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Jiahui Guo
Pennsylvania State University
Ruohao Zhang
rjz5261@psu.edu

Pennsylvania State University
Jiameng Zheng
Louisiana State University
Yongwang Ren
Kansas State University
Xibo Wan
North Dakota State University
Wendong Zhang
Cornell University

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Jiahui Guo[a], Ruohao Zhang[a,*], Jiameng Zheng[b], Yongwang Ren[c], Xibo Wan[d], Wendong Zhang[e]

[a] *Department of Agricultural Economics, Sociology, and Education, The Pennsylvania State University, University Park, PA, USA*

[b] *Department of Agricultural Economics and Agribusiness, Louisiana State University, Baton Rouge, LA, USA*

[c] *Department of Agricultural Economics, Kansas State University, Manhattan, KS, USA*

[d] *Department of Agribusiness and Applied Economics, North Dakota State University, Fargo, ND, USA*

[e] *Charles H. Dyson School of Applied Economics and Management, Cornell University, Ithaca, NY, USA*

Abstract

Per- and polyfluoroalkyl substances (PFAS) are persistent contaminants widely detected in food, yet nationwide variation in dietary exposure across the United States is poorly characterized. This study estimates household-level dietary PFAS exposure by linking NielsenIQ grocery purchases (138,075 household-year observations, 2021–2023) to a harmonized database of PFAS concentrations compiled from eight studies and seven federal Total Diet Study datasets, producing exposure estimates with a geometric mean of approximately 2,300 ng. Estimates are higher among Northeastern, older, and Black/African American households, while income-group differences are small after conditioning on per-capita spending. Oaxaca–Blinder decomposition separating grocery spending from dietary composition indicates that composition accounts for most regional and age-group gaps, while both components matter across racial and ethnic comparisons. Butter, potatoes, cooking oils, and seafood contribute substantially to estimated exposure. By linking food-testing literature to household purchase data, this study offers a scalable framework for nationwide assessment of dietary exposure to other contaminants.

Keywords: PFAS; dietary exposure; food contamination; socioeconomic disparities; household consumption

* Corresponding author: Ruohao Zhang: rjz5261@psu.edu; Address: 104 Ferguson Building, University Park, PA 16802, USA.

Introduction

Per- and polyfluoroalkyl substances (PFAS) are a family of synthetic chemicals comprising over 4,700 uniquely identified fluorinated organic compounds that have been synthetically manufactured since the end of the 1940s [1]. The chemical and thermal stability of PFAS results in highly useful and enduring properties, including resistance to heat, oils, stains, grease, and water [2]. PFAS have been used in a wide array of industrial and consumer products, such as textiles, paper coating, food packaging, aqueous film-forming foam (AFFF), and metal plating [3]. However, this stability also makes PFAS resistant to degradation, leading to environmental persistence and accumulation [2, 4]. As of February 2026, the widespread presence of PFAS has contaminated over 3,539 locations in 49 states across the U.S [5]. The health impacts of exposure to PFAS are increasingly recognized. Literature has found associations between PFAS exposure and multiple adverse health outcomes, including but not limited to, hypercholesterolemia [6], kidney and testicular cancer [7], immunotoxicity [8], and more. The existing literature identifies various pathways of humans' exposure to PFAS [4, 9, 10], among which diet and drinking water are generally considered the dominant contributors to chronic PFAS exposure in the general population [11].

In April 2024, the U.S. Environmental Protection Agency (EPA) established the first national drinking-water standards for six PFAS. In May 2026, the EPA proposed upholding the standards for perfluorooctanoic acid (PFOA) and perfluorooctane sulfonate (PFOS) while extending their compliance deadline to 2031, and separately proposed rescinding the regulations for four other PFAS [12, 13]. These proposed changes create uncertainty about the future scope and timing of federal drinking-water protections. At the same time, the Federal Food and Drug Administration (FDA) states that PFAS enter the food supply through crops and animals grown, raised, or processed in contaminated areas [14]. However, there are no existing regulations on food PFAS contamination.

Together, these scientific findings and policy development underscore the importance of understanding foodborne PFAS exposure: regardless of uncertainties in drinking-water regulation, food remains an important and distinct pathway for exposure assessment and public-health policy. In addition, there is limited understanding of the geographical and socioeconomic patterns of PFAS exposure and potential social disparities. Therefore, it is important to characterize variation in dietary exposure across populations, particularly since vulnerable and socially disadvantaged populations may face disproportionate exposure.

Literature has detected PFAS in a wide range of foods, including seafood, meat, dairy products, and other grocery items, and concentrations vary substantially across food categories [9, 15, 16, 17]. Seafood, in particular, is frequently identified as a major contributor to dietary PFAS exposure because of its relatively high measured concentrations and its importance in some diets [11, 18, 19]. Because contamination varies across food categories, dietary behavior plays an important role in

shaping variation in exposure across demographic groups [20, 21, 22]. Prior studies have examined dietary PFAS exposure using different approaches. Some use small, geographically constrained samples to link dietary survey data with serum PFAS measurements and examine associations between food intake and biomonitored body burdens [23, 24, 25]. Others combine PFAS measurements in selected food items with population-level consumption statistics to estimate dietary exposure [9, 26, 11]. A third approach uses duplicate-diet studies, which collect and analyze foods actually consumed by sampled individuals, providing detailed but typically small-scale exposure assessments [27]. Across these approaches, the literature consistently reports socioeconomic disparities: PFAS exposure tends to be higher among individuals with greater meat and seafood consumption and lower among those with diets richer in fruits, vegetables, and other plant-based foods [28, 29]; and systematic variations exist across demographic and socioeconomic groups, including vulnerable populations such as pregnant women and infants, potentially reflecting differences in diet, food access, proximity to contamination sources, and drinking water conditions [21, 30, 31, 32, 33, 34].

Given that existing measurements of household dietary PFAS exposure in the United States often rely on blood tests at a relatively small scale and have not been systematically linked to household demographics, dietary patterns and food purchase behaviors [23, 25], important research gaps therefore remain in providing a comprehensive, large-scale assessment of how foodborne PFAS exposure is distributed across socioeconomic groups nationwide, mainly because PFAS measurements in food are limited, outdated, and fragmented across individual studies, often missing for the wide range of products on today's commercial market [16, 35]. This paper helps fill that gap by constructing a household-level measure of annual PFAS exposure through grocery purchases and examining how it varies across geographic regions and socioeconomic groups in the contiguous United States. To do so, we combine detailed household grocery purchase data from the NielsenIQ Consumer Panel, which covers 138,075 household-year observations nationwide with rich demographic information, with PFAS concentration measurements compiled and harmonized from the recent literature on PFAS contamination in foods in the U.S. since 2010. We further decompose variation in PFAS exposure into contributions from annual per-capita grocery spending and dietary composition. We estimate that the geometric mean of estimated per capita annual PFAS exposure between 2021 and 2023 was approximately 2,300 ng. Substantial heterogeneity emerges across regions, race and ethnicity, income, and age. Older households tend to experience higher average PFAS exposure. Income-group differences remain small after conditioning on annual per-capita grocery spending. Oaxaca-Blinder decomposition shows that dietary composition accounts for most regional and age-group gaps, while both spending and composition are important across racial and ethnic comparisons. Butter, potatoes, edible oils, and seafood account for a large share of exposure variation.

This study makes four contributions to the literature. First, to our knowledge, it is the first large-

scale study to use observed household grocery purchases to characterize variation in estimated dietary PFAS exposure across regions and socioeconomic groups in the continental United States. Second, we compile and harmonize PFAS concentration measurements in foods from contemporary U.S.-based studies conducted after 2010, helping to address the data gap in the measurement of food PFAS contamination under more recent regulatory and environmental conditions. Third, by linking food contamination data to observed household grocery purchases, we are able to capture heterogeneity in dietary composition and grocery spending patterns and examine how those differences shape socioeconomic disparities. Fourth, our method is not specific to PFAS: by pairing contaminant measurements synthesized from the food-testing literature with household food purchase data, it provides a scalable approach for conducting large-scale, nationwide assessments of dietary exposure risk for other foodborne contaminants.

