We combined consumption of shellfish, store-bought fish, and fish caught by the participant or someone they knew into a single "seafood" category in order to harmonize FFQ data across surveys from the three study areas in CARE. To reduce the number of food terms in the model, dairy milk and butter were combined into a single "dairy" category. Some foods were chosen as a proxy for a larger group of foods to reduce the number of food terms in the model. Brown rice was chosen over white rice or other rice products based on evidence that PFAS accumulate in the protein-rich bran and germ layers in brown rice more than white rice, as these layers are removed from white rice during processing.[72] Takeout food and store-bought food heated in paper or cardboard containers were included as packaged foods likely to be contaminated by PFAS used in food contact materials.[51,53,69,73] Popcorn consumption was too low to be analyzed.

Drinking Water Exposure. Participants' geocoded residential addresses at the time of survey were spatially joined to the corresponding public water system (PWS) service area using the California Drinking Water System Area Boundaries Layer (SABL) shapefile developed by the Division of Drinking Water of the California State Water Resources Control Board.[74] PFAS concentrations measured under UCMR 3 were linked to participants via Public Water System Identification (PWSID) numbers. The UCMR 3 cycle in 2013−2015 measured six PFAS (PFOA, PFOS, PFHxS, PFHpA, PFNA, and PFBS) in finished drinking water sampled from large public water systems (serving >10,000 people) and a subset of smaller water systems.[60] Monitoring in UCMR 3 occurred between January 2013 and December 2015, with the sampling frequency varying based on the water source.[75]

Because of the high reporting limits in UCMR 3 (20−40 ng/L) and consequent low detection frequencies, participants' exposure to PFAS in drinking water was characterized using a binary indicator of detection or nondetection of each analyte from any sampling event within the corresponding PWS (SI Section S-5, Table S4). While lower reporting limits are being used in UCMR 5 (2023−2025), serum collection in CARE took place beforehand, and various events in California occurred concurrently with or after serum collection that may have influenced drinking water concentrations (e.g., interim notification levels for PFOA and PFOS and state-mandated monitoring and reporting requirements).[76]

Statistical Analysis. *Selection Criteria.* A study population of 700 participants with demographic and dietary survey data for which both drinking water and serum levels could be evaluated was achieved after excluding participants based on the following criteria: if no serum was provided for laboratory analysis (n = 7) or a PFAS value was indicated as nonreportable (n = 2); if the PWS corresponding to the residential address could not be verified via SABL as of January 2024 (n = 78); if participants reported using private well water (n = 9); or if participants matched to a PWS not monitored in UCMR 3 (n = 19). Imputed values, calculated as part of creating a weighted data set, were used to replace missing values for CARE-LA and CARE-2 demographic data. Because CARE-3 sampling was not completed, weights and imputed values were not available for CARE-3. For CARE-LA and CARE-2, the SAS hot-deck procedure[77] indexed by age and geography was used to impute data for variables with fewer missing values (i.e., sex, race/ethnicity, and education); the procedure was indexed again using age, geography, and the previously imputed variables for imputing missing income. Participants were also excluded if they were missing diet or imputed covariate data (n = 64). Selection criteria are presented in Figure S2. Demographic characteristics for all participants enrolled into CARE (n = 879) (SI Section S-6, Table S5) are very similar to the population analyzed in this analysis (SI Section S-6, Table S6).

Causal Modeling and Covariate Selection. Since an estimation of the causal relationship between diet, water, and levels of PFAS in serum was an aim of this study, we constructed a directed acyclic graph (DAG) containing the outcome, exposure pathways—water, diet, and other sources (e.g., indoor environments)—and covariates to guide modeling of variables (Figure S3).[78,79] Different food types may contribute to PFAS in the diet, but their consumption cannot be assessed separately since greater consumption of some may be related to greater or lower consumption of others, potentially confounding the estimated effect. In contrast to studies that consider water and diet (or individual foods) separately, we assessed associations of serum PFAS with water concentration and consumption of various foods in the same model to account for potential confounding. Models were mutually adjusted using the same set of covariates and run separately for each PFAS analyte measured in serum. Indoor exposures have been shown to contribute to elevated levels of legacy PFAS in serum and can differ based on demographics.[18,80] Potential confounding from indoor exposures was minimized by an appropriate choice of demographic variables.

Covariates were considered for inclusion in the model if they influenced exposure and acted as predictors of PFAS concentration in serum. Previous studies have shown that

https://doi.org/10.1021/acs.est.4c11872
Environ. Sci. Technol. 2025, 59, 9896−9906

Table 1. Serum Concentrations of PFAS (ng/mL) in Study Participants, 2018−2020 (N = 700)

Analyte	25th percentile	median	75th percentile	maximum	LOD[a] (ng/mL)	percent > LOD (%)
PFHxS	0.41	0.75	1.30	17.1	0.0177	99.0
PFOA	0.74	1.14	1.65	15.0	0.0606	98.4
PFOS	1.33	2.57	4.14	30.8	0.0615	97.7
PFNA	0.17	0.29	0.45	8.78	0.0424	94.7
Me-PFOSA-AcOH	0.03	0.05	0.08	3.39	0.0114	88.3
PFUnDA	0.02	0.07	0.14	1.70	0.0285	73.6
PFDeA	0.04	0.09	0.16	4.74	0.0560	68.1

[a]LOD = limit of detection.

dietary habits differ by age, sex, race, education, and nativity,[52,53,81−86] and these factors have also been associated with differences in PFAS body burden.[5,71,87−89] Few studies have assessed the association between drinking water contamination and demographic characteristics, but evidence from 18 states, including California, suggests that disparities in drinking water exposure may be linked to race/ethnicity and socioeconomic status.[90,91]

All model covariates were categorical, except for age, which was treated as a continuous variable. To avoid issues with collinearity in multivariate analyses, sex and parity were modeled as a composite variable with three groups (i.e., male, female and nulliparous, and female with one or more pregnancies brought to term). Race/ethnicity was used as a proxy variable to account for social differences related to race and was categorized into five mutually exclusive groups (i.e., White, Hispanic or Latino, Black, Asian, and non-Hispanic multiracial/other). Due to low numbers, the "other" category included Native Hawaiian or Other Pacific Islander (NHOPI) and American Indian or Alaska Native (AIAN). Educational attainment was categorized as above or below high school-level education. Nativity was categorized into four groups (i.e., born in the U.S., Central/South America or the Caribbean, Asia, and other). Household income was categorized into four groups (i.e., <$25,000, $25,001−$75,000, $75,001−$150,000, or greater than $150,000).

Statistical analysis was restricted to PFAS detected in at least 65% of participants. Adjusted multivariate models for PFOS, PFOA, and PFHxS included drinking water and dietary exposure variables. Models examined only dietary variables if water data were not available or did not have any detections (i.e., PFDeA, PFUnDA, Me-PFOSA-AcOH, and PFNA).

Robust Linear Regression. In modeling the effects of diet and drinking water on PFAS body burden, we chose not to log-transform serum concentrations, as is routinely done in the context of skewed data, since this imposes an exponential dose−response function; previous studies have shown the relationship to be consistent with a linear function.[16,64] To address the right-skew of the serum data and apply an additive structure of the variables, we used robust linear regression on the nontransformed data since these methods are less sensitive to extreme values commonly observed with environmental data. Details of robust regression methods have been described elsewhere.[16,92−94] Briefly, the robust linear model applies a set of weights to reduce the influence of extreme values and allows for estimation between exposures and an outcome in the context of heavily tailed data, relaxing the assumptions of normally distributed residuals and constant variance. Given the relatively large sample size (n = 700), the normality assumption was not expected to be of major importance due to the central limit theorem. To ensure that model-based standard errors were not influenced by violations of constant variance, robust standard errors were determined using a bootstrapping procedure in R and used to generate 95% confidence intervals for estimated beta coefficients. Statistical significance is reported at the 0.05 level.

We performed model diagnostics and assessed linearity by visually examining diagnostic plots of residual vs fitted values. We assessed multicollinearity between multiple independent variables and concentrations of PFAS in serum using generalized variance inflation factors (GVIF),[95] with values over 10 identified as problematic. All analyses were performed in R (version 4.3.2), and robust regressions were performed using the *robustbase* package.

Sensitivity Analyses. To compare differences between robust regression and other modeling approaches, we performed a sensitivity analysis using traditional linear regression for crude and adjusted models for PFOA, PFOS, and PFHxS. We evaluated potential confounding of water and diet by modeling their effects on serum separately, adjusting for all other covariates. To assess whether combining fish and shellfish influenced results for seafood consumption, we analyzed fish and shellfish meals as separate terms in the model.

