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Determinants of per- and polyfluoroalkyl substances (PFAS) exposure among Wisconsin residents

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ABSTRACT

Background: Per- and polyfluoroalkyl substances (PFAS) include thousands of manufactured compounds with growing public health concerns due to their potential for widespread human exposure and adverse health outcomes. While PFAS contamination remains a significant concern, especially from ingestion of contaminated food and water, determinants of the variability in PFAS exposure among regional and statewide populations in the United States remains unclear.

Objectives: The objective of this study was to leverage The Survey of the Health of Wisconsin (SHOW), the only statewide representative cohort in the US, to assess and characterize the variability of PFAS exposure in a general population.

Methods: This study sample included a sub-sample of 605 adult participants from the 2014–2016 tri-annual statewide representative sample. Geometric means for PFOS, PFOA, PFNA, PFHxS, PFPeS, PFHpA, and a summed measure of 38 analyzed serum PFAS were presented by demographic, diet, behavioral, and residential characteristics. Multivariate linear regression was used to determine significant predictors of serum PFAS after adjustment.

Results: Overall, higher serum concentrations of long-chain PFAS were observed compared with short-chain PFAS. Older adults, males, and non-Hispanic White individuals had higher serum PFAS compared to younger adults, females, and non-White individuals. Eating caught fish in the past year was associated with elevated levels of several PFAS.

Discussion: This is among the first studies to characterize serum PFAS among a representative statewide sample in Wisconsin. Both short- and long-chain serum PFAS were detectable for six prominent PFAS. Age and consumption of great lakes fish were the most significant predictors of serum PFAS. State-level PFAS biomonitoring is important for identifying high risk populations and informing state public health standards and interventions, especially among those not living near known contamination sites.

1. Introduction

Per- and polyfluoroalkyl substances (PFAS) are a large family of manufactured chemicals that were first produced in the 1940s for a variety of consumer products, industrial uses, and other applications due to their hydrophobic and oleophobic properties (Buck et al., 2011). They are well-known for their use in nonstick cookware, firefighting foams, and fast-food packaging, and a myriad of other applications (Glüge et al., 2020). Many PFAS are highly stable due to their carbon-fluorine bonds, which are among the strongest chemical bonds (Wang and Liu, 2020). While perfluorooctane sulfonate (PFOS) and perfluorooctanoic acid (PFOA) have been voluntarily phased out of production in recent decades in the United States due to their potential harmful effects, they remain ubiquitous in the environment (Sunderland et al., 2019). To date, thousands of PFAS have been identified and new compounds are identified regularly (NIEHS, 2023).

Tens of thousands of PFAS contamination sites have been identified

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Abbreviations:

PFAS	Per- and polyfluoroalkyl substances substances
PFOS	perfluorooctane sulfonic acid
PFOA	perfluorooctanoic acid
PFNA	perfluorononanoic acid
PFHxS	perfluorohexane sulfonic acid
PFPeS	perfluoropentane sulfonic acid
PFHpA	perfluoroheptanoic acid
SHOW	Survey of the Health of Wisconsin
WSLH	Wisconsin State Laboratory of Hygiene

in the United States (US) due to industrial, commercial, and other uses at military sites, airports, and sites of PFAS production (Marusic, 2022). In a National Health and Nutrition Examination Survey (NHANES) sample of the US population, 98% of participants had detectable PFAS in their blood serum (ATSDR, 2022). Humans are most often exposed to PFAS through food and water, and PFAS typically distribute to the blood, kidney, and liver where they often bond to lipoproteins (Andersen et al., 2021). Human exposure to PFAS presents potential human health concerns. PFAS exposure has been associated with metabolic changes, like elevated cholesterol levels and changes in liver enzymes (Roth et al., 2021). PFOA is listed by the International Organization for Research on Cancer (IARC) as a potential carcinogen and has been linked to several cancers (ACS, 2023). Both long-chain PFAS (a perfluoroalkyl carboxylic acid with 8 or more carbon atoms, or a perfluoroalkyl sulfonic acid with 6 or more carbon atoms) and shorter-chain PFAS (a perfluoroalkyl carboxylic acid with fewer than 8 carbon atoms, or a perfluoroalkyl sulfonic acid with fewer than 6 carbon atoms) have been associated with changes in puberty onset, lower bone density, adverse cardiometabolic effects during adolescence (Panieri et al., 2021), low birth weight (Wikström et al., 2020), preeclampsia (Bommarito et al., 2021), and decreased vaccine efficacy in children (von Holst et al., 2021).

Human exposure to PFAS through diet is especially salient because some PFAS bio-magnify, becoming more concentrated in organisms in higher trophic levels (Lewis et al., 2022). Fish and red meat consumption have been associated with elevated PFAS levels in humans (Sunderland et al., 2019; Susmann et al., 2019; Lin et al., 2020; Liu et al., 2022). PFAS have also been found in cereals, vegetables, and dairy products, although concentrations were lower than that of fish and other proteins in many cases (Vestergren et al., 2012; Herzke et al., 2013). Fast food and processed food packaging can contain PFAS (Dueñas-Mas et al., 2023), which may allow small amounts of PFAS to leach into the food. For example, consumption of microwave popcorn has been associated with elevated serum PFAS because the interior non-stick liners of the popcorn bag can contain PFAS (Susmann et al., 2019).