Our findings have important policy implications. Given the substantial influence of dietary choices on PFAS exposure, policies aimed at reducing PFAS exposure risks may benefit from targeting food categories with high PFAS concentrations and enhancing information transparency about contamination risks to encourage risk-avoidance behaviors, especially among vulnerable populations. Our analysis contributes to this policy discussion by identifying how foodborne PFAS exposure varies across regions and socioeconomic groups and documenting differences associated with grocery spending and dietary preferences. More broadly, our study demonstrates how scanner data can be leveraged to incorporate consumption behavior into environmental exposure assessment, helping policymakers design interventions that better address unequal exposure burdens and promote more equitable environmental health outcomes.

Results

Study sample and representativeness

Table 1 reports summary statistics for the sample of 138,075 observations and benchmarks them against the nationwide representative sample from the American Community Survey (ACS) on key socioeconomic characteristics. Average annual per-capita grocery spending is $2,595, and a household records an average of 11.64 shopping trips per month. The regional distribution of the sample is broadly similar to that of the ACS, with relatively over representation of the North Central region and slight underrepresentation of the West. Across other socioeconomic dimensions, the sample underrepresents younger households, households with children, and households in the highest income group, as well as those with post-graduate degrees, while overrepresenting non-Hispanic White and married households. The employment percentages should not be compared directly across the two

datasets because their definitions differ.[1] Although our sample may not be perfectly representative, within-sample comparisons across demographic groups remain informative, as the panel retains sufficient geographic and socioeconomic variation for studying heterogeneity in PFAS exposure.

National magnitude and distribution of dietary PFAS exposure

Here and below, estimated dietary PFAS exposure refers to the annual per-capita mass of PFOA and PFOS, the two most commonly detected PFAS compounds and those of greatest health concern, associated with observed household grocery purchases, expressed in ng per person-year; full details of its construction are provided in the Methods. Table 2 summarizes the annual distribution of the grocery-based exposure measure from 2021 to 2023. Its geometric mean ranges from 2,318 to 2,368 ng; both the arithmetic mean and median also show only modest year-to-year variation. [2] This temporal stability suggests that the estimated exposure distribution is fairly persistent across the study period. Excluding the most highly contaminated products such as walleye, whitefish, and yellow perch changes the annual geometric means by no more than 0.4%, indicating that the central exposure estimates are not sensitive to these high-concentration fish species (Supplementary Table A4).

At the same time, the distribution of per-capita PFAS exposure is strongly right-skewed. In each year, the mean substantially exceeds both the geometric mean and the median, indicating that a small subset of households experiences disproportionately high PFAS exposure. The wide interquartile range and extremely large upper-tail values further highlight substantial cross-household heterogeneity. Taken together, these patterns suggest that variation in PFAS exposure is driven less by year-to-year changes in the aggregate distribution and more by persistent differences across households.

Geographic and demographic heterogeneity

Figure 1 maps the state-level geometric mean of household per-capita PFAS exposure, with bubble size proportional to the number of sample households in each state. State geometric means range from 1,521 ng in California to 2,985 ng in Vermont. California and Texas, which have among the largest household samples, are located at the lower end of the distribution, whereas relatively high means are observed in northern New England states and Idaho.[3] A clear regional pattern emerges,

[1] NielsenIQ classifies household-head employment configurations rather than individual labor-force status. Its "Unemployment" category includes households with at least one head not employed for pay and may therefore include retirees and others outside the labor force; see Supplementary Table A3.

[2] Zero-exposure observations are excluded from this and all subsequent geometric-mean calculations.

[3] California shows lower consumption of butter, olive oil, and potatoes, which have relatively high PFAS concentrations and baseline consumption levels. The product-level results are presented in Supplementary Figure A5.

Table 1: Summary statistics and comparison with the American Community Survey (ACS)

Variables	Sample Group	Summary	ACS Group	Summary
Region	Mid Atlantic	24,613 (17.8%)	Mid Atlantic	52,533,943 (15.7%)
	North Central	28,587 (20.7%)	North Central	57,298,443 (17.1%)
	Northeast	24,194 (17.5%)	Northeast	57,414,970 (17.1%)
	Southeast	23,283 (16.9%)	Southeast	54,347,608 (16.2%)
	Southwest	20,560 (14.9%)	Southwest	56,155,274 (16.8%)
	West	16,838 (12.2%)	West	57,172,261 (17.1%)
Age	<30	1,112 (0.8%)	<45	73,699,646 (34.8%)
	30-34	3,096 (2.2%)		
	35-39	6,526 (4.7%)		
	40-44	9,587 (6.9%)		
	45-49	10,855 (7.9%)	45-64	75,628,165 (35.7%)
	50-54	14,215 (10.3%)		
	55-64	37,829 (27.4%)		
	>65	54,855 (39.7%)	65+	62,426,367 (29.5%)
Household income	<29,999	23,750 (17.2%)	<49,999	40,533,490 (31.4%)
	30,000-49,999	28,215 (20.4%)		
	50,000-69,999	23,473 (17.0%)	50,000-99,999	36,351,482 (28.1%)
	70,000-99,999	30,402 (22.0%)		
	>100,000	32,235 (23.3%)	100,000+	52,342,524 (40.5%)
Education	< High school	23,821 (17.3%)	< High school	23,992,410 (10.4%)
	High school-college	86,639 (62.7%)	High school-college	124,451,677 (53.9%)
	Post College	27,615 (20.0%)	Post College	82,363,216 (35.7%)
Race ethnicity	Asian	5,930 (4.3%)	Asian	6,861,207 (5.3%)
	Black/African American	16,254 (11.8%)	Black/African American	15,641,707 (12.1%)
	Hispanic White	5,870 (4.3%)	Hispanic White	3,710,781 (2.9%)
	Non-Hispanic White	103,249 (74.8%)	Non-Hispanic White	82,526,682 (63.9%)
	Other	6,772 (4.9%)	Other	20,487,119 (15.9%)
Children18	No	113,421 (82.1%)	No	95,172,074 (73.6%)
	Yes	24,654 (17.9%)	Yes	34,055,422 (26.4%)
Marital status	No	53,067 (38.4%)	No	59,301,707 (45.9%)
	Yes	85,008 (61.6%)	Yes	69,925,789 (54.1%)
Employment	Unemployment	71,704 (51.9%)	Unemployed	8,902,365 (3.3%)
	Part-time	45,101 (32.7%)	Employed	161,297,155 (60.0%)
	Full-time	21,270 (15.4%)		
			Not in labor force	98,688,351 (36.7%)
WIC	No	125,575 (90.9%)		
	Yes	12,500 (9.1%)		
Annual per-capita grocery spending (USD)		2,595.39 [1,542.02]		
Trip count		11.64 [7.39]		
N		138,075		

Notes: The table reports summary statistics for the study sample and corresponding population benchmarks from the American Community Survey (ACS). For demographic variables, entries report the number of households, with percentages in parentheses. For annual per-capita grocery spending and average monthly trip count, entries report sample means, with standard deviations reported in square brackets. Categories are harmonized across the two data sources to ensure comparability; in particular, finer sample categories are aggregated to match broader ACS group definitions where necessary. Some variables are not directly comparable across data sources. In particular, employment status, education, and region in the ACS are measured at the population level rather than the household level, as comparable household-level information is not available. These differences may contribute to residual discrepancies between the sample and ACS distributions. Annual per-capita grocery spending is calculated as annual household grocery spending divided by household size excluding children under 12 and is reported in U.S. dollars.