RESULTS

Study Population. Of the 879 participants initially enrolled in the study, 700 with complete or imputed demographic, serum, and public drinking water data were included in the analysis (Table S6). The study population was mainly female (59%) and Hispanic or Latino (39%) or White (34%). Of the 414 female participants, 254 (61%) reported having at least one pregnancy brought to term. The majority of participants had completed a college degree (59%). The average age of participants was 49 years old, and 56% of participants reported living in their home for longer than 5 years.

PFAS in Serum. Seven analytes were detected in 65% or more of participants' serum and included in analyses (Table 1 and Table S3). PFHxS had the highest detection frequency (99%). Spearman's correlation coefficients between concentrations of PFAS in serum ranged from 0.13 to 0.75, with the strongest correlation observed between PFHxS and PFOS (Figure S4). Weighted geometric mean concentrations from all participants in CARE-LA and CARE-2 were 49−72% of national levels reported in NHANES from a similar time period (Table S1).[65,96]

PFAS in Drinking Water. The overlap between PFAS monitored and detected in UCMR 3 and those detected in 65% or more of participants' serum samples included three PFAS: PFOA, PFOS, and PFHxS (Table 1). Eight percent of study participants had at least one of these three compounds

https://doi.org/10.1021/acs.est.4c11872
Environ. Sci. Technol. 2025, 59, 9896−9906

detected above the MRL in finished drinking water between 2013 and 2015; PFOA and PFOS were detected in the drinking water supply of roughly 7% of participants, and PFHxS was detected in 3.7% (SI Section S-5, Table S4). PFNA was not detected in any water systems in our study; so, drinking water contributions for PFNA could not be assessed. Drinking water concentrations in CARE were orders of magnitude lower than those reported from highly impacted areas.[16,57,97,98] Water concentrations in samples above the MRL in CARE are provided in Table S7.

Dietary Consumption. In CARE, dairy was consumed most frequently (4.3 times per week) while packaged heat-at-home foods were consumed least frequently (1.2 times per week) (Figure S5 and Table S8). Spearman's rank correlation coefficients between all foods showed weak to moderate associations (Figure S6). Foods included in the analysis were not strongly correlated with foods excluded from the analysis for the purpose of data reduction (chips, fries, margarine, white rice, and other rice) or due to infrequent consumption (popcorn). We observed the largest correlation between consumption of takeout food and fries ($\rho = 0.57$). Correlation coefficients between foods were positive with the exception of consumption of brown rice and red meat ($\rho = -0.10$) and brown rice and dairy ($\rho = -0.08$).

Associations between Serum and Diet. Reported seafood consumption was associated with increased serum PFDeA (0.0059 ng/mL, 95% CI: 0.0026, 0.0092), PFUnDA (0.010 ng/mL, 95% CI: 0.0054, 0.015), and PFNA (0.013 ng/mL, 95% CI: 0.0058, 0.021) (Table 2). Egg consumption was associated with increased serum PFDeA (0.0035 ng/mL; 95% CI: 0.0001, 0.0069) and PFNA (0.0073 ng/mL; 95% CI: 0.00017, 0.014). Brown rice consumption was associated with increased Me-PFOSA-AcOH (0.0030 ng/mL; 95% CI: 0.00056, 0.0054). Consumption of heated packaged food was negatively associated with serum PFOS (−0.11 ng/mL, 95% CI: −0.22, −0.011), PFUnDA (−0.0042 ng/mL; 95% CI: −0.0076, −0.00083), and PFDeA (−0.0039 ng/mL; 95% CI: −0.0072, −0.00057).

We observed other positive but not significant effects, including increased serum PFOS and reported consumption of seafood (0.014 ng/mL, 95% CI: −0.046, 0.074), eggs (0.044 ng/mL, 95% CI: −0.024, 0.11), dairy (0.022 ng/mL, 95% CI: −0.015, 0.059), brown rice (0.11 ng/mL, 95% CI: −0.0016, 0.21), poultry (0.024, 95% CI: −0.061, 0.11), and red meat (0.083 ng/mL, 95% CI: −0.019, 0.19), as well as increased serum PFOA and reported consumption of eggs (0.017 ng/mL; 95% CI: −0.011, 0.044) and brown rice (0.024 ng/mL; 95% CI: −0.015, 0.063) (Table 2).

Associations between Serum and Drinking Water. The largest effect for drinking water was observed for PFHxS, with participants linked to detectable levels of PFHxS in their PWS having 0.64 ng/mL (95% CI: 0.058, 1.23) higher levels of PFHxS in serum, compared to those without detectable levels in water (Table 2). Water system detection of PFOA was associated with a 0.26 ng/mL (95% CI: 0.077, 0.43) increase in serum concentration. The estimated increase in serum PFOS associated with detection of PFOS in water was of similar magnitude, though nonsignificant (0.39 ng/mL, 95% CI: −0.076, 0.86).

Model Diagnostics. Visual inspection of residual vs fitted value plots helped to verify the assumption of linearity (SI Section S-12). Plots did not show any discernible pattern in the residuals, which would have indicated a nonlinear

Table 2. Estimated Effects of Diet and Drinking Water on Concentrations of PFAS in Serum (ng/mL) from CARE Participants ($N = 700$) and 95% Confidence Intervals[a]

	PFOA	PFOS	PFHxS	PFDeA	PFUnDA	Me-PFOSA-AcOH	PFNA
intercept	0.51 (0.21, 0.81)	0.55 (−0.34, 1.40)	0.44 (0.17, 0.71)	0.047 (0.014, 0.080)	0.050 (0.018, 0.081)	0.034 (0.019, 0.050)	0.11 (0.065, 0.21)
water	0.26 (0.077, 0.43)	0.39 (−0.076, 0.86)	0.64 (0.058, 1.23)	N/A	N/A	N/A	N/A
seafood	−0.010 (−0.036, 0.015)	0.014 (−0.046, 0.074)	0.00013 (−0.022, 0.023)	0.0059 (0.0026, 0.0092)	0.010 (0.0054, 0.015)	−0.00043 (−0.0019, 0.0010)	0.013 (0.0058, 0.021)
eggs	0.017 (−0.011, 0.044)	0.044 (−0.024, 0.11)	0.014 (−0.011, 0.040)	0.0035 (0.0001, 0.0069)	0.0032 (−0.00038, 0.0067)	−0.00047 (−0.0019, 0.00095)	0.0073 (0.00017, 0.014)
dairy	−0.0061 (−0.020, 0.0078)	0.022 (−0.015, 0.059)	0.0023 (−0.0099, 0.015)	0.0000 (−0.0019, 0.0018)	−0.0006 (−0.0022, 0.0010)	0.00035 (−0.00056, 0.0013)	−0.00015 (−0.0041, 0.0038)
brown rice	0.024 (−0.015, 0.063)	0.11 (−0.0016, 0.21)	0.024 (−0.018, 0.066)	0.00049 (−0.0041, 0.0051)	0.0011 (−0.0042, 0.0063)	0.0030 (0.00056, 0.0054)	−0.00037 (−0.012, 0.011)
poultry	0.0019 (−0.029, 0.033)	0.024 (−0.061, 0.11)	−0.0024 (−0.030, 0.025)	−0.0017 (−0.0045, 0.0011)	−0.0024 (−0.0057, 0.00099)	0.00056 (−0.00095, 0.0021)	−0.0045 (−0.013, 0.0040)
potato	−0.012 (−0.052, 0.027)	−0.047 (−0.15, 0.052)	−0.022 (−0.055, 0.010)	−0.00077 (−0.0050, 0.0035)	−0.0037 (−0.0077, 0.00034)	0.00083 (−0.00094, 0.0026)	−0.0047 (−0.014, 0.0052)
red meat	0.0023 (−0.033, 0.037)	0.083 (−0.019, 0.19)	−0.011 (−0.046, 0.023)	0.0025 (−0.0014, 0.0065)	0.0018 (−0.0017, 0.0052)	0.00023 (−0.0016, 0.0021)	0.0084 (−0.0012, 0.018)
takeout	−0.0065 (−0.058, 0.045)	−0.074 (−0.17, 0.020)	−0.0050 (−0.047, 0.037)	−0.0043 (−0.0081, −0.00045)	−0.0036 (−0.0073, 0.00019)	−0.00099 (−0.0030, 0.0010)	−0.011 (−0.022, −0.00060)
heat-at-home	−0.022 (−0.065, 0.022)	−0.11 (−0.22, −0.011)	−0.021 (−0.063, 0.022)	−0.0039 (−0.0072, −0.00057)	−0.0042 (−0.0076, −0.00083)	−0.0016 (−0.0036, 0.00031)	−0.010 (−0.022, 0.0019)