Previous studies have also found human PFAS serum or plasma levels vary by demographics and geography. Pre-menopausal females tend to have lower PFAS serum concentrations compared to post-menopausal females and males, which is thought to be related to regular loss of contaminated blood through menstruation and reproduction (Park et al., 2019; Chang et al., 2021). Higher income and educational attainment have also been associated with higher serum PFAS (Chang et al., 2021; Buekers et al., 2018). Individual and total PFAS have been negatively correlated with home age and positively correlated with the percentage of carpeting in the home (Kubwabo et al., 2005), which is thought to be due to older construction materials and carpeting which was more likely to contain PFAS. Geographic location is also an important driver of PFAS concentrations, both in food products (Herzke et al., 2013) and human exposure from the environment (Park et al., 2019; DeLuca et al., 2023; Karásková et al., 2016), which is thought to be due to varying geographical differences in industrial effluent and emissions, as well as PFAS-contaminated wastewater sludge application on farm fields. Thus, human exposure profiles can differ locally, especially among those residing or working near a contamination site.

Furthermore, national biomonitoring, while informative as a baseline for comparison, does not provide the regional and state-level granularity necessary for state-level public health intervention. Identifying determinants of elevated PFAS concentrations in the general population at the state level may help to identify subgroups at higher risk of exposure who do not live near known contamination sites. A handful of statewide biomonitoring studies have been conducted for PFAS, mostly in the Northeast and Southern US (Yu et al., 2021; Nair et al., 2021; Petriello et al., 2022), and in California (Aylward et al., 2015). While these studies have more geographic granularity than the national level, these studies do not contain many questions regarding dietary components, a known major route of PFAS exposure (Sunderland et al., 2019), or around fish and seafood consumption, where PFAS are known to accumulate (Lewis et al., 2022). Furthermore, an understanding of statewide PFAS exposure can inform state-level awareness, education, and policy toward decreasing risk of exposure in vulnerable populations.

The upper Midwest of the United States is a key area of concern for potential PFAS exposure due to the strong culture of fishing (Ullman, 2022) and history of manufacturing (Williams and Schrank, 2016). However, a lack of PFAS biomonitoring data among the Midwestern United States population remains. This study fills that data gap by presenting biomonitoring data from a population-based adult sample in Wisconsin, a state in the upper Midwestern US, that investigates dietary components, especially eating caught fish, among other demographic, behavioral, and environmental characteristics, as determinants of PFAS exposure. The Survey of the Health of Wisconsin (SHOW) is the only U.S. state survey modeled after the National Health and Nutrition Examination Survey (NHANES) and collects biological samples, objective body measurements, and health survey data. In the present study we aim to characterize elevated serum perfluorooctane sulfonic acid (PFOS), perfluorooctanoic acid (PFOA), perfluorononanoic acid (PFNA), perfluorohexane sulfonic acid (PFHxS), perfluoropentane sulfonic acid (PFPeS), and perfluoroheptanoic acid (PFHpA), and summed PFAS (38 PFAS) among a state-wide sample in Wisconsin. Findings from this study will help to identify potential subgroups in the state who may be at increased risk for elevated PFAS exposure.

2. Methods

2.1. Study sample

This study consists of a sub-sample of n = 605 adult participants randomly selected from the 2014–2016 triannual, statewide sample of adult residents (n = 1957) in the Survey of the Health of Wisconsin (SHOW) cohort. The SHOW is a population-based health survey of Wisconsin residents. SHOW sampling and methods are further described elsewhere (Malecki et al., 2022). The PFAS subsample was selected to be representative of Wisconsin by racial makeup and urbanicity using simple random sampling from the larger SHOW sample. In brief, the partner SHOW sample included a state-wide address-based sampling frame and three-stage, area probability sampling without replacement (PPSWOR) was used to generate the tri-annual statewide representative sample, where county was the primary sampling unit, upon which census block groups and then households were selected. Ten Wisconsin counties are represented in the 2014–2016 statewide sample. A map of the of the study sample used in this study is available elsewhere (Schultz et al., 2023). Participants completed in-person interviews and a self-administered survey which captures demographics, social determinants of health, health history and health behaviors. They also visited an exam center near their home for specimen collection (blood, urine, and stool). Blood specimens are collected from trained phlebotomists, centrifuged, processed, and specimen derivatives are stored in −80C freezers at SHOW headquarters at the University of

Wisconsin-Madison. The SHOW protocol and informed consent documents are approved by the Health Sciences Institutional Review Board of the University of Wisconsin-Madison. Participants in SHOW gave consent to their information being used for research prior to this study.