Table 2: PFAS exposure per capita, summarized by year (ng)

Year	Geometric Mean	Mean	SD	Min	P25	Median	P75	Max
2021	2317.96	3528.18	3363.08	0.00	1376.13	2690.16	4661.85	122928.24
2022	2368.47	3618.70	3439.83	0.00	1410.21	2749.00	4788.77	102099.80
2023	2334.08	3624.34	3512.59	0.00	1360.35	2726.57	4829.37	86787.95

Notes: The table summarizes the distribution of estimated PFAS exposure across households for each year. Because PFAS exposure is highly right-skewed, the geometric mean is reported alongside conventional summary statistics. P25 and P75 represent the 25th and 75th percentiles, respectively. All values are computed at the household–year level.

with consistently higher PFAS exposure along the Eastern seaboard, especially in the Northeast, whereas states in the West and Southwest tend to have lower exposure levels. Thus, although estimates for sparsely represented states should be interpreted cautiously, the overall geographic pattern does not appear to be driven solely by small state samples.

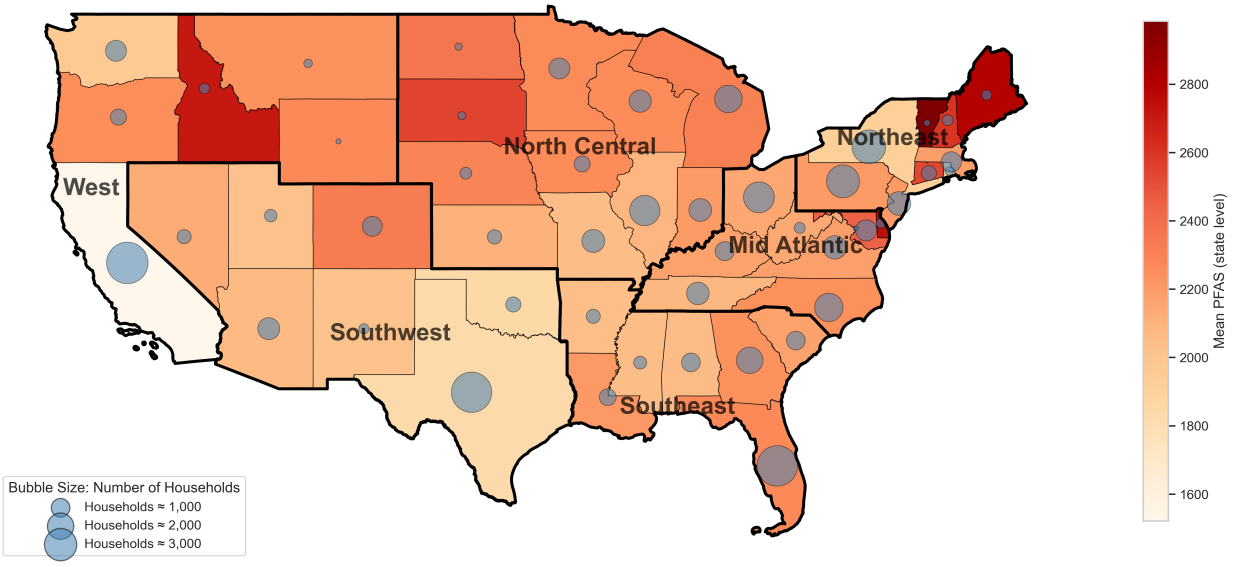
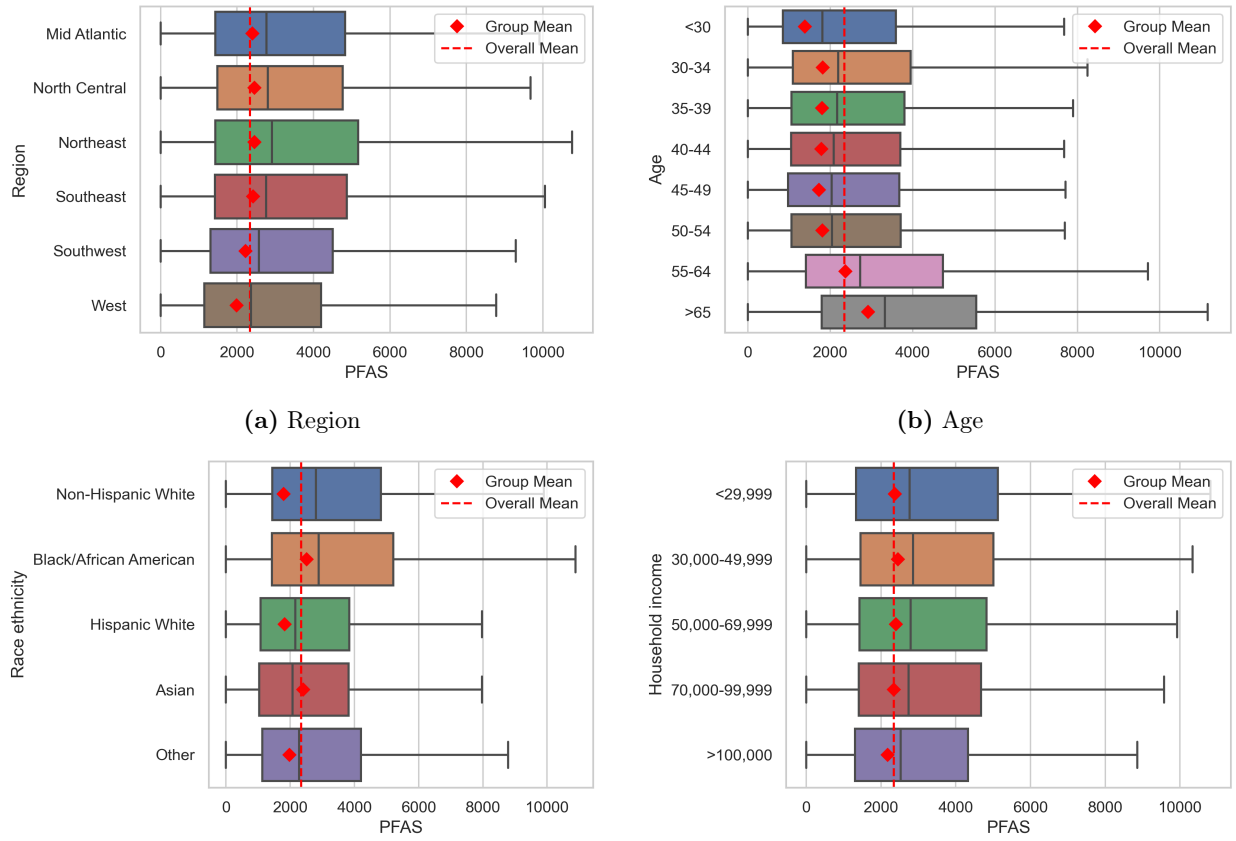


Figure 1: PFAS exposure (ng) and household coverage by U.S. state. The choropleth reports the projection-weighted geometric mean of annual household per-capita PFAS exposure in each state. Bubble area is proportional to the number of unique sampled households observed in that state during 2021–2023. Alaska and Hawaii are not displayed on the contiguous-state map.

Figure 2 presents the distribution of per-capita PFAS exposure across four key socioeconomic dimension groups. Each panel reports the within-group distribution together with the group geometric mean. The distributions of additional socioeconomic dimensions are shown in Supplementary Figure A3. Overall, PFAS exposure exhibits heterogeneity across socioeconomic groups. Regionally, households in the Northeast show the highest PFAS exposure, whereas those in the West show the lowest. PFAS exposure also varies by age group: younger households display lower median exposure,

whereas older age groups—particularly those aged 55 and above—exhibit higher average exposure and greater dispersion, consistent with prior evidence documenting age-related variations [30, 31]. Moreover, PFAS exposure differs across racial groups, with Black/African American households showing higher median levels. However, PFAS exposure appears relatively similar across income groups, with no clear monotonic gradient.