[a]Robust regression models were run separately for each analyte and adjusted for drinking water (PFOA, PFOS, and PFHxS), other dietary items, age, sex, parity, education, income, race, and nativity. Estimates represent the change in serum concentrations (ng/mL) associated with detect vs nondetect for water, or weekly meal consumed, adjusting for potential confounders. Drinking water contributions were not assessed for PFDeA, PFUnDA, Me-PFOSA-AcOH, or PFNA.

https://doi.org/10.1021/acs.est.4c11872
Environ. Sci. Technol. 2025, 59, 9896−9906

relationship between the predictors and serum concentrations. Diagnostic plots also helped identify observations with large residuals that were down-weighted in the robust regression. Assessment of multicollinearity showed that GVIF values were below 10 for all terms in the adjusted models and therefore no corrective actions were taken (SI Section S-9, Table S9). Bootstrapped robust errors are provided in Tables S10 and S11.

Sensitivity Analyses. Robust regression and traditional regression showed overall similar beta estimates (Table S10a–c). Dietary estimates from robust regression models without adjustment for drinking water did not meaningfully differ from those reported in Table 2, suggesting that diet was not confounded by water in this population, after adjusting for other covariates (Table S12). The drinking water estimates for PFOA and PFHxS, adjusting for all covariates except diet, were very similar to those with adjustment for diet, but the estimate for PFOS, when modeled without adjustment for diet, decreased from 0.39 to 0.29 (Table S13). When modeled separately from fish meals, the beta coefficient for shellfish appeared larger for PFOS (0.16 ng/mL; 95% CI: −0.023, 0.35) compared to the beta coefficient for seafood combined (0.014 ng/mL, 95% CI: −0.046, 0.074) (Tables S14 and S15). Similar trends but smaller in magnitude were seen for PFOA and PFDeA; otherwise, beta coefficients stayed the same.

DISCUSSION

This study examined the association between diet, drinking water, and concentrations of legacy PFAS in serum among a large and diverse adult study population living in southern and eastern California in 2018–2020. Serum concentrations in CARE-LA and CARE-2 (2018–2019) were lower than concentrations reported in NHANES (2017–2018) (Table 1 and Table S1).[5] Additionally, drinking water concentrations for CARE participants (Table S7) were orders of magnitude lower than concentrations that have been reported in highly impacted areas.[16,97,98] We observed positive associations between some PFAS and consumption of seafood, eggs, and brown rice, as well as elevated serum concentrations in participants with detectable levels of PFAS in drinking water supplies. Our results suggest that diet and drinking water both contributed to the body burden of legacy PFAS in this study population. However, we observed distinct differences for diet in our study compared to other U.S. studies that rely on dietary data from more than 10 years ago, which we hypothesize may be due to diminished levels of legacy PFAS in foods.

Associations between Diet and Serum. While the positive associations we observed between serum concentrations of some PFAS and animal products were consistent with previous epidemiological studies in the U.S. (e.g., seafood and eggs),[49,50,53] we did not observe as many associations between legacy PFAS and diet compared to earlier studies from before 2014. For example, in the Diabetes Prevention Program (DPP) study from 1996 to 1999, reported consumption of poultry was associated with increased concentrations of PFOS, PFHxS, and PFNA, and meat consumption was associated with elevated PFOS, PFOA, PFHxS, Et-PFOSA-AcOH, Me-PFOSA-AcOH, and PFNA. In CARE, serum PFAS did not appear strongly associated with red meat and poultry consumption. We observed the largest positive effects for poultry and red meat consumption on serum PFOS, but these estimates were not significant (Table 2). We observed positive associations between seafood consumption and levels of PFNA, PFDeA, and PFUnDA, as well as a nonsignificant positive effect for PFOS. However, we did not observe associations between seafood and PFHxS or Me-PFOSA-ACOH, which was reported in the DPP study. Another study from Norway conducted in 2003 showed associations between multiple legacy PFAS and seafood consumption (PFHxS, PFHpS, PFOS, PFOSA, PFOA, PFNA, PFDA, PFUnDA, PFDoDA, and PFTrDA) and meat consumption (PFNA, PFDA, PFHpS, and PFOS).[21] Qualitatively, these differences suggest fewer associations between diet and serum PFAS in CARE (2018–2020) compared to associations with diet observed 20 years ago.

Data on the concentrations of PFAS in foods from U.S. stores are limited, but concentrations recently reported by the U.S. FDA were relatively low compared to concentrations reported previously and could partly explain differences between CARE and studies from many years ago. In contrast to the Canadian Total Diet Study (1992–2004), which measured relatively high concentrations of PFOA in popcorn (3.6 ng/g) as well as PFOS in meat (4.5 ng/g) and fish (2.6 ng/g),[69] processed foods from Kansas (n = 172) collected in 2018 and analyzed as part of the U.S. FDA Total Diet Study (TDS) showed limited detection of PFAS in canned tuna, frozen fish sticks, and protein powder, all with concentrations below 0.14 ng/g.[47] To date, less than 3% of TDS samples analyzed by the FDA have shown detectable concentrations of PFAS.[99] More recent TDS analyses by the FDA from 2019 and 2021 with subpart per billion (ng/g) detection limits found mostly nondetectable concentrations of 16 PFAS in a range of foods analyzed.[99] Of the foods analyzed, two composite beef samples showed 0.041–0.086 ng/g PFOS; otherwise, most of the other TDS samples with detectable concentrations were seafood, all with concentrations less than 0.23 ng/g.[99] In 2021, targeted FDA sampling of seafood from Washington DC supermarkets showed concentrations below 0.51 ng/g, except for clams imported from China and one sample of canned tuna, which had 0.89 ng/g PFUnDA.[48] While these data are limited in sample size and may not represent all commercially available foods in the U.S., samples purchased from large grocery store chains, whose products are distributed nationally, could be informative for the U.S. population. The recent samples analyzed by the U.S. FDA show lower concentrations than the Canadian Total Diet Study samples from over 20 years ago. This could suggest declining concentrations of legacy PFAS in food over the past 20 years and/or geographical differences in food samples.

In contrast to findings from NHANES (2003–2014) that showed that consumption of fast food and pizza was positively associated with serum concentrations of PFOA, PFNA, PFDA, PFHxS, and PFOS (adjusting for the cycle year),[51] we observed unexpected negative associations with takeout food and packaged heat-at-home foods. Differences between current patterns of use for legacy PFAS in food contact materials and those from 20 years ago may explain why we did not observe positive associations. While foods contaminated via bioaccumulation likely reflect historic uses of PFAS and their release into the environment, foods contaminated via packaging are likely to reflect more recent patterns of use in food contact materials.[19] In addition to the voluntary phaseout of legacy PFAS from manufacturing, which started in the early 2000s, the FDA restricted the use of PFOA and PFOS in paper food contact materials in favor of shorter-chain alternatives in 2016.[100] These shifts would be expected to produce a steep

https://doi.org/10.1021/acs.est.4c11872
Environ. Sci. Technol. 2025, 59, 9896–9906

decline of legacy PFAS in packaged food products manufactured in the U.S. and, by extension, in serum analyzed several years later. Our findings are supported by the Chemicals in our Body (CIOB) study (2014−2019), which similarly observed associations between serum and consumption of animal products (e.g., seafood, meat, and dairy), but not for processed foods or takeout food.[53] The reason for the negative relationship between takeout and packaged heat-at-home foods and serum in our study is unknown but could be due to residual confounding, as discussed in the limitations below.