2.2. PFAS quantification

Serum samples collected at the time of participation were extracted from the SHOW biobank (−80C freezer) and the concentration of serum PFAS was quantified by the Wisconsin State Laboratory of Hygiene (WSLH) using liquid chromatography-tandem mass spectrometry following isotope dilution and extraction with solid phase extraction. In 2020, the WSLH developed a new method which enabled the detection of PFAS compounds at lower concentrations, increasing the number of compounds available. A total of 38 PFAS compounds were selected and analyzed among SHOW serum samples; eight PFAS were compared demographically to NHANES (Schultz et al., 2023). The complete PFAS compound list can be viewed in the supplementary materials (Supplementary Table 1) The WSLH test method was adapted primarily from a method developed by Minnesota Department of Public Health (MDPH, 2024) with elements from CDC Method 6304.08 (CDC, 2024), the New York State Department of Health (NYSDH, 2023), and the Michigan Department of Community Health (MDCH, 2018). Multiple quality checks were used to ensure accurate measurement, including a method blank and seven standard linear calibration standards ($r \geq 0.995$), verified with a second source material. Three levels of analytical controls were measured in every analytical run. Methods were also validated using precision, analytic measurement range, and spike recovery assessments. Further documentation on the WSLH analytic methods of PFAS quantification for all PFAS compounds, detection frequencies in the SHOW sample, and overall serum PFAS values are available elsewhere (Schultz et al., 2023). Only compounds with a 50% detection rate or higher were included in analyses. These compounds included perfluorooctane sulfonic acid (PFOS), perfluorooctanoic acid (PFOA), perfluorononanoic acid (PFNA), perfluorohexane sulfonic acid (PFHxS), perfluoropentane sulfonic acid (PFPeS), and perfluoroheptanoic acid (PFHpA). The lower limits of quantification for these compounds were 0.036 ng/mL for PFOS, 0.025 ng/mL for PFOA, 0.021 ng/mL for PFNA, 0.028 ng/mL for PFHxS, 0.015 ng/mL for PFPeS, and 0.031 ng/mL for PFHpA.

2.3. Demographics, behaviors, and characteristics

Demographics (including race and ethnicity), diet, and behaviors were self-reported and obtained using self-administered questionnaires and interviews from trained field staff using a computer-assisted personal interview (CAPI). Body measurements (height and weight) were collected at in-home visits from trained field staff members and body mass index (BMI) was calculated as kg/m^2. BMI was categorized into underweight or normal weight (BMI<25), overweight (25<BMI<30), or obese (BMI>30), based on guidelines from the Centers of Disease Control and Prevention (CDC) (CDC, 2023). Age at time of participation was categorized into ages 18–39, 40–59, and 60 years of age and older for descriptive tables but were retained as a continuous variable (divided by 10 for ease of interpretation) for adjusted regression models. Household income and the number of people supported on the income was used to calculate the poverty level using the U.S. Federal Poverty Level (FPL) via the U.S. Department of Health and Human Services' annual poverty guidelines (US HHS, 2024). Poverty Income Ratio (PIR) was calculated as household income divided by poverty level, multiplied by 100, and expressed as a percent - %FPL (i.e. a PIR of "1" indicated a household income at 100% of the FPL). Race was grouped into non-Hispanic White and non-White participants due to a small percentage of participants who did not self-identify as Caucasian/White being their only race. Rurality was determined by Census 2010 classification, and participants were considered rural if their residence was in a census block group with fewer than 2500 people (U.S. Census Bureau, 2023). Eating caught fish was assessed by asking participants "Did you ever eat fish caught by you or given to you in the last year?" which was coded as a binary variable (1 = yes and 2 = no). Red meat and processed meat consumption were considered "high intake" if participants reported eating the food items one or more times per week. Daily fruit, vegetable, and legume intake, and daily dairy intake were assessed by asking participants "During the past month, how often did you eat [food item]?" This was modified from the NHANES 26-item dietary screener questionnaire (NHANES, 2021a). Predicted daily intake of fruit, vegetables, and legumes were calculated using the Self-Administered Questionnaire: Paper SAS program from the National Cancer Institute (NHANES, 2021b). High fruit, vegetable, and legume intake was defined as either meeting (≥3 cups) or not meeting (<3 cups) the minimum daily recommendation for vegetable intake for adults according to the CDC (Lee et al., 2022). High dairy intake was defined as predicted intake higher than 1.25 cups per day, based on the average daily dairy intake for American adults (Hess et al., 2021). Land use percentages within one mile of participant households was calculated using ESRI ArcGIS and linked to WISCLAND land cover data from the Wisconsin Department of Natural Resources, which derived land cover data from satellite imagery acquired from the Landsat 5 Thematic Mapper, Landsat 7 Enhanced Thematic Mapper, and Landsat 8 Operational Land Imager between 2010 and 2014 (WDNR, 2023).