(a) Region

(b) Age

(c) Household race and ethnicity

(d) Household income

Figure 2: Distribution of PFAS exposure (ng) across demography. Boxes show the interquartile range, center lines show medians, and whiskers extend to 1.5 times the interquartile range; observations beyond the whiskers are not displayed. Red diamonds indicate unweighted group geometric means, and the red dashed line indicates the unweighted overall geometric mean. Additional distributions for other socioeconomic dimensions are reported in Supplementary Figure A3.

Multivariable associations

The descriptive evidence points to limited temporal variation but notable geographic and demographic heterogeneity in household per capita PFAS exposure. This motivates the following regression analysis, which examines more rigorously whether these descriptive differences persist after controlling for observed household characteristics and helps quantify the relative importance of demographic,

socioeconomic, and geographic correlations.

The regression results are reported in Table 3. Column (1) presents estimates without controlling for grocery spending, while Column (2) additionally includes annual per-capita grocery spending, measured in U.S. dollars. Per-capita grocery spending is strongly and positively associated with estimated PFAS exposure, and its inclusion increases the R^2 from 0.05 to 0.21. Nevertheless, substantial unexplained heterogeneity remains across households. Standard errors are clustered at the household level, and the coefficients are interpreted as conditional differences in group mean exposure rather than as causal effects or predictions of individual exposure.

Table 3: Regression results

	(1) Without spending		(2) With spending	
Variable	Estimate	SE	Estimate	SE
Region (reference: Mid Atlantic)				
North Central	-40.39	(43.24)	8.88	(39.59)
Northeast	233.70***	(49.41)	269.69***	(45.39)
Southeast	-42.09	(47.02)	-55.15	(42.97)
Southwest	-261.37***	(46.33)	-214.96***	(42.25)
West	-470.57***	(48.10)	-474.10***	(44.03)
Age (reference: < 30)				
30–34	448.93***	(125.17)	222.85**	(111.73)
35–39	484.54***	(119.56)	112.59	(106.41)
40–44	453.80***	(116.82)	156.69	(104.57)
45–49	306.58***	(116.32)	110.67	(103.97)
50–54	234.63**	(114.67)	65.40	(102.75)
55–64	846.58***	(113.80)	528.60***	(101.91)
> 65	1,354.54***	(115.99)	971.48***	(103.95)
Household income (reference: $<$ \$29,999)				
\$30,000–49,999	4.84	(45.32)	-27.71	(41.49)
\$50,000–69,999	51.07	(49.67)	-39.47	(45.31)
\$70,000–99,999	62.89	(46.79)	-20.08	(42.76)
$>$ \$100,000	31.01	(49.32)	-60.29	(44.95)
Race and ethnicity (reference: Asian)				
Black/African American	624.33***	(72.41)	576.75***	(64.60)
Hispanic White	-140.93*	(74.98)	-207.45***	(66.49)
Non-Hispanic White	179.52***	(59.56)	-9.51	(53.00)
Other	127.43	(78.68)	9.86	(70.07)
Marital status (reference: No)				
Yes	-384.69***	(33.93)	343.16***	(29.93)
Employment (reference: Full-time)				
Part-time	163.12***	(35.22)	6.39	(31.51)
Unemployment	394.92***	(34.98)	180.83***	(31.55)
Education (reference: less than high school)				
High school-college	19.26	(37.61)	-33.63	(34.27)
Post College	-103.72**	(46.36)	-158.48***	(42.45)
Children under 18 (reference: No)				
Yes	-531.37***	(29.43)	-65.95**	(25.99)
WIC (reference: No)				
Yes	46.88	(43.59)	-53.37	(39.01)

Continued on next page

Table 3 continued

	(1) Without spending		(2) With spending	
Variable	Estimate	SE	Estimate	SE
Annual per-capita grocery spending (USD)			0.95***	(0.01)
Observations	138,075		138,075	
R^2	0.05		0.21	

Notes: OLS estimates are for annual household per-capita PFOA+PFOS exposure (ng). Both models include the listed covariates and year fixed effects; Model 2 adds annual per-capita grocery spending (annual household spending divided by household size excluding children under 12), in U.S. dollars. Standard errors are clustered by household (61,532 clusters). State-level wild-cluster-bootstrap inference using 49 clusters is reported in Supplementary Table A5. *, **, and *** indicate significance at the 10%, 5%, and 1% levels.

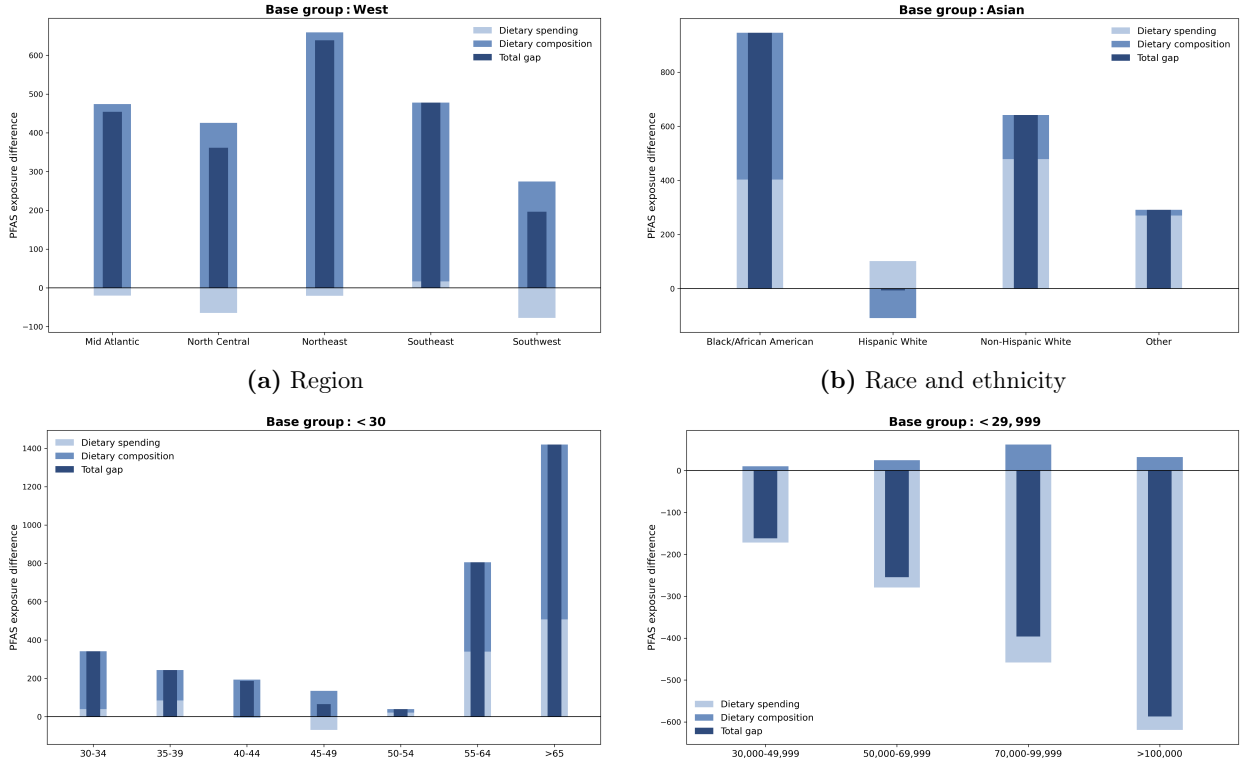
The regression results broadly corroborate the descriptive evidence, including the main geographic, older-age, and racial and ethnic patterns. After controlling for annual per-capita grocery spending, however, several income, age, and household-structure coefficients attenuate or change, indicating that part of the unconditional group differences is associated with spending levels. The remaining variation points to other factors and motivates the further decomposition analysis separating the contributions of grocery spending and dietary composition.

Decomposition of drivers of exposure heterogeneity

To understand what drives heterogeneity in dietary PFAS exposure across socioeconomic groups, two complementary decomposition approaches are conducted. First, an Oaxaca–Blinder decomposition separates the component associated with observed grocery spending from a residual component labeled dietary composition. Second, a product-level analysis decomposes estimated exposure into contributions from specific food and beverage categories.