Associations between Drinking Water and Serum. Drinking water has been identified as an important source of exposure, particularly in heavily contaminated populations.[16,55,64] In this analysis, drinking water concentrations from UCMR 3 linked to CARE participants are considerably lower than those reported in heavily contaminated areas (Table S7). However, we observed significantly higher serum concentrations in participants residing in water service areas with detectable levels of PFOA and PFHxS compared to those residing in service areas without detectable levels (Table 2), suggesting that relatively low concentrations in drinking water can lead to elevated levels in serum. While the state of California does not have fluorochemical manufacturers, drinking water supplies in CARE may have been contaminated by other industrial facilities that use PFAS, landfills, wastewater treatment plants, or AFFF used at airports and military training areas.[55,61]

While the methods used in our analysis make it difficult to quantitatively compare our results to empirical studies that assessed the continuous relationship between water and serum (e.g., Hoffman et al.[16]), the positive associations we observed between water and serum were qualitatively consistent with findings reported previously in the U.S. for general populations. Our results are partially supported by findings from the California Teachers' Study (CTS) (2011−2015), which observed higher serum concentrations of PFOA and PFOS, but not PFHxS, among women living in zip codes with detectable levels of these analytes in drinking water systems compared to those living in zip codes without any detections (using water data from UCMR 3).[59] The positive association we observed between PFOA and serum was also consistent with the older nationwide Nurses' Health Study, which observed significant associations between PFOA and PFNA in drinking water from 1989 to 1990 and levels in serum from women who consumed eight or more cups of water a day.[58] PFNA was not detected in any of the water systems included in CARE, possibly due to high reporting levels in UCMR 3 relative to methods used in NHS; so, contributions from PFNA in our study could not be assessed.

The larger effect we observed for PFHxS in drinking water compared to PFOA and PFOS may be partly due to the longer half-life of PFHxS (estimated to be 5−8 years) compared to PFAS with shorter half-lives (3−5 years for PFOS and 2−4 years for PFOA).[101,102] The Child Health and Development Studies (CHDS) (2010−2013) also observed larger effects for PFHxS in drinking water for women living in U.S. cities with detectable concentrations in UCMR 3 (n = 6).[52]

Strengths and Limitations. Study strengths included the use of recently collected biomonitoring data from a large and diverse general study population and the application of prior knowledge on exposure and pharmacokinetics to guide our model assumptions. We structured this analysis to utilize causal inference to estimate exposures via water and diet, such that, if correct, intervening on an exposure could be interpreted as leading to a corresponding reduction in serum concentrations. To our knowledge, this is the first analysis to estimate the effects of drinking water and diet simultaneously on concentrations of PFAS measured in U.S. serum. We used robust linear regression to model nontransformed serum concentrations based on empirical studies that have demonstrated a linear relationship between water and serum,[16,98] as well as evidence from toxicokinetic models that are consistent with first-order pharmacokinetics at steady-state concentrations.[64]

We developed a DAG based on the literature to evaluate the causal pathways between drinking water, diet, and PFAS body burden. Even when analyzed one at a time, the DAG used for this analysis implied that drinking water and diet should not be confounded in this analysis because certain variables that link the two exposures were controlled for (Figure S3). Sensitivity analyses that modeled diet and drinking water separately were overall consistent with our DAG (Tables S12 and S13). Finally, our analysis modeled foods together to reduce potential confounding by other foods.

Our analysis had several limitations, including the use of FFQs that may have introduced nondifferential error associated with self-reported diet,[103] and the high reporting limits in UCMR 3 (20−40 ng/L), which resulted in few water system detections in our study area. This limited our ability to analyze the continuous relationship between water and serum, leading to the use of a dichotomous exposure variable, which may have introduced exposure misclassification. Our dietary assessment lacked detailed information on fish and shellfish consumption; concentrations in seafood can vary by species, source (e.g., marine vs freshwater), and geography.[28,48] A sensitivity analysis suggested that combining fish and shellfish may have introduced measurement error, biasing results toward the null (Tables S14 and S15).

We cannot rule out residual confounding from other sources, such as indoor air, dust, or dermal exposure to cosmetics. Inhalation of PFAS precursors in indoor air can lead to increased concentrations of legacy PFAS in serum and may be an important source of exposure.[18,80,104] Although we controlled for sociodemographic factors linked to indoor exposures (e.g., age, sex, income, and race/ethnicity), residual confounding is possible, as indicated by the unexpected negative associations with takeout and heat-at-home foods, which are unlikely to be causal. Our study's conclusions are limited to the legacy PFAS analyzed in the CARE study and UCMR 3, and while dietary exposure to legacy PFAS may have decreased, exposure to newer PFAS, or those not routinely measured, may have increased.[105,106]

Overall, our study quantified legacy PFAS exposure in the CARE population and identified possible sources and behaviors associated with increased body burden, providing knowledge that can guide strategies to protect public health. While diet is suggested to be a major source of exposure in general populations, we did not observe as many associations with diet in this population compared to findings in other populations from many years ago, possibly due to changes in the contribution of diet over the past 20 years. Future analyses using paired measurements of PFAS in food, water, and serum, while not available for this population, would be instructive for source derivation. Future work should also consider additional PFAS and exposure pathways, including inhalation of precursor

https://doi.org/10.1021/acs.est.4c11872
Environ. Sci. Technol. 2025, 59, 9896−9906

compounds in the indoor environment, which might be important contributors to PFAS body burden.

■ ASSOCIATED CONTENT

Supporting Information

The Supporting Information is available free of charge at https://pubs.acs.org/doi/10.1021/acs.est.4c11872.

Additional methodological details and supporting tables, including quality control, study population demographics, weighted serum concentrations for comparison with NHANES, drinking water detection frequencies, detectable drinking water concentrations, dietary consumption frequencies, sensitivity analyses, and model diagnostics (PDF)

■ AUTHOR INFORMATION

Corresponding Author

Emily H. Pennoyer − *Boston University School of Public Health, Boston, Massachusetts 02118, United States;* orcid.org/0000-0002-5181-4851; Email: eghammel@bu.edu

Authors

Toki Fillman − *California Department of Public Health, Richmond, California 94804, United States*

Wendy Heiger-Bernays − *Boston University School of Public Health, Boston, Massachusetts 02118, United States;* orcid.org/0000-0002-0091-9629

Kathleen R. Attfield − *California Department of Public Health, Richmond, California 94804, United States*

Hyoung-Gee Baek − *California Department of Toxic Substances Control, Berkeley, California 94710, United States*

Songmei Gao − *California Department of Toxic Substances Control, Berkeley, California 94710, United States*

Sabrina Smith − *California Department of Toxic Substances Control, Berkeley, California 94710, United States*

June-Soo Park − *California Department of Toxic Substances Control, Berkeley, California 94710, United States; University of California San Francisco, San Francisco, California 94158, United States*

Nerissa Wu − *California Department of Public Health, Richmond, California 94804, United States;* orcid.org/0000-0001-6680-9900

Thomas F. Webster − *Boston University School of Public Health, Boston, Massachusetts 02118, United States;* orcid.org/0000-0003-4896-9323

Complete contact information is available at: https://pubs.acs.org/10.1021/acs.est.4c11872

Funding

The California Regional Exposure Study was supported by the Centers for Disease Control and Prevention Cooperative Agreement U88EH001148. E.H.P. received support from the National Institute of Environmental Health Sciences (T32 304 E2014562).

Notes

The authors declare no competing financial interest.

■ ACKNOWLEDGMENTS

We express our appreciation to the CARE study participants who contributed to this research. The authors thank Alan Hubbard, Ph.D. and Scott Bartell, M.S., Ph.D. for their assistance with robust methods and statistical approaches used in our analysis.

■ REFERENCES

(1) Hammel, E.; Webster, T. F.; Gurney, R.; Heiger-Bernays, W. Implications of PFAS definitions using fluorinated pharmaceuticals. *iScience* **2022**, *25* (4), No. 104020.

(2) Glüge, J.; Scheringer, M.; Cousins, I. T.; Dewitt, J. C.; Goldenman, G.; Herzke, D.; Lohmann, R.; Ng, C. A.; Trier, X.; Wang, Z. An overview of the uses of per- and polyfluoroalkyl substances (PFAS). *Environ. Sci. Process. Impacts.* **2020**, *22* (12), 2345−2373.

(3) Brase, R. A.; Mullin, E. J.; Spink, D. C. Legacy and Emerging Per- and Polyfluoroalkyl Substances: Analytical Techniques, Environmental Fate, and Health Effects. *Int. J. Mol. Sci.* **2021**, *22* (3), 995.

(4) Buck, R. C.; Franklin, J.; Berger, U.; Conder, J. M.; Cousins, I. T.; Voogt, P. D.; Jensen, A. A.; Kannan, K.; Mabury, S. A.; van Leeuwen, S. P. J. Perfluoroalkyl and polyfluoroalkyl substances in the environment: Terminology, classification, and origins. *Integr. Environ. Assess. Manage.* **2011**, *7* (4), 513−541.