2.4. Statistical analysis

Stratified geometric means were generated by demographic and residential characteristics (age, sex, race, education, annual household income, rurality, water source, water filter use, and home construction date), dietary components (eating caught fish, eating Great Lakes fish or Inland Wisconsin fish, fruit, vegetable, and legume intake, red meat intake, processed meat intake, dairy intake, frequency of popcorn consumption, and frequency of fast-food consumption), biological metrics, and health behaviors (smoking status, BMI, creatinine, triglycerides, and dental floss use frequency). To obtain geometric means across strata a lower limit of reporting was set at $0.1/\sqrt{2}$ based on the PFAS quantification methods of the National Health and Nutrition Examination Survey (NHANES) (ATSDR, 2022). Univariate associations were assessed via Wilcoxon rank-sum tests. Participants were included in analyses if they had complete data on the stratified variable. To correct for multiple comparisons, we controlled for the false discovery rate at an alpha level of 0.05 using the Benjamini-Hochberg procedure. Groups were only considered statistically significantly different at the 0.05 level after controlling for the false discovery rate. Self-reported smoking status contributed to most missing data (n = 41), so we included those with missing smoking information as its own category in regression models to reduce selection bias. For regression models, only those with complete data on all other covariates were included (n = 577).

Multivariate linear regression modeling was used to model predictors of natural log-transformed serum PFAS after covariate adjustment. Log transformation weakens the assumption of data normality in linear regression modeling, making multivariate linear regression possible. Separate linear regression models were constructed for PFOS, PFOA, PFNA, PFHxS, PFPeS, and PFHpA; a summed measure of total serum PFAS was used to assess overall body burden. The total serum PFAS measure was created by summing the serum concentration of all 38 compounds initially quantified for each participant by the WSLH (Schultz et al., 2023). The following demographics and characteristics were considered in fully adjusted models: age, sex, race, annual household income as % FPL, caught fish consumption, smoking status, creatinine level, popcorn consumption, fast food consumption, percent agriculture land use near residence, and percent urban development near residence. Percent agricultural land use and percent urban development are intended as crude proxy measures for exposure routes through agricultural lands and industrial sites that urban/rural categorizations may not capture. Age, gender, race/ethnicity, and eating

caught fish, were selected a-priori to remain in all models based on previous literature. Covariates were retained if they were an established demographic predictor selected a-priori, or if they showed a lower Bayesian Information Criterion (BIC) compared to the previous model. Variables included in the model were not guaranteed to be retained in the final parsimonious model based on BIC change. Following regression modeling, beta values were back transformed into percent difference in serum PFAS using methods from (Boronow et al., 2019) to express values as percent difference in PFAS (Boronow et al., 2019). All statistical analyses were performed in SAS v9.4.

3. Results

3.1. Sample demographics

The study sample was majority non-Hispanic White (83.4%), female (57.4%), and living in an urban area (69.3%). Seventy-three percent of the subsample were over the age of 40 years and 38.5% were over the age of 60 years. Just under half of the sample (n = 264) reported eating fish that was caught or given to them in the past year, and among those, 35.7% and 77.9% reported eating fish from the Great Lakes or inland Wisconsin streams and lakes, respectively. Thirty-nine percent of participants had a bachelor's degree or higher and sixty-one percent were classified as never smokers. Eighty-one percent of participants had a predicted dietary intake of less than three cups of fruits and vegetables per day. Sixty-two percent and 38.2% of the sample reported eating red and processed meats once per week or more, respectively. About a quarter of participants had borderline high or high triglycerides (higher than 170 mg/dL). A complete description of participant characteristics is available in Table 1.

3.2. Characterization of PFAS serum levels

Table 2 depicts serum PFAS geometric means overall, by demographic, and by residential characteristics. PFOS serum concentrations were higher among the sample (4.78 ng/mL) than all other compounds. PFOA and PFHxS had the next highest geometric means (1.24 and 1.16 ng/mL, respectively), indicating that PFOS, PFOA, and PFHxS contributed most to the summed PFAS measure, and overall PFAS body burden. Older age, male sex, and non-Hispanic White race showed the most consistent univariate associations with PFAS levels, with differences seen in serum levels of PFOS, PFOA, PFNA, PFHxS and sum PFAS by the three demographics. In univariate analyses, older individuals and non-Hispanic White adults had higher serum PFOS, PFOA, PFNA, PFHxS, and total serum PFAS compared to younger individuals and non-White adults, respectively. Males had higher PFOS (5.61 vs 4.24 ng/mL), PFHxS (1.44 vs 0.99 ng/mL) and summed PFAS (12.99 vs 10.61 ng/mL) compared to females. Those with an annual household income greater than $100,000 had higher serum PFOA compared to those with an annual household income less than $50,000 (1.49 vs 1.13 ng/mL). Residing with one mile of agricultural land was not associated with PFAS serum levels. Those with lower urban developed land use within 1 mile of their home had slightly higher PFPeS. When examining dietary characteristics and PFAS levels in univariate analyses, associations were found between PFAS serum levels and eating caught fish, and fast-food consumption frequency. Those who reported eating caught fish in the past year had statistically significantly higher PFOS (5.66 vs 4.19 ng/mL), PFNA (0.52 vs 0.44 ng/mL), and total serum PFAS (12.76 vs 10.72 ng/mL) compared to those who did not report eating caught fish in the last year. Interestingly, those who reported never eating fast food had, on average, higher serum concentrations of PFOS, PFOA, PFNA, and total serum PFAS compared to those who reported eating fast food more frequently. Smoking status was statistically significantly associated with PFAS serum levels. Never smokers had higher and total serum PFAS (11.80 ng/mL vs 9.77 ng/mL) compared to current smokers, and former smokers also had higher serum PFOA (1.28 vs 1.01 ng/mL)

Table 1
Selected Demographics and Characteristics of the Survey of the Health of Wisconsin (SHOW) n = 605 subsample.