Figure 3 presents the Oaxaca–Blinder decomposition results across geographic, demographic, and socioeconomic dimensions. In each panel, the dark blue bar reflects the total difference in PFAS exposure between the corresponding group and the indicated base group; light and medium blue bars show the grocery spending and residual dietary composition components. The relative contributions vary across comparisons rather than following one uniform pattern. The decomposition reveals a common pattern across regions and age groups: exposure gaps are driven primarily by the composition-related component rather than by per-capita spending. Across regions, spending differences are generally small or offsetting. Across age groups, spending accounts for a larger share of the gaps, but its contribution remains smaller than that of dietary composition. This pattern is less uniform across racial and ethnic groups, for which both components matter. Compared with the lowest-income group, higher-income households have lower estimated per-capita PFAS exposure. The decomposition attributes most of this difference to grocery spending: although high income households have higher total grocery spending, their per-capita spending is lower. In

addition, their small positive composition differences partially offset the difference. This may reflect greater food-away-from-home consumption among high income households, which the NIQ grocery data do not capture. Additional results are reported in Supplementary Figure A10. The following product-level analysis examines which purchased food categories underlie the composition-related differences.



(a) Region **(b)** Race and ethnicity

(c) Age **(d)** Household income

Figure 3: Oaxaca–Blinder decomposition by demographic characteristics. The figure presents decomposition results for differences in household per-capita PFAS exposure across geographic, demographic, and socioeconomic groups. Dark blue bars indicate the total exposure gap relative to the base group, while light and medium blue bars show the contributions of grocery spending and dietary composition, respectively.

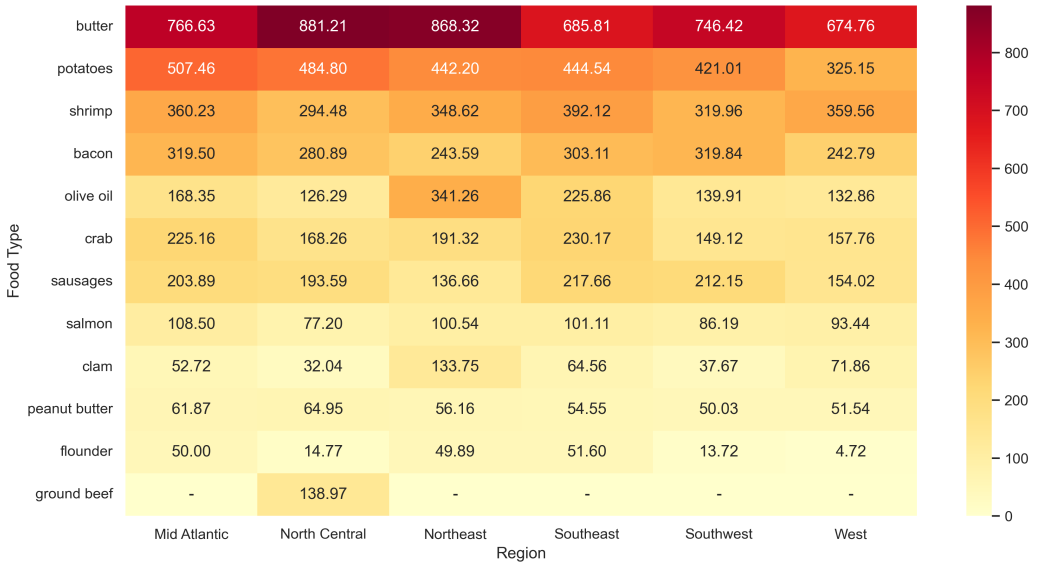
To better explain the mechanisms suggested by the Oaxaca–Blinder decomposition, we study the more detailed household purchase data at product-level to investigate which products account for differences in exposure patterns across populations and provides further insight into how dietary composition shapes PFAS exposure. Figure 4(a) shows the dietary composition of the top PFAS-contaminated products that account for most variation in PFAS exposure across regions. According to the results, butter, potatoes, and shrimp emerge as dominant contributors across regions. The Northeast exhibits the highest overall estimated PFAS exposure, largely driven by higher contributions from butter and olive oil, together with comparatively elevated seafood consumption. This pattern

is consistent with documented regional dietary differences in the United States, where Northeastern diets tend to include greater consumption of dairy products and seafood, reflecting both cultural preferences and access to coastal food markets. By contrast, households in the West show lower exposure from these products. These product-level patterns are consistent with the Oaxaca–Blinder result that dietary composition accounts for most regional exposure gaps. Figure 4(b) presents analogous results across racial and ethnic groups. Non-Hispanic White households exhibit the highest estimated PFAS exposure from butter and potatoes, which together account for a large share of the overall exposure gap across groups. Black/African American households draw a comparatively large share of their exposure from seafood and processed foods such as shrimp and bacon. Asian, Hispanic White, and other households display more balanced exposure across several products (except for Asian households' high exposure from shrimp), resulting in their lower overall exposure levels.

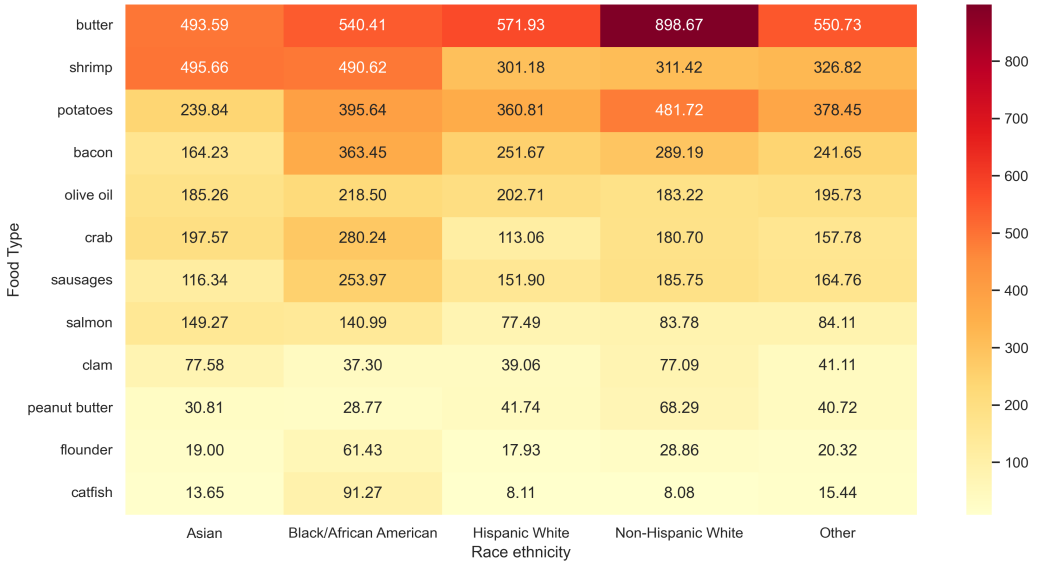
The Supplementary Information includes complete heatmaps for all PFAS-contaminated products with available PFAS contamination measurements (Supplementary Figures A4–A6), as well as consumption heatmaps across demographic groups (Supplementary Figures A7–A9). Supplementary Figure A4 shows that product-level contributions differ across age and income groups, but a limited number of PFAS-contaminated products account for a large share of estimated exposure in each group. These results align with the Oaxaca–Blinder decomposition above, showing that differences in grocery-basket composition are concentrated in a relatively small number of PFAS-intensive products.