(5) Centers for Disease Control and Prevention, *Early Release: Per- and Polyfluorinated Substances (PFAS) Tables, NHANES 2011−2018*. December 16, 2021. https://www.cdc.gov/exposurereport/pfas_early_release.html (accessed 2022−02−07).

(6) Mukherjee Das, A.; Janardhanan, R. Accelerating Research and Policy on PFAS in India. *Environ. Epidemiol.* **2022**, *6* (2), No. e199.

(7) Kenji, A. *China releases draft plan for new chemical pollutants management: Regulations on PFOA, decaBDE, other hazardous substances*. Enviliacne ASIA, October 2021. https://enviliance.com/regions/east-asia/cn/report_4504. (accessed 2024−02−12).

(8) Gomis, M. I.; Vestergren, R.; Borg, D.; Cousins, I. T. Comparing the toxic potency in vivo of long-chain perfluoroalkyl acids and fluorinated alternatives. *Environ. Int.* **2018**, *113*, 1−9.

(9) Nelson, J. W.; Hatch, E. E.; Webster, T. F. Exposure to polyfluoroalkyl chemicals and cholesterol, body weight, and insulin resistance in the general U.S. population. *Environ. Health Perspect.* **2010**, *118* (2), 197−202.

(10) Fenton, S. E.; Ducatman, A.; Boobis, A.; DeWitt, J. C.; Lau, C.; Ng, C.; Smith, J. S.; Roberts, S. M. Per- and Polyfluoroalkyl Substance Toxicity and Human Health Review: Current State of Knowledge and Strategies for Informing Future Research. *Environ. Toxicol. Chem.* **2020**, *40* (3), 606−630.

(11) Grandjean, P.; Heilmann, C.; Weihe, P.; Nielsen, F.; Mogensen, U. B.; Budtz-Jorgensen, E. Serum Vaccine Antibody Concentrations in Adolescents Exposed to Perfluorinated Compounds. *Environ. Health Perspect.* **2017**, *125* (7), No. 77018.

(12) Zahm, S.; Bonde, J. P.; Chiu, W. A.; Hoppin, J.; Kanno, J.; Abdallah, M.; Blystone, C. R.; Calkins, M. M.; Dong, G. H.; Dorman, D. C.; Fry, R.; Guo, H.; Haug, L. S.; Hofmann, J. N.; Iwasaki, M.; Machala, M.; Mancini, F. R.; Maria-Engler, S. S.; Møller, P.; Ng, J. C.; Pallardy, M.; Post, G. B.; Salihovic, S.; Schlezinger, J.; Soshilov, A.; Steenland, K.; Steffensen, I. L.; Tryndyak, V.; White, A.; Woskie, S.; Fletcher, T.; Ahmadi, A.; Ahmadi, N.; Benbrahim-Tallaa, L.; Bijoux, W.; Chittiboyina, S.; de Conti, A.; Facchin, C.; Madia, F.; Mattock, H.; Merdas, M.; Pasqual, E.; Suonio, E.; Viegas, S.; Zupunski, L.; Wedekind, R.; Schubauer-Berigan, M. K. Carcinogenicity of perfluorooctanoic acid and perfluorooctanesulfonic acid. *Lancet Oncol.* **2024**, *25* (1), 16−17.

(13) *Toxicological Profile for Perfluoroalkyls*. Agency for Toxic Substances and Disease Registry (US): Atlanta (GA); May 2021.

(14) Schlezinger, J. J.; Hyötyläinen, T.; Sinioja, T.; Boston, C.; Puckett, H.; Oliver, J.; Heiger-Bernays, W.; Webster, T. F. Perfluorooctanoic acid induces liver and serum dyslipidemia in humanized PPARα mice fed an American diet. *Toxicol. Appl. Pharmacol.* **2021**, *426*, No. 115644.

(15) De Silva, A.; Armitage, J.; Bruton, T.; Dassuncao, C.; Heiger-Bernays, W.; Hu, X.; Kärrman, A.; Kelly, B.; Ng, C.; Robuck, A.; Sun,