	Total Sample (n = 605)	
	n	%
Gender		
Male	258	42.6
Female	347	57.4
Age (in years)		
18–39	163	26.9
40–59	209	34.6
60–94	233	38.5
Race		
White (non-Hispanic)	504	83.4
Non-White	100	16.6
Missing	1	0.1
Education		
H.S.[a]/GED[b] or less	152	25.1
Some college	217	35.9
Bachelor's degree or higher	236	39.0
Income		
<$50,000	239	39.5
$50,000–$99,999	209	34.5
>$99,999	130	21.5
Missing	27	4.5
Smoking Status		
Current	73	12.1
Former	147	24.3
Never	345	57.0
Missing	40	6.6
BMI[c]		
<25	167	27.6
25–30	201	33.2
>30	237	39.2
Physical Activity		
<600 MET[d] Minutes/week	160	26.5
≥ 600 MET[d] Minutes/week	445	73.5
Urbanicity		
Urban	419	69.3
Rural	186	30.7
Eat Caught Fish		
Yes	264	43.6
No	341	56.4
Popcorn Consumption Frequency		
Never	128	21.2
Less than Once per Week	326	53.9
More than Once per Week	120	19.8
Missing	31	5.1
Fast Food Consumption Frequency		
Never	32	5.3
Less than Once per Week	383	63.3
More than Once per Week	190	31.4
Fruit and Vegetable Intake		
Less than 3 cups per day	490	81.0
3 cups per day or more	115	19.0
Dairy Consumption		
Less than 1.25 cups per day	329	54.4
1.25 cups per day or more	276	45.6
Red Meat Consumption Frequency		
Once per week or less	228	37.7
More than once per week	377	62.3
Processed Meat Consumption Frequency		
Once per week or less	374	61.8
More than once per week	231	38.2
Creatinine (mg/dL)		
<0.7	61	10.1
0.7–1	360	59.5
>1	184	30.4
Triglycerides (mg/dL)		
<78	158	26.1
78–170	298	49.3
>170	149	24.6
Flossing Frequency		
Every day or Most Days	290	47.9
Some Days or Less	255	42.1
Does not Apply	17	2.8
Missing	43	7.1

[a] HS = High School.
[b] GED = General Educational Development.
[c] BMI = Body Mass Index (kg/m^2).
[d] MET = Metabolic Equivalent of Task.

compared to current smokers (Table 2). A complete list of stratified geometric mean analyses is available in the supplementary material (Supplementary Tables 2–4).

3.3. Regression modeling

Covariates were selected based on BIC criteria. Age, sex, race, eating caught fish in the last year, and household income remained in all adjusted models. Popcorn consumption, fast food consumption, smoking status, creatinine, and percent agricultural land use near the home no longer showed strong associations with PFAS serum levels in adjusted analyses. Fig. 1 depicts the percent difference in geometric mean PFAS serum levels by characteristics in parsimonious, mutually adjusted models. The largest differences in PFAS serum levels were seen with age, sex, and race. Older age was associated with higher serum PFAS for all compounds, except for PFPeS. The greatest difference was seen with PFOS, where a 24% (95% CI: 19.9–29.1) increase was seen for every 10-year increase in age. Compared to females, males' serum levels were, on average, 30% (95% CI: 14.4–47.7) higher for PFOS and 44% (95% CI: 23.8–66.8) higher for PFHxS; similar but smaller differences were seen for all PFAS except PFHpA, where females had a slightly higher average serum level compared to males, though this association was not statistically significant. Non-Hispanic White participants had elevated PFOS, PFOA, PFNA, PFHxS, and Sum-PFAS compared to non-White participants. Most notably, non-Hispanic White adults had 38% (95% CI: 17.4–62.0), 34% (95% CI: 9.1–64.6), and 31% (95% CI: 9.5–55.7) higher levels of PFOA, PFHxS and PFOS compared to non-White adults, respectively. Eating caught fish within the last year at the time of the survey was associated with 21% (95% CI: 6.6–37.7) higher PFOS, with similar but smaller associations seen with PFNA, PFHxS, PFPeS and summed PFAS. Higher annual household income was consistently associated with slightly higher serum PFAS, with a 100% increase in the federal poverty level being associated with 4% higher PFOA and PFNA (95% CI: 1.9–6.4, and 1.4–5.9, respectively), and 3% higher PFOS and PFHxS (95%CI: 0.9–5.9, and 0.4–6.2, respectively). Having 100% urban development within a one-mile radius of the participant's residence was associated with a 28.8% (95% CI: 8.9–52.4) higher PFOA serum level compared to households with 0% of landcover within one mile being urban development (Fig. 2). Percent differences tended to be smaller and/or inverse for PFPeS and PFHpA by each characteristic, when compared to the other PFAS examined. Estimates of preliminary, full, and parsimonious models are available in the Supplementary Materials (Supplementary Tables 5–11).