Discussion

This nationwide purchase-based analysis identifies substantial geographic and demographic variation in estimated household PFAS exposure. The dietary composition dominates regional and age-group gaps, while both dietary composition and grocery spending are important across racial and ethnic comparisons. The large share of variation explained by dietary composition is consistent with findings from the biomonitoring literature. For example, a recent study of 160 rural New York residents found that shellfish and selected dairy products predicted higher serum PFAS concentrations [36]. A study of adults in California during a similar period found positive associations of drinking water, eggs, and seafood with adult serum PFAS concentrations [37]. Similarly, fish and packaged food consumption has been positively associated with children's serum PFAS concentrations [21]. In our analysis, butter, shrimp, and potatoes are among the largest contributors to estimated grocery-based PFAS exposure, reflecting both household purchase volumes and the concentration estimates available for these food categories. Given the limited food-monitoring literature, category-specific contributions should be interpreted in relation to the available measurements.



(a) By region



(b) By race and ethnicity

Figure 4: PFAS exposure (ng) by PFAS-contaminated product across regions and demographic groups. The heatmaps display the estimated contribution of selected PFAS-contaminated products across regions (Panel a) and racial and ethnic groups (Panel b). Each cell reports the annual per-capita PFAS exposure contribution of the corresponding product, in ng per person-year. Values are calculated by dividing weighted product-level PFAS quantity by the corresponding weighted household-size denominator. The weights are NIQ household projection factors, which scale each sample household to the number of households it represents in its regional market. Products are selected if either their mean PFAS contribution across groups or the range of their contributions across groups falls in the top quartile for that panel. Products are ordered by mean PFAS contribution. In Panel (a), "–" indicates regional products for which PFAS concentration measurements are available only in selected regions. Results for the full list of PFAS-contaminated products are reported in Supplementary Figure A6.

We also place the magnitude of our exposure measure in the context of prior work. Estimated dietary intakes of PFAS across countries span more than three orders of magnitude, from roughly 0.1 to over 200 ng/kg body weight/day [26]. For North American and European consumers, Trudel et al. [9] estimated chronic uptake of PFOA of 1–130 ng/kg body weight/day and of PFOS of 3–220 ng/kg body weight/day in the mid-2000s, with food and drinking water as the dominant pathways, while Egeghy and Lorber [38] estimated typical adult PFOS intake of about 160 ng/day from all pathways. Our geometric mean of roughly 2,300 ng of PFAS per year is not directly comparable with these comprehensive intake or uptake estimates for three reasons. First, the earlier estimates predate the decline of PFAS production in the U.S., as biomonitoring shows that serum concentrations of both compounds have since declined substantially [30, 31]. Second, our measure covers only grocery products with published U.S. retail measurements and excludes drinking water and food away from home. Third, household-level purchase records allow grocery purchasing patterns to vary across households, complementing previous estimates based on per-capita consumption, scenario-based intake assumptions, or short-term dietary recalls [26, 9, 39]. Our estimate is therefore best read as a lower bound on grocery-based legacy-PFAS exposure potential rather than as an estimate of total dietary intake or total exposure.

This low central estimate should not be taken as evidence of negligible risk or compared directly with a health-based intake threshold. Our measure includes only PFOA and PFOS, excludes other PFAS compounds listed in the European Food Safety Authority's tolerable weekly intake [11], and does not include drinking water or other exposure pathways. It is therefore designed primarily to characterize variation across households rather than to provide a complete dose or risk estimate.

Several limitations should be considered when interpreting the results of this paper. First, our measure reflects estimated dietary PFAS exposure based on household food purchases and does not capture total PFAS exposure, which may also arise from drinking water, other consumer products, and environmental pathways. Importantly, food and beverage consumed away from home is not captured in grocery scanner data, despite representing approximately half of U.S. food expenditures and caloric intake. PFAS contamination in food service settings may arise from packaging, grease-resistant wrappers, and cooking equipment [40]. Because reliance on food away from home is inversely correlated with income, this omission may cause us to understate PFAS exposure among lower-income households and obscure income-related differences. Second, the analysis relies on a limited number of sources measuring PFAS concentrations in foods and beverage, which may not fully represent the diversity of the U.S. food supply. The eight peer-reviewed studies and seven FDA TDS datasets identified through our literature search vary in geographic scope, analytical methods, and the set of PFAS compounds reported, introducing heterogeneity in measurement quality that equal-weighting across studies does not fully address. Applying product-level concentrations to household purchases

also cannot capture contamination differences across brands, production origins, batches, or purchase years and may therefore obscure substantial within-category heterogeneity. Several high-consumption food and beverage categories common in U.S. diets, including eggs, fresh fruits and vegetables, and grain products, lack reliable PFAS concentration measurements in the literature and are therefore absent from our exposure estimates, likely causing us to understate both the level of exposure and the share attributable to plant-based foods. Third, mapping PFAS-contaminated food and beverage products to household purchase data involves classification and matching assumptions that may introduce measurement error. Fourth, purchase data do not directly measure consumption, as they do not account for food waste, storage, or intra-household allocation. Food waste rates vary systematically across both food and beverage categories and demographic groups: higher-income households discard a larger share of perishables on average, and seafood and fresh produce exhibit higher waste rates than shelf-stable items [41]. Because butter, potato, and seafood are the dominant contributors to our estimated PFAS exposure, and because these products have differential waste rates across the demographic groups we compare, waste-induced measurement error may not be random and negligible.

As a result, the findings should be interpreted as reflecting patterns in dietary PFAS exposure potential rather than precise measures of individual intake or total exposure. The primary contribution of the analysis is therefore the characterization of within-sample variation and its decomposition into grocery spending and dietary composition, rather than estimation of a complete population dose. By integrating contamination data with observed purchase behavior, this study provides a complementary perspective to the biomonitoring literature and offers new insights into how environmental contaminants translate into heterogeneous exposure through everyday grocery choices. More broadly, the same linkage strategy can be extended to other foodborne contaminants by combining measurements reported in the food-testing literature with nationwide purchase data, enabling large-scale assessments of dietary exposure risk whenever sufficiently compatible occurrence data are available.

Methods

Literature Search on Food PFAS Contamination

We compiled PFAS contamination data for dietary items, including food and beverages, through a comprehensive literature search of peer-reviewed and government-sponsored studies that sampled and tested grocery products for PFAS. Our search was conducted in the Scopus and Web of Science databases using combinations of keywords, including “PFAS” paired with “U.S.” or “United States”

and product-specific names drawn from two U.S. FDA Total Diet Study (TDS) collections. The search terms covered food and beverage samples spanning multiple categories, including meats, seafood, dairy products, grains, packaged meals, and beverages, and were selected to reflect typical dietary patterns in the United States.[4] All search results were screened using predefined criteria of inclusion: (1) studies must be published between 2010 to 2023 to ensure it reflects the most recent PFAS concentration in the U.S.; (2) studies must report quantitative measurements of PFAS contamination in samples of commercially available food or beverage products obtained from U.S. retail sources; (3) studies must be excluded if they focused solely on environmental media (such as ambient water), occupational exposures, packaging materials without food-contact testing, or products not intended for human consumption. Full-text articles are reviewed in detail to assess methodological quality, analytical methods, and consistency of reporting. From this process, eight peer-reviewed studies and seven FDA TDS datasets were identified as sources of usable concentration measurements. Together, these sources provided a dataset of PFAS concentrations that formed the foundation for subsequent analysis, allowing the estimation of exposure across different product types and the evaluation of the prevalence and variability of PFAS exposure from foods and beverages among U.S. consumers across diverse demographics and geographic regions.

Following TDS definitions, we divide the contiguous United States into six market regions: West, North Central, Northeast, Mid-Atlantic, Southeast, and Southwest, and categorize food and beverage items with detected PFAS contamination into two groups: national products and regional products. National products are defined as products with similar nutrient and contaminant profiles and are produced and distributed nationwide, such that the literature sampled the products across the U.S. or intentionally sampled the products from one or more local markets to represent the entire country. Otherwise, products are considered regional and only contribute to PFAS exposure estimates for their respective region. Testing results below the method detection limit are assigned zero under the lower-bound convention. For products with multiple samples within a study, we calculated the average contamination level across samples. For products tested in multiple studies, we calculated the average contamination level by first averaging the samples within each study, then averaging those study-specific measurements using equal weight across studies. Supplementary Table A1 summarizes the results of the literature search. It reports 45 national or region-specific concentration entries, corresponding to 40 distinct food and beverage products. Most food and beverage products are national, except for ground beef (North Central), tilapia & ground turkey (Mid Atlantic), baked cod (Southeast and Southwest), cooked shrimp (Northeast), tilapia (West) and baked tilapia (separate samples for the North Central and Northeast markets).