M.; Webster, T.; Sunderland, E. PFAS Exposure Pathways for Humans and Wildlife: A Synthesis of Current Knowledge and Key Gaps in Understanding. *Environ. Toxicol. Chem.* **2020**, *40* (3), 631−657.
(16) Hoffman, K.; Webster, T. F.; Bartell, S. M.; Weisskopf, M. G.; Fletcher, T.; Vieira, V. M. Private drinking water wells as a source of exposure to perfluorooctanoic acid (PFOA) in communities surrounding a fluoropolymer production facility. *Environ. Health Perspect.* **2011**, *119* (1), 92−97.
(17) Steenland, K.; Jin, C.; MacNeil, J.; Lally, C.; Ducatman, A.; Vieira, V.; Fletcher, T. Predictors of PFOA Levels in a Community Surrounding a Chemical Plant. *Environ. Health Perspect.* **2009**, *117* (7), 1083−1088.
(18) Poothong, S.; Papadopoulou, E.; Padilla-Sánchez, J. A.; Thomsen, C.; Haug, L. S. Multiple pathways of human exposure to poly- and perfluoroalkyl substances (PFASs): From external exposure to human blood. *Environ. Int.* **2020**, *134*, No. 105244.
(19) Schrenk, D.; Bignami, M.; Bodin, L.; Chipman, J. K.; del Mazo, J.; Grasl-Kraupp, B.; Hogstrand, C.; Hoogenboom, L.; Leblanc, J. C.; Nebbia, C. S.; Nielsen, E.; Ntzani, E.; Petersen, A.; Sand, S.; Vleminckx, C.; Wallace, H.; Barregård, L.; Ceccatelli, S.; Cravedi, J. P.; Halldorsson, T. I.; Haug, L. S.; Johansson, N.; Knutsen, H. K.; Rose, M.; Roudot, A. C.; Van Loveren, H.; Vollmer, G.; Mackay, K.; Riolo, F.; Schwerdtle, T. Risk to human health related to the presence of perfluoroalkyl substances in food. *EFSA J.* **2020**, *18* (9), No. e06223.
(20) Fromme, H.; Tittlemier, S. A.; Völkel, W.; Wilhelm, M.; Twardella, D. Perfluorinated compounds − Exposure assessment for the general population in western countries. *Int. J. Hyg. Environ. Health.* **2009**, *212* (3), 239−270.
(21) Haug, L. S.; Thomsen, C.; Brantsæter, A. L.; Kvalem, H. E.; Haugen, M.; Becher, G.; Alexander, J.; Meltzer, H. M.; Knutsen, H. K. Diet and particularly seafood are major sources of perfluorinated compounds in humans. *Environ. Int.* **2010**, *36* (7), 772−778.
(22) National Academies of Sciences, Engineering and Medicine *Guidance on PFAS Exposure, Testing, and Clinical Follow-Up*, National Academies Press, 2022.
(23) Vestergren, R; Cousins, I. T. Human dietary exposure to per- and poly-fluoroalkyl substances (PFASs). In *Persistent Organic Pollutants and Toxic Metals in Foods*; Rose, M., Fernandes, A., Eds; Woodhead Publishing: Cambridge, 2013; pp 279−307.
(24) Berger, U.; Glynn, A.; Holmström, K. E.; Berglund, M.; Ankarberg, E. H.; Törnkvist, A. Fish consumption as a source of human exposure to perfluorinated alkyl substances in Sweden - analysis of edible fish from Lake Vättern and the Baltic Sea. *Chemosphere.* **2009**, *76* (6), 799−804.
(25) Weihe, P.; Kato, K.; Calafat, A. M.; Nielsen, F.; Wanigatunga, A. A.; Needham, L. L.; Grandjean, P. Serum concentrations of polyfluoroalkyl compounds in Faroese whale meat consumers. *Environ. Sci. Technol.* **2008**, *42* (16), 6291−6295.
(26) Wattigney, W. A.; Savadatti, S. S.; Liu, M.; Pavuk, M.; Lewis-Michl, E.; Kannan, K.; Wang, W.; Spliethoff, H.; Marquez-Bravo, L.; Hwang, S. A. Biomonitoring of per- and polyfluoroalkyl substances in minority angler communities in central New York State. *Environ. Res.* **2022**, *204*, No. 112309.
(27) Zafeiraki, E.; Gebbink, W. A.; Hoogenboom, R. L. A. P.; Kotterman, M.; Kwadijk, C.; Dassenakis, E.; van Leeuwen, S. P. J. Occurrence of perfluoroalkyl substances (PFASs) in a large number of wild and farmed aquatic animals collected in the Netherlands. *Chemosphere* **2019**, *232*, 415−423.
(28) Barbo, N.; Stoiber, T.; Naidenko, O. V.; Andrews, D. Q. Locally caught freshwater fish across the United States are likely a significant source of exposure to PFOS and other perfluorinated compounds. *Environ. Res.* **2023**, *220*, No. 115165.
(29) Vi, P. T.; Ngoc, N. T.; Quang, P. D.; Dam, N. T.; Tue, N. M.; Tuyen, L. H.; Viet, P. H.; Anh, D. H. Perfluoroalkyl substances in freshwater and marine fish from northern Vietnam: Accumulation levels, profiles, and implications for human consumption. *Mar. Pollut. Bull.* **2022**, *182*, No. 113995.
(30) Costello, M. C. S.; Lee, L. S. Sources, Fate, and Plant Uptake in Agricultural Systems of Per- and Polyfluoroalkyl Substances. *Curr. Pollut. Rep.* **2020**, *10*, 799−819.
(31) Scearce, A. E.; Goossen, C. P.; Schattman, R. E.; Mallory, E. B.; MaCrae, J. D. Linking drivers of plant per- and polyfluoroalkyl substance (PFAS) uptake to agricultural land management decisions. *Biointerphases.* **2023**, *18* (4), No. 040801.
(32) Stahl, T.; Heyn, J.; Thiele, H.; Hüther, J.; Failing, K.; Georgii, S.; Brunn, H. Carryover of perfluorooctanoic acid (PFOA) and perfluorooctane sulfonate (PFOS) from soil to plants. *Arch. Environ. Contam. Toxicol.* **2009**, *57* (2), 289−298.
(33) Brown, J. B.; Conder, J. M.; Arblaster, J. A.; Higgins, C. P. Assessing Human Health Risks from Per- And Polyfluoroalkyl Substance (PFAS)-Impacted Vegetable Consumption: A Tiered Modeling Approach. *Environ. Sci. Technol.* **2020**, *54* (23), 15202−15214.
(34) Domingo, J. L.; Nadal, M. Per- and polyfluoroalkyl substances (PFASs) in food and human dietary intake: A review of the recent scientific literature. *J. Agric. Food Chem.* **2017**, *65* (3), 533−543.
(35) Johnson, G. R. PFAS in soil and groundwater following historical land application of biosolids. *Water Res.* **2022**, *211*, No. 118035.
(36) Pepper, I. L.; Brusseau, M. L.; Prevatt, F. J.; Escobar, B. A. Incidence of PFAS in soil following long-term application of class B biosolids. *Sci. Total Environ.* **2021**, *793*, No. 148449.
(37) Stahl, T.; Gassmann, M.; Falk, S.; Brunn, H. Concentrations and Distribution Patterns of Perfluoroalkyl Acids in Sewage Sludge and in Biowaste in Hesse, Germany. *J. Agric. Food Chem.* **2018**, *66* (39), 10147−10153.
(38) Simones, T. L.; Evans, C.; Goossen, C. P.; Kersbergen, R.; Mallory, E. B.; Genualdi, S.; Young, W.; Smith, A. E. Uptake of Per- and Polyfluoroalkyl Substances in Mixed Forages on Biosolid-Amended Farm Fields. *J. Agric. Food Chem.* **2024**, *72* (42), 23108−23117.
(39) Still, M.; Schlummer, M.; Gruber, L.; Fiedler, D.; Wolz, G. Impact of industrial production and packaging processes on the concentration of per- and polyfluorinated compounds in milk and dairy products. *J. Agric. Food Chem.* **2013**, *61* (38), 9052−9062.
(40) Begley, T. H.; White, K.; Honigfort, P.; Twaroski, M. L.; Neches, R.; Walker, R. A. Perfluorochemicals: potential sources of and migration from food packaging. *Food Addit. Contam.* **2005**, *22* (10), 1023−1031.
(41) Trier, X.; Granby, K.; Christensen, J. H. Polyfluorinated surfactants (PFS) in paper and board coatings for food packaging. *Environ. Sci. Pollut. Res. Int.* **2011**, *18* (7), 1108−1120.
(42) Picó, Y.; Farré, M.; Llorca, M.; Barceló, D. Perfluorinated Compounds in Food: A Global Perspective. *Crit. Rev. Food Sci. Nutr.* **2011**, *51* (7), 605−625.
(43) 3M Company. *Analysis of PFOS, FOSA and PFOA From Various Food Matrices Using HPLC Electrospray Mass Spectrometry (Analytical Report)*; 2001.
(44) Schecter, A.; Colacino, J.; Haffner, D.; Patel, K.; Opel, M.; Päpke, O.; Birnbaum, L. Perfluorinated Compounds, Polychlorinated Biphenyls, and Organochlorine Pesticide Contamination in Composite Food Samples from Dallas, Texas, USA. *Environ. Health Perspect.* **2010**, *118* (6), 796−802.
(45) Ruffle, B.; Vedagiri, U.; Bogdan, D.; Maier, M.; Schwach, C.; Murphy-Hagan, C. Perfluoroalkyl Substances in U.S. market basket fish and shellfish. *Environ. Res.* **2020**, *190*, No. 109932.
(46) Young, W. M.; South, P.; Begley, T. H.; Noonan, G. O. Determination of Perfluorochemicals in Fish and Shellfish Using Liquid Chromatography−Tandem Mass Spectrometry. *J. Agric. Food Chem.* **2013**, *61* (46), 11166−11172.
(47) Genualdi, S.; Beekman, J.; Carlos, K.; Fisher, C. M.; Young, W.; DeJager, L.; Begley, T. Analysis of per- and poly-fluoroalkyl substances (PFAS) in processed foods from FDA's Total Diet Study. *Anal. Bioanal. Chem.* **2022**, *414* (3), 1189−1199.
(48) Young, W.; Wiggins, S.; Limm, W.; Fisher, C. M.; DeJager, L.; Genualdi, S. Analysis of Per- and Poly(fluoroalkyl) Substances

https://doi.org/10.1021/acs.est.4c11872
Environ. Sci. Technol. 2025, 59, 9896−9906