4. Discussion

To our knowledge this is the first study to characterize serum PFAS among a statewide, population-based sample of adults in Wisconsin, and model predictors of exposure to both short- and long-chain PFAS and the sum of 38 PFAS. This study fills a data gap by examining trends in PFAS exposure within a more general population-based sample not located or selected due to a known PFAS contamination source. The state-wide sample is unique because it provides more geographic granularity than national studies might while maintaining a more regional and general population set of characteristics. The extenstive questionnaire data available also provided a novel opportunity to examine detailed information on novel sources of PFAS including specific dietary (popcorn and caught fish consumption), behavioral data (smoking status, floss use). These are in addition to more traditional pathways and routes of PFAS exposure including drinking water source, and residential proximity data to agricultural lands and urban centers with PFAS. While other studies have examined PFAS body burden with demographic, behavioral, and environmental predictors, few have been able to examine several within the same study population. We found higher serum concentrations of long-chain PFAS compared with short-chain PFAS, and stronger associations among predictors with long-chain PFAS compared with short-chain. We found older age, male sex, White race, higher household income, and eating caught fish were positively associated with higher serum PFAS in adjusted regression models. Income attenuated the relationship between race and PFAS, and eating caught fish attenuated the relationship between sex and PFAS, though both associations remained significant after adjustment. Our findings align with those from other studies. These state-level biomonitoring findings are important for understanding variability in exposure and identifying at-risk subpopulations upon which can shape state-level monitoring, policy regulations, and inform targeted educational efforts.

Higher serum concentrations of long-chain PFAS were observed compared to short-chain PFAS, in agreement with previous studies. Short-chain PFAS have shorter half-lives in human serum and are more readily eliminated than long-chain PFAS (Xu et al., 2020), which may explain our findings. PFPeS has shown a particularly rapid decrease in human serum (Li et al., 2022). We also found smaller, and sometimes inverse associations compared to long-chain compounds, which reflects similar findings by (Huo et al., 2023), who found fewer associations between demographic factors and PFHpA compared to other compounds (Huo et al., 2023), which may be similarly due to the shorter half-lives of short-chain PFAS. Short-chain PFAS had slightly different univariate associations than long-chain PFAS. PFPeS and PFHpA were not associated with age, sex, and race, which contrasts with the long-chain PFAS included in analyses. Additionally, short-chain compounds were not significantly associated with dietary components, unlike their long-chain counterparts.

Age, sex, and race remained important predictors of PFAS exposure, even after adjustment for covariates, and especially for long-chain compounds and summed PFAS concentrations. Older individuals, males, and non-Hispanic White adults had significantly higher serum PFAS compared to younger individuals, females, and non-White adults, respectively. Many other studies have found similar results among highly exposed New Jersey and Pennsylvania residents, as well as pregnant women (DeLuca et al., 2023; Nair et al., 2021; Graber et al., 2019). Older adults and males having higher PFDA, PFHSxS, PFOA, PFNA, and PFOS have been observed nationally in the US, as well as in China, Germany, and Australia. Older individuals are likely to have higher serum PFAS due to the bioaccumulative potential of some PFAS and having more time at risk of exposure to PFAS. Males are known to have higher serum PFAS due to females having more elimination pathways for PFAS due to menstruation, childbirth, and breastfeeding (Park et al., 2019). Although, a unique strength of our study was its ability to examine consumption of caught fish with PFAS. Eating caught fish explained some of the differences in serum PFAS values (5–8%) seen between males and females. In NHANES, associations between non-restaurant food consumption and PFAS stronger in females compared to males (Susmann et al., 2019). Yet, similar proportions of caloric intake from these foods were seen across both genders, suggesting there may be some additional confounding, differences in metabolism, and/or pharmacokinetics between the sexes. In addition to potential dietary differences by sex, differences in occupation and cosmetic use may differentially affect exposure to PFAS (ATSDR, 2024; Susmann et al., 2019). More research is needed to better understand the extent to which differences in diet, consumer product, and occupational PFAS exposures explain differences in body burden by sex.