[4]Product-specific search terms were drawn from the 93 samples in the FY2021 North Central regional collection [17] and the 167 samples in the FY2018 national collection [42]. Because product names may overlap across collections, these sample counts are not interpreted as 260 distinct items.

Although PFAS include many compounds, our analysis measures the combined concentration of PFOA and PFOS. This focus is motivated by four considerations. First, PFOA and PFOS are the most consistently reported contaminants in the literature, thereby maximizing study coverage. Second, because PFAS reporting varies across studies and food categories, combining them improves comparability by reducing missing observations due to compound-specific measurement differences. Third, PFOA and PFOS share similar environmental persistence, exposure pathways, and regulatory treatment under the Safe Drinking Water Act (SDWA), making their combined level a policy-relevant indicator of legacy PFAS exposure.[5] Fourth, dietary exposure typically occurs through mixtures of PFAS rather than isolated compounds, and toxicological and epidemiological evidence suggests that PFOA and PFOS together account for a substantial share of the overall PFAS burden observed in biomonitoring studies [43].

This restriction has important implications for interpretation. PFOA and PFOS production has been substantially curtailed in the United States since the mid-2000s, and contemporary dietary exposure increasingly involves short-chain and replacement PFAS compounds (e.g., PFBS, PFBA, GenX) that are not captured here. Our estimates should therefore be interpreted as a lower bound on total dietary PFAS exposure. The directional implications for disparity findings are uncertain: if replacement compounds are disproportionately present in processed and packaged foods, which are consumed more heavily by lower-income households, our analysis may understate the income gradient in total PFAS exposure. This limitation is discussed in the preceding Discussion section. Unless otherwise specified, our exposure measure refers throughout to "PFOA+PFOS dietary exposure," making the compound scope explicit.

Household Consumption Data

We measure household consumption of PFAS-contaminated foods and beverages using the NielsenIQ Consumer Panel Dataset, which records detailed household-level grocery purchases and demographic information including race, age, education, income, household size, and the number and ages of children. This dataset covers a longitudinal panel of approximately 40,000-60,000 U.S. households beginning from 2004, with new households entering and existing households exiting the panel over time. Products are identified by universal product codes (UPCs), classified into hierarchical categories (see Supplementary Figure A1), and reported with purchase quantities for each household's shopping trip. We focus on the 2021–2023 period for two reasons. First, NielsenIQ substantially revised its product classification system in 2021. The earlier classification structure was considerably less detailed and did not include the subcategory level used in our analysis. Second, UPC descriptions in

[5] https://www.epa.gov/sdwa/and-polyfluoroalkyl-substances-pfas

the pre-2021 data are often heavily abbreviated, making it difficult to accurately identify products through keyword-based searches and map them to PFAS-contaminated food and beverage categories. To further reduce data measurement errors and improve the representativeness of the sample, we impose several restrictions on household purchase records. First, we compute monthly household spending on food departments and exclude observations below the 1^{st} percentile and above the 99^{th} percentile of the spending distribution.[6] Second, we exclude households with fewer than 12 months of purchase records within a given year. After applying these restrictions, the sample size decreases from 170,754 to 138,075 household-year observations, representing a 19.14% reduction. The final sample contains 61,532 unique households.

The names and descriptions of UPC categories and subcategories are used to link the NielsenIQ data with PFAS-contaminated products identified through the literature search.[7] The main challenge in this process is that, although each UPC is assigned to categories and subcategories with descriptive names, not all products can be matched directly to a UPC based on this information.[8] To address this issue, a scenario-based matching procedure is applied. First, for a given PFAS-contaminated product, if the name of a UPC category or subcategory exactly matches the product, it is identified as a Confirmed Category/Subcategory, and all UPCs within that category or subcategory are identified as the product (e.g., the UPC subcategory "GROUND" within the UPC category "BEEF" is a Confirmed Subcategory for the product "Ground Beef"). Otherwise, all UPC categories and subcategories that may be associated with the PFAS-contaminated product are collected as Candidate Categories/Subcategories, and the UPC-level descriptions within them are searched using the product name as keywords, with only the matched UPCs retained. Note that one UPC category, "reference cards", records products sold without specific category description; the same keyword search is applied to the descriptions of all UPCs within that category. The overall procedure is illustrated in Supplementary Figure A2, and Supplementary Table A2 summarizes the Confirmed and Candidate Categories/Subcategories used to identify each PFAS-contaminated product.

After identifying the PFAS-contaminated products as described above, we compile a comprehensive set of corresponding UPCs, denoted by $\boldsymbol{A}_i$, for each PFAS-contaminated product i. Let c_{aht} denote household h's consumption of UPC a in year t. The annual consumption of each PFAS-contaminated product i by household h in year t, denoted C_{iht}, is given by:

$$C_{iht} = \sum_{(a \in \boldsymbol{A_i})} c_{aht}. \tag{1}$$

[6] Food Departments are GROCERY, MEAT, DELI, DAIRY, FROZEN, SEAFOOD, BAKERY, PRODUCE, BULK BIN, and MISCELLANEOUS FRESH.

[7] Segment is not used because Subcategory provides a relatively more detailed and relevant classification.

[8] For example, while NielsenIQ has a UPC category for "PERCH", none of its subcategories specifically corresponds to the PFAS-contaminated product "yellow perch", and not all products within these subcategories are yellow perch.

Household h's annual PFAS exposure per capita through food and beverage in year t, denoted by y_{ht}, is therefore computed from the consumption of each PFAS-contaminated product C_{iht} and its PFAS concentration θ_i:

$$y_{ht} = \frac{\sum_i (C_{iht} \times \theta_i)}{z_{ht}}, \tag{2}$$

where z_{ht} is the household size excluding children under 12.[9]

Purchase quantities c_{aht} were standardized to kilograms before being linked to PFAS concentrations.[10] Concentrations θ_i are expressed in ng of PFAS per kilogram of product. We retained the numerical summary values reported by each source study, including reported zeros.[11] Supplementary Table A1 reports the source study and concentration used. Exposure is expressed in ng per person per year.

Throughout this paper, we use the term *dietary exposure* to refer to the estimated mass of PFOA+PFOS consumed through grocery purchases. This is distinct from *total intake*, which reflects the amount actually absorbed after accounting for bioavailability and other exposure pathways, and from *body burden*, the serum or tissue concentrations measured through biomonitoring. Our measure is best interpreted as an index of dietary PFAS exposure potential.

Descriptive Analysis

A household-by-year longitudinal dataset is constructed, consisting of household-level per capita PFAS exposure through grocery products from 2021 to 2023. To provide a thorough overview of the dataset, summary statistics are reported for PFAS exposure and key household demographic and socioeconomic characteristics, including region, age, income, education, race and ethnicity, employment status, household composition, and participation in the Special Supplemental Nutrition Program for Women, Infants, and Children (WIC). Regions are defined following the Total Diet Study (TDS) classification, which groups states with similar dietary patterns. To analyze the heterogeneity across demographic and socioeconomic groups, we report the distribution of annual per

[9] Children under 12 are excluded because their food consumption is generally lower and qualitatively different from adult consumption; treating them equivalently to adults in per-capita calculations would understate adult exposure and misrepresent child exposure.

[10] Mass units were converted using standard conversion factors (e.g., 1 oz = 0.02835 kg and 1 lb = 0.4536 kg), and volume units were converted assuming 1 L = 1 kg. Records reported in counts or without a usable package-size unit were assigned the mean converted mass for the same matched food category and year; 12.1% of PFAS-matched purchase records required this procedure. Concentrations for cooked or prepared foods were applied to the standardized purchase mass of the matched category without a separate raw-to-cooked yield conversion.