(PFASs) in Highly Consumed Seafood Products from U.S. Markets. *J. Agric. Food Chem.* **2022**, *70* (42), 13545−13553.
(49) Lin, P. I. D.; Cardenas, A.; Hauser, R.; Gold, D. R.; Kleinman, K. P.; Hivert, M. F.; Fleisch, A. F.; Calafat, A. M.; Sanchez-Guerra, M.; Osorio-Yáñez, C.; Webster, T. F.; Horton, E. S.; Oken, E. Dietary characteristics associated with plasma concentrations of per- and polyfluoroalkyl substances among adults with pre-diabetes: Cross-sectional results from the Diabetes Prevention Program Trial. *Environ. Int.* **2020**, *137*, No. 105217.
(50) Christensen, K. Y.; Raymond, M.; Blackowicz, M.; Liu, Y.; Thompson, B. A.; Anderson, H. A.; Turyk, M. Perfluoroalkyl substances and fish consumption. *Environ. Res.* **2017**, *154*, 145−151.
(51) Susmann, H. P.; Schaider, L. A.; Rodgers, K. M.; Rudel, R. A. Dietary Habits Related to Food Packaging and Population Exposure to PFASs. *Environ. Health Perspect.* **2019**, *127* (10), No. 107003.
(52) Boronow, K. E.; Brody, J. G.; Schaider, L. A.; Peaslee, G. F.; Havas, L.; Cohn, B. A. Serum concentrations of PFASs and exposure-related behaviors in African American and non-Hispanic white women. *J. Expo. Sci. Environ. Epidemiol.* **2019**, *29* (2), 206−217.
(53) Eick, S. M.; Goin, D. E.; Trowbridge, J.; Cushing, L.; Smith, S. C.; Park, J. S.; DeMicco, E.; Padula, A. M.; Woodruff, T. J.; Morello-Frosch, R. Dietary predictors of prenatal per- and poly-fluoroalkyl substances exposure. *J. Exposure Sci. Environ. Epidemiol.* **2023**, *33*, 32−39.
(54) Wang, Y.; Gui, J.; Howe, C. G.; Emond, J. A.; Criswell, R. L.; Gallagher, L. G.; Huset, C. A.; Peterson, L. A.; Botelho, J. C.; Calafat, A. M.; Christensen, B.; Karagas, M. R.; Romano, M. E. Association of diet with per- and polyfluoroalkyl substances in plasma and human milk in the New Hampshire Birth Cohort Study. *Sci. Total Environ.* **2024**, *933*, No. 173157.
(55) Sunderland, E. M.; Hu, X. C.; Dassuncao, C.; Tokranov, A. K.; Wagner, C. C.; Allen, J. G. 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), 131−147.
(56) Long, R.; Abington, L.; Manani, P.; van 't Erve, T. J.; Che, X.; Gray, J.; Wasilevich, E. *Drinking Water PFAS Concentrations and Exposure Factors Influencing Measured and Predicted Serum PFAS Concentrations: Report 2 of the North Kent County Exposure Assessment*; Michigan Department of Health and Human Services, 2024. https://www.michigan.gov/mdhhs/safety-injury-prev/environmental-health/topics/dehbio/nkcea/results.
(57) McDonough, C. A.; Choyke, S.; Barton, K. E.; Mass, S.; Starling, A. P.; Adgate, J. L.; Higgins, C. P. Unsaturated PFOS and other PFASs in human serum and drinking water from an AFFF-impacted community. *Environ. Sci. Technol.* **2021**, *55* (12), 8139−8148.
(58) Hu, X. C.; Tokranov, A. K.; Liddie, J.; Zhang, X.; Grandjean, P.; Hart, J. E.; Laden, F.; Sun, Q.; Yeung, L. W. Y.; Sunderland, E. M. Tap Water Contributions to Plasma Concentrations of Poly- and Perfluoroalkyl Substances (PFAS) in a Nationwide Prospective Cohort of U.S. Women. *Environ. Health Perspect.* **2019**, *127* (6), No. 067006.
(59) Hurley, S.; Houtz, E.; Goldberg, D.; Wang, M.; Park, J. S.; Nelson, D. O.; Reynolds, P.; Bernstein, L.; Anton-Culver, H.; Horn-Ross, P.; Petreas, M. Preliminary Associations between the Detection of Perfluoroalkyl Acids (PFAAs) in Drinking Water and Serum Concentrations in a Sample of California Women. *Environ. Sci. Technol. Lett.* **2016**, *3* (7), 264−269.
(60) USEPA. *Occurrence Data for the Unregulated Contaminant Monitoring Rule: UCMR 3 (2013−2015)*; 2017. https://www.epa.gov/dwucmr/occurrence-data-unregulated-contaminant-monitoring-rule.
(61) Hu, X. C.; Andrews, D. Q.; Lindstrom, A. B.; Bruton, T. A.; Schaider, L. A.; Grandjean, P.; Lohmann, R.; Carignan, C. C.; Blum, A.; Balan, S. A.; Higgins, C. P.; Sunderland, E. M. Detection of Poly- and Perfluoroalkyl Substances (PFASs) in U.S. Drinking Water Linked to Industrial Sites, Military Fire Training Areas, and Wastewater Treatment Plants. *Environ. Sci. Technol. Lett.* **2016**, *3* (10), 344−350.
(62) Smalling, K. L.; Romanok, K. M.; Bradley, P. M.; Morriss, M. C.; Gray, J. L.; Kanagy, L. K.; Gordon, S. E.; Williams, B. M.; Breitmeyer, S. E.; Jones, D. K.; DeCicco, L. A.; Eagles-Smith, C. A.; Wagner, T. Per- and polyfluoroalkyl substances (PFAS) in United States tapwater: Comparison of underserved private-well and public-supply exposures and associated health implications. *Environ. Int.* **2023**, *178*, No. 108033.
(63) Romanok, K.; Smalling, K.; Bradley, P. M.; Hladik, M. L.; McCleskey, R. B.; Gray, J. L.; Kanagy, L. K. *Concentrations of organic and inorganic constituents in tapwater samples from California in 2020−21 (ver.3.1, March 2024)*. United States Geological Survey, 2024.
(64) Bartell, S. M.; Calafat, A. M.; Lyu, C.; Kato, K.; Ryan, P. B.; Steenland, K. Rate of decline in serum PFOA concentrations after granular activated carbon filtration at two public water systems in Ohio and West Virginia. *Environ. Health Perspect.* **2010**, *118* (2), 222−228.
(65) Attfield, K.; Kauffman, D.; D'Amico, A.; Dobraca, D.; Andrews, F.; Wu, N. *The California Regional Exposure (CARE) Study, 2018 − 2020*. Department of Public Health Environmental Health Investigations Branch Biomonitoring California: Richmond, CA, 2023. https://biomonitoring.ca.gov/sites/default/files/downloads/CARE_Report.pdf.
(66) Wang, M.; Park, J. S.; Petreas, M. Temporal Changes in the Levels of Perfluorinated Compounds in California Women's Serum over the Past 50 Years. *Environ. Sci. Technol.* **2011**, *45* (17), 7510−7516.
(67) Wu, X. M.; Bennett, D. H.; Calafat, A. M.; Kato, K.; Strynar, M.; Andersen, E.; Moran, R. E.; Tancredi, D. J.; Tulve, N. S.; Hertz-Picciotto, I. Serum concentrations of perfluorinated compounds (PFC) among selected populations of children and adults in California. *Environ. Res.* **2015**, *136*, 264−273.
(68) Fromme, H.; Schlummer, M.; Möller, A.; Gruber, L.; Wolz, G.; Ungewiss, J.; Böhmer, S.; Dekant, W.; Mayer, R.; Liebl, B.; Twardella, D. Exposure of an adult population to perfluorinated substances using duplicate diet portions and biomonitoring data. *Environ. Sci. Technol.* **2007**, *41* (22), 7928−7933.
(69) Tittlemier, S. A.; Pepper, K.; Seymour, C.; Moisey, J.; Bronson, R.; Cao, X. L.; Dabeka, R. W. Dietary exposure of Canadians to perfluorinated carboxylates and perfluorooctane sulfonate via consumption of meat, fish, fast foods, and food items prepared in their packaging. *J. Agric. Food Chem.* **2007**, *55* (8), 3203−3210.
(70) Jain, R. B. Contribution of diet and other factors to the levels of selected polyfluorinated compounds: Data from NHANES 2003−2008. *Int. J. Hyg. Environ. Health.* **2014**, *217* (1), 52−61.
(71) McAdam, J.; Bell, E. M. Determinants of maternal and neonatal PFAS concentrations: a review. *Environ. Health.* **2023**, *22* (1), No. 41.
(72) Kim, H.; Ekpe, O. D.; Lee, J. H.; Kim, D. H.; Oh, J. E. Field-scale evaluation of the uptake of Perfluoroalkyl substances from soil by rice in paddy fields in South Korea. *Sci. Total Environ.* **2019**, *671*, 714−721.
(73) Schwartz-Narbonne, H.; Xia, C.; Shalin, A.; Whitehead, H. D.; Yang, D.; Peaslee, G. F.; Wang, Z.; Wu, Y.; Peng, H.; Blum, A.; Venier, M.; Diamond, M. L. Per- and Polyfluoroalkyl Substances in Canadian Fast Food Packaging. *Environ. Sci. Technol. Lett.* **2023**, *10* (4), 343−349.
(74) *Division of Drinking Water California Water Resources Control Board Tracking California*. Public Health Institute Water Boundary Tool. 2019. https://gispublic.waterboards.ca.gov/portalserver/rest/services/Drinking_Water/California_Drinking_Water_System_Area_Boundaries/FeatureServer/0.
(75) EPA United States Environmental Protection Agency Office of Water. *UCMR 3 Fact Sheet for Screening Survey Monitoring of List 2 Contaminants*; 2012. http://water.epa.gov/drink/.
(76) CA State Water Resources Control Board. *CA PFAS Timeline*. (accessed 2025−02−10). https://www.waterboards.ca.gov/pfas/ca_pfas_timeline.html.

https://doi.org/10.1021/acs.est.4c11872
Environ. Sci. Technol. 2025, 59, 9896−9906