It remains unclear why higher serum PFAS are observed among non-Hispanic White adults compared to non-White adults. Research in this area is sparse, but other studies have also found higher levels of some serum PFAS among non-Hispanic White adults (Barton et al., 2020; Bangma et al., 2020). Our findings by race are similar to those seen in a

Table 2
Serum PFAS geometric means (ng/mL) and 95% confidence intervals by selected demographics and characteristics.

	n	PFOS	PFOA	PFNA	PFHxS	PFPeS	PFHpA	Sum-PFAS
		Mean (95% CI)	Mean (95% CI)	Mean (95% CI)	Mean (95% CI)	Mean (95% CI)	Mean (95% CI)	Mean (95% CI)
	605	4.78 (4.45–5.13)	1.24 (1.17–1.31)	0.47 (0.45–0.50)	1.16 (1.08–1.25)	0.08 (0.07–0.08)	0.08 (0.07–0.08)	11.57 (11.07–12.09)
Age								
18–39	163	2.93 (2.55–3.38)**	0.87 (0.77–1.00)**	0.34 (0.3–0.38)**	0.89 (0.76–1.04)**	0.08 (0.07–0.08)	0.07 (0.07–0.08)	8.55 (7.94–9.22)**
40–59	209	4.27 (3.82–4.78)**	1.26 (1.14–1.38)**	0.44 (0.40–0.49)**	1.04 (0.92–1.19)**	0.07 (0.07–0.08)	0.08 (0.07–0.08)	10.69 (9.97–11.46)**
60+ (ref)	233	7.42 (6.77–8.13)	1.55 (1.43–1.68)	0.63 (0.58–0.69)	1.54 (1.39–1.72)	0.07 (0.07–0.08)	0.08 (0.07–0.08)	15.34 (14.39–16.35)
Sex								
Male	258	5.61 (5.01–6.29)**	1.30 (1.19–1.42)*	0.49 (0.45–0.54)	1.44 (1.28–1.62)**	0.08 (0.07–0.08)	0.07 (0.07–0.08)*	12.99 (12.17–13.87)**
Female (ref)	347	4.24 (3.87–4.63)	1.19 (1.10–1.29)	0.46 (0.42–0.50)	0.99 (0.90–1.09)	0.07 (0.07–0.08)	0.08 (0.07–0.08)	10.61 (10.01–11.25)
Race								
Non-White	100	3.00 (2.43–3.69)**	0.88 (0.73–1.01)**	0.34 (0.30–0.40)**	0.78 (0.63–0.96)**	0.07 (0.07–0.08)	0.07 (0.07–0.08)	8.70 (7.81–9.70)**
White (Non-Hispanic) (ref)	504	5.25 (4.88–5.64)	1.33 (1.25–1.42)	0.50 (0.47–0.54)	1.23 (1.12–1.34)	0.07 (0.07–0.08)	0.08 (0.07–0.08)	12.26 (11.69–12.84)
Annual Household Income								
>$100,000	130	5.38 (4.84–5.98)	1.49 (1.35–1.65)*	0.54 (0.49–0.59)	1.35 (1.19–1.53)	0.07 (0.07–0.08)	0.08 (0.07–0.08)	12.29 (11.40–13.24)
$50,000–$99,999	209	4.75 (4.19–5.38)	1.22 (1.11–1.35)*	0.48 (0.43–0.53)	1.14 (0.99–1.31)	0.08 (0.07–0.08)	0.07 (0.07–0.08)	11.52 (10.68–12.42)
<$50,000 (ref)	239	4.50 (3.97–5.11)	1.13 (1.02–1.26)	0.44 (0.39–0.48)	1.08 (0.95–1.23)	0.08 (0.07–0.08)	0.08 (0.07–0.08)	11.24 (10.42–12.13)
Land Use: Agriculture[c]								
0% Agriculture	213	4.30 (3.79–4.87)	1.24 (1.12–1.37)	0.44 (0.40–0.49)	1.09 (0.96–1.25)	0.08 (0.07–0.08)	0.08 (0.07–0.08)	10.96 (10.16–11.82)
Agriculture Present (ref)	388	5.06 (4.64–5.51)	1.24 (1.15–1.33)	0.49 (0.45–0.53)	1.20 (1.09–1.32)	0.07 (0.07–0.08)	0.08 (0.07–0.08)	11.91 (11.28–12.58)
Land Use: Urban Development[c]								
0–25% Urban	183	5.06 (4.44–5.76)	1.14 (1.03–1.26)	0.48 (0.43–0.53)	1.11 (0.98–1.27)	0.07 (0.07–0.07)*	0.07 (0.07–0.07)	11.69 (10.79–12.66)
25–75% Urban	191	5.05 (4.47–5.69)	1.39 (1.25–1.55)	0.50 (0.45–0.55)	1.35 (1.18–1.55)	0.08 (0.07–0.08)*	0.08 (0.07–0.08)	12.27 (11.36–13.25)