[11] We did not replace non-detects with their reported limits of detection. Such substitution would assign a positive concentration to every non-detect and could cause estimated exposure to be driven by purchase volumes of high-consumption foods and differences in study-specific reporting limits rather than by measured PFAS concentrations.

capita PFAS exposure using box plots and visualize geographic variation using state-level exposure maps. Additional details on variable construction are provided in the Supplementary Information ("Detailed construction of variables").

Regression Analysis

While the descriptive analysis provides unconditional and qualitative comparisons, multivariate regression analyses are conducted to estimate the conditional socioeconomic variation in PFAS exposure and to distinguish whether the variation is driven by differences in per-capita grocery spending or by differences in dietary composition. Specifically, we estimate the following regression model:

$$y_{ht} = \boldsymbol{\beta} \boldsymbol{X}_{ht} + \gamma q_{ht} + u_t + \varepsilon_{ht}, \tag{3}$$

where $\boldsymbol{X}_{ht}$ is a vector of dummy variables for categorical demographic and geographic groups, including region, income brackets, education levels, race and ethnicity, employment status, household composition (such as marital status and whether the household has children), and participation in WIC; $q_{ht} = Q_{ht}/z_{ht}$ is annual per-capita grocery spending measured in U.S. dollars, with Q_{ht} denoting total annual household grocery spending and z_{ht} denoting household size excluding children under 12; u_t denotes year fixed effect; and ε_{ht} is the idiosyncratic error term. Thus, γ reports the change in annual per-capita PFAS exposure associated with a one-dollar increase in annual per-capita grocery spending. The estimated coefficients $\boldsymbol{\beta}$ represent differences in PFAS exposure across demographic and geographic groups, conditional on other household characteristics. After controlling for annual per-capita grocery spending, these coefficients capture remaining group differences not accounted for by spending or the included covariates; such differences may reflect grocery-basket composition as well as other unobserved factors. Standard errors in the main specification are clustered at the household level to account for repeated observations of the same household. As a robustness check for broader geographic error dependence, we report state-level wild-cluster-bootstrap inference in Supplementary Table A5.

Oaxaca–Blinder Decomposition

To formally decompose and quantify the mechanisms causing disparities in PFAS exposure across socioeconomic groups, we implement an Oaxaca–Blinder decomposition [44, 45], a regression-based method that partitions the difference in mean PFAS exposure across groups into a component associated with differences in observed characteristics (annual per-capita grocery spending) and a

residual unexplained component labeled as "dietary composition". Consider rewriting total PFAS exposure as two separate components:

$$y_{ht} = q_{ht} \cdot s_{ht}, \tag{4}$$

where $q_{ht} = \frac{Q_{ht}}{z_{ht}}$ denotes household per capita grocery spending, and $s_{ht} = \frac{\sum_i C_{iht}\theta_i}{Q_{ht}}$ denotes household's PFAS exposure per dollar of spending, capturing variation driven by differences in the composition of the household's grocery basket.

Consider using the Oaxaca–Blinder method to decompose the difference in average PFAS exposure between two socioeconomic groups, A and B, in the following two steps. First, two group-specific regressions are estimated:

$$\begin{aligned} y_{ht} &= \beta_A\, q_{ht} + \varepsilon_{ht}, \quad \text{if } h \in \text{Group } A, \\ y_{ht} &= \beta_B\, q_{ht} + \varepsilon_{ht}, \quad \text{if } h \in \text{Group } B. \end{aligned} \tag{5}$$

Equation (5) is estimated without an intercept.[12] Here, β_A and β_B estimate group-specific PFAS exposure per dollar of grocery spending and can be interpreted as the increase in annual per-capita PFAS exposure associated with a one-dollar increase in annual per-capita grocery spending. Second, the group difference is decomposed as follows:

$$\Delta y_{AB} \;=\; \hat{\beta}_A \bar{q}_A - \hat{\beta}_B \bar{q}_B \;=\; \underbrace{(\bar{q}_A - \bar{q}_B)\,\hat{\beta}_B}_{\text{grocery spending effect}} \;+\; \underbrace{\bar{q}_A\,(\hat{\beta}_A - \hat{\beta}_B)}_{\text{dietary composition effect}}, \tag{6}$$

where $\bar{q}_A$ and $\bar{q}_B$ denote the group means of household per-capita spending q_{ht}, averaged over all years, and $\hat{\beta}_A$ and $\hat{\beta}_B$ are estimated from Eq. (5). In the decomposition, the first term represents the change in PFAS exposure if Group B's spending were shifted to Group A's spending level while holding Group B's dietary composition fixed, capturing the difference attributable to grocery spending across groups; the second term represents the additional change in PFAS exposure if Group B's dietary composition were shifted to Group A's (conditional on the previous spending adjustment), capturing the difference attributable to dietary composition.

Statistical analyses were performed using Stata 18 (StataCorp LLC, College Station, TX, USA), and data processing and visualization, including geographic mapping, were performed using Python 3.11.5 with GeoPandas 1.0.1 and Matplotlib 3.7.2.

[12]The no-intercept specification follows from the construction of the outcome: household exposure is calculated from purchased quantities and the corresponding PFAS concentrations, so $y_{ht} = q_{ht}s_{ht}$ by construction and zero purchases imply zero purchase-based exposure. The resulting two components provide an exact decomposition of the difference between the group-specific fitted means. Because no-intercept OLS does not require the residuals to have a zero group mean, this model-fitted gap may differ from the raw arithmetic mean exposure gap.

Ethics statement

This study did not require ethical approval because it relies exclusively on de-identified, secondary household purchase data licensed from the NielsenIQ Consumer Panel (NIQ); the authors did not directly recruit, interact with, or collect data from human participants.

Data availability

The NielsenIQ Consumer Panel household purchase data used in this study were obtained under a restricted-access license through the Kilts Center for Marketing at the University of Chicago Booth School of Business. The authors are contractually prohibited from redistributing either the source purchase records or the household-level analytical dataset derived from them. Eligible academic researchers may apply independently for access through the Kilts Center (https://www.chicagobooth.edu/research/kilts/research-data/nielseniq), subject to institutional subscription, researcher registration and approval, applicable fees, and the Center's data-use policies. Questions about access may be directed to marketingdata@chicagobooth.edu. The complete harmonized food-level PFAS concentration dataset compiled by the authors for this study, including product, concentration, geographic scope, and source, is provided in Supplementary Table A1. The underlying concentration measurements are available from the publications and federal Total Diet Study datasets cited in that table.

Code availability

Analysis code sufficient to reproduce the reported results from the licensed source data is available from the corresponding author on reasonable request, subject to the restrictions of the NIQ data-use agreement. The code will also be made available to editors and peer reviewers upon request.

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Researcher(s)' own analyses calculated (or derived) based in part on data from Nielsen Consumer LLC and marketing databases provided through the NielsenIQ Datasets at the Kilts Center for Marketing Data Center at The University of Chicago Booth School of Business. The conclusions drawn from the NielsenIQ data are those of the researcher(s) and do not reflect the views of NielsenIQ. NielsenIQ is not responsible for, had no role in, and was not involved in analyzing and preparing the results reported herein.

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Author Contributions

J.G.: Data curation, formal analysis, methodology, visualization, and writing (original draft). R.Z.: Conceptualization, formal analysis, methodology, funding acquisition, and writing (review and editing). J.Z.: Conceptualization, formal analysis, methodology, and writing (review and editing). Y.R.: Data curation, formal analysis, methodology, and writing (review and editing). X.W.: Conceptualization, formal analysis, methodology, and writing (review and editing). W.Z.: Conceptualization, formal analysis, methodology, and writing (review and editing). All authors reviewed and approved the final manuscript.

Competing Interests

The authors declare no competing interests.

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