(77) Mukhopadhyay, P. *Survey Data Imputation with PROC SURVEYIMPUTE* SAS Institute Inc. 2016.
(78) Glass, T. A.; Goodman, S. N.; Hernán, M. A.; Samet, J. M. Causal Inference in Public Health. *Annu. Rev. Public Health.* **2013**, *34*, 61−75.
(79) Greenland, S.; Pearl, J.; Robins, J. M. Causal diagrams for epidemiologic research. *Epidemiology.* **1999**, *10* (1), 37−48.
(80) Fraser, A. J.; Webster, T. F.; Watkins, D. J.; Nelson, J. W.; Stapleton, H. M.; Calafat, A. M.; Kato, K.; Shoeib, M.; Vieira, V. M.; McClean, M. D. Polyfluorinated compounds in serum linked to indoor air in office environments. *Environ. Sci. Technol.* **2012**, *46* (2), 1209−1215.
(81) Raffensperger, S.; Kuczmarski, M. F.; Hotchkiss, L.; Cotugna, N.; Evans, M. K.; Zonderman, A. B. Effect of race and predictors of socioeconomic status on diet quality in the HANDLS Study sample. *J. Natl. Med. Assoc.* **2010**, *102* (10), 923−930.
(82) Marvin-Dowle, K.; Kilner, K.; Burley, V.; Soltani, H.; Rota, M. Differences in dietary pattern by maternal age in the Born in Bradford cohort: A comparative analysis. *PloS One* **2018**, *13* (12), No. e0208879.
(83) Crozier, S. R.; Robinson, S. M.; Godfrey, K. M.; Cooper, C.; Inskip, H. M. Women's Dietary Patterns Change Little from Before to During Pregnancy. *J. Nutr.* **2009**, *139* (10), 1956−1963.
(84) Thorpe, M. G.; Milte, C. M.; Crawford, D.; McNaughton, S. A. Education and lifestyle predict change in dietary patterns and diet quality of adults 55 years and over. *Nutr. J.* **2019**, *18* (1), No. 67.
(85) Zhou, W.; Zhao, S.; Tong, C.; Chen, L.; Yu, X.; Yuan, T.; Aimuzi, R.; Luo, F.; Tian, Y.; Zhang, J. Dietary intake, drinking water ingestion and plasma perfluoroalkyl substances concentration in reproductive aged Chinese women. *Environ. Int.* **2019**, *127*, 487−494.
(86) USDA Nutrient Intakes from Food and Beverages: Mean Amounts Consumed per Individual, by Gender and Age, *What We Eat in America*, NHANES 2015−2016.
(87) Calafat, A. M.; Needham, L. L.; Kuklenyik, Z.; Reidy, J. A.; Tully, J. S.; Aguilar-Villalobos, M.; Naeher, L. P. Perfluorinated chemicals in selected residents of the American continent. *Chemosphere.* **2006**, *63* (3), 490−496.
(88) Sagiv, S. K.; Rifas-Shiman, S. L.; Webster, T. F.; Mora, A. M.; Harris, M. H.; Calafat, A. M.; Ye, X.; Gillman, M. W.; Oken, E. Sociodemographic and Perinatal Predictors of Early Pregnancy Per- and Polyfluoroalkyl Substance (PFAS) Concentrations. *Environ. Sci. Technol.* **2015**, *49* (19), 11849−11858.
(89) Park, S. K.; Peng, Q.; Ding, N.; Mukherjee, B.; Harlow, S. D. Determinants of per- and polyfluoroalkyl substances (PFAS) in midlife women: Evidence of racial/ethnic and geographic differences in PFAS exposure. *Environ. Res.* **2019**, *175*, 186−199.
(90) Liddie, J. M.; Schaider, L. A.; Sunderland, E. M. Sociodemographic Factors Are Associated with the Abundance of PFAS Sources and Detection in U.S. Community Water Systems. *Environ. Sci. Technol.* **2023**, *57* (21), 7902−7912.
(91) Mueller, R.; Salvatore, D.; Brown, P.; Cordner, A. Quantifying Disparities in Per- and Polyfluoroalkyl Substances (PFAS) Levels in Drinking Water from Overburdened Communities in New Jersey, 2019−2021. *Environ. Health Perspect.* **2024**, *132* (4), No. 047011.
(92) Yohai, V. J. High Breakdown-Point and High Efficiency Robust Estimates for Regression. *Ann. Stat.* **1987**, *15* (2), 642−656.
(93) Hampel, F. R.; Ronchetti, E. M.; Rousseeuw, P. J.; Stahel, W. A. Linear Models: Robust Estimation. In *Robust Statistics: the approach based on influence functions*. Wiley Series in Probability and Statistics; Wiley 2005, 307−341.
(94) Jana, P.; Rosadi, D.; Supandi, E. D. Comparison of robust estimation on multiple regression model. *BAREKENG: J. Math. & App.* **2023**, *17* (2), 0979−0988.
(95) Fox, J.; Monette, G. Generalized Collinearity Diagnostics. *J. Am. Stat. Assoc.* **1992**, *87* (417), 178−183.
(96) Centers for Disease Control and Prevention (CDC) *National Health and Nutrition Examination Survey Data 2017−2018*. National Center for Health Statistics (NCHS). 2020. https://wwwn.cdc.gov/nchs/nhanes/continuousnhanes/default.aspx?BeginYear=2017.
(97) Criswell, R. L.; Simones, T.; Chatterjee, M.; Waite, J.; Diaz, S.; Smith, A. Quantifying levels of per- and polyfluoroalkyl substances (PFAS) in water and serum after contamination from agricultural biosolid application. *Environ. Int.* **2024**, *190*, No. 108850.
(98) Johanson, G.; Gyllenhammar, I.; Ekstrand, C.; Pyko, A.; Xu, Y.; Li, Y.; Norström, K.; Lilja, K.; Lindh, C.; Benskin, J. P.; Georgelis, A.; Forsell, K.; Jakobsson, K.; Glynn, A.; Vogs, C. Quantitative relationships of perfluoroalkyl acids in drinking water associated with serum concentrations above background in adults living near contamination hotspots in Sweden. *Environ. Res.* **2023**, *219*, No. 115024.
(99) U.S. FDA. *Analytical Results of Testing Food for PFAS from Environmental Contamination*. U.S. FDA. 2024. (accessed 2024−07−20). https://www.fda.gov/food/process-contaminants-food/analytical-results-testing-food-pfas-environmental-contamination.
(100) U.S. Food and Drug Administration *Indirect Food Additives: Paper and Paperboard Components*. Rules and Regulations. Office of the Federal Register, National Archives and Records Administration; 2016, 83623−84387. (accessed 2024−03−08). https://www.govinfo.gov/app/details/FR-2016-11-22/2016-28116.
(101) Li, Y.; Fletcher, T.; Mucs, D.; Scott, K.; Lindh, C. H.; Tallving, P.; Jakobsson, K. Half-lives of PFOS, PFHxS and PFOA after end of exposure to contaminated drinking water. *J. Occup. Environ. Med.* **2018**, *75* (1), 46−51.
(102) Olsen, G. W.; Burris, J. M.; Ehresman, D. J.; Froehlich, J. W.; Seacat, A. M.; Butenhoff, J. L.; Zobel, L. R. Half-life of serum elimination of perfluorooctanesulfonate,perfluorohexanesulfonate, and perfluorooctanoate in retired fluorochemical production workers. *Environ. Health Perspect.* **2007**, *115* (9), 1298−1305.
(103) Thompson, F. E.; Kirkpatrick, S. I.; Subar, A. F.; Reedy, J.; Schap, T. E.; Wilson, M. M.; Krebs-Smith, S. M. The National Cancer Institute's Dietary Assessment Primer: A Resource for Diet Research. *J. Acad. Nutr. Diet.* **2015**, *115* (12), 1986−1995.
(104) Haug, L. S.; Huber, S.; Schlabach, M.; Becher, G.; Thomsen, C. Investigation on Per- and Polyfluorinated Compounds in Paired Samples of House Dust and Indoor Air from Norwegian Homes. *Environ. Sci. Technol.* **2011**, *45* (19), 7991−7998.
(105) Zheng, G.; Eick, S. M.; Salamova, A. Elevated Levels of Ultrashort- and Short-Chain Perfluoroalkyl Acids in US Homes and People. *Environ. Sci. Technol.* **2023**, *57* (42), 15782−15793.
(106) Phelps, D. W.; Parkinson, L. V.; Boucher, J. M.; Muncke, J.; Geueke, B. Per- and Polyfluoroalkyl Substances in Food Packaging: Migration, Toxicity, and Management Strategies. *Environ. Sci. Technol.* **2024**, *58*, 5670−5684.

https://doi.org/10.1021/acs.est.4c11872
Environ. Sci. Technol. 2025, 59, 9896−9906