More than 75% Urban (ref)	227	4.35 (3.86–4.90)	1.20 (1.09–1.32)	0.45 (0.41–0.49)	1.06 (0.93–1.20)	0.07 (0.07–0.08)	0.08 (0.07–0.08)	10.91 (10.15–11.73)
Smoking Staus								
Current	73	3.67 (2.97–4.53)*	1.01 (0.84–1.21)*	0.41 (0.34–0.50)	0.96 (0.77–1.20)	0.08 (0.07–0.08)	0.08 (0.07–0.08)	9.77 (8.64–11.04)*
Former	147	5.45 (4.78–6.20)*	1.34 (1.21–1.48)*	0.53 (0.47–0.59)	1.33 (1.16–1.52)	0.07 (0.07–0.08)	0.07 (0.07–0.08)	12.49 (11.49–13.57)*
Never (ref)	345	4.91 (4.47–5.40)	1.28 (1.18–1.39)	0.47 (0.44–0.51)	1.16 (1.05–1.29)	0.07 (0.07–0.08)	0.08 (0.07–0.08)	11.80 (11.11–12.52)
Missing	41	3.74 (2.71–5.15)*	1.02 (0.79–1.30)*	0.41 (0.32–0.52)	0.98 (0.71–1.36)	0.08 (0.07–0.09)	0.07 (0.07–0.08)	10.05 (8.46–11.93)*
Creatinine (mg/dL)								
<0.7	61	3.44 (2.81–4.22)**	0.92 (0.76–1.13)**	0.35 (0.29–0.44)*	0.83 (0.66–1.03)**	0.08 (0.07–0.08)	0.08 (0.07–0.08)	9.06 (7.98–10.27)**
0.7–1 (ref)	360	4.46 (4.06–4.89)	1.22 (1.13–1.32)	0.47 (0.43–0.51)	1.06 (0.96–1.17)	0.07 (0.07–0.08)	0.08 (0.07–0.08)	11.05 (10.45–11.67)
>1	184	6.09 (5.37–6.90)**	1.39 (1.26–1.54)**	0.53 (0.48–0.58)*	1.56 (1.36–1.79)**	0.08 (0.07–0.08)	0.08 (0.07–0.08)	13.72 (12.68–14.85)**
Eat Caught Fish								
Yes	264	5.66 (5.08–6.30)**	1.27 (1.17–1.39)	0.52 (0.48–0.57)*	1.27 (1.14–1.42)	0.08 (0.07–0.08)	0.07 (0.07–0.08)	12.76 (11.93–13.65)**
No (ref)	341	4.19 (3.82–4.60)	1.21 (1.12–1.31)	0.44 (0.40–0.47)	1.08 (0.98–1.20)	0.07 (0.07–0.08)	0.08 (0.07–0.08)	10.72 (10.12–11.35)
Popcorn Frequency								
Never (ref)	128	4.14 (3.46–4.95)	1.16 (1.02–1.32)	0.43 (0.38–0.49)	1.11 (0.93–1.33)	0.08 (0.07–0.08)	0.07 (0.07–0.08)	10.82 (9.78–11.97)
1 time/month	185	4.92 (4.32–5.60)	1.24 (1.10–1.40)	0.48 (0.43–0.54)	1.15 (0.99–1.32)	0.07 (0.07–0.08)	0.08 (0.07–0.08)	11.77 (10.85–12.78)
2–3 times/month	140	5.22 (4.56–5.98)	1.29 (1.14–1.45)	0.52 (0.46–0.59)	1.25 (1.07–1.46)	0.07 (0.07–0.08)	0.08 (0.07–0.08)	12.21 (11.14–13.39)
1 time/week	66	4.76 (4.05–5.60)	1.30 (1.16–1.45)	0.47 (0.41–0.55)	1.20 (0.99–1.44)	0.07 (0.07–0.08)	0.07 (0.07–0.08)	11.22 (10.06–12.51)
2–3 times/week	32	5.49 (4.42–6.81)	1.31 (1.07–1.62)	0.48 (0.40–0.57)	1.18 (0.87–1.61)	0.08 (0.07–0.08)	0.08 (0.07–0.08)	12.10 (10.47–13.98)
3–4 times/week	11	9.30 (6.24–13.86)	2.21 (1.68–2.90)	0.61 (0.45–0.84)	1.71 (1.03–2.82)	0.08 (0.07–0.08)	0.08 (0.07–0.08)	17.93 (13.19–24.35)
5+ times/week	11	6.35 (3.94–10.22)	1.38 (0.94–2.03)	0.49 (0.35–0.67)	1.24 (0.80–1.93)	0.07 (0.07–0.08)	0.08 (0.07–0.10)	13.27 (9.57–18.39)
Fast Food Frequency								
Never	32	6.34 (4.99–8.06)*	1.73 (1.47–2.04)*	0.66 (0.54–0.80)*	1.57 (1.24–1.99)	0.07 (0.07–0.08)	0.08 (0.07–0.08)	14.32 (12.23–16.78)*
Less than 1 time/month	149	5.59 (4.87–6.43)*	1.35 (1.20–1.53)*	0.53 (0.47–0.60)*	1.30 (1.11–1.53)	0.08 (0.07–0.08)	0.08 (0.07–0.08)	12.97 (11.85–14.18)*
1–3 times/month (ref)	232	4.74 (4.21–5.33)	1.22 (1.11–1.35)	0.47 (0.43–0.52)	1.13 (0.99–1.27)	0.08 (0.07–0.08)	0.08 (0.07–0.08)	11.48 (10.68–12.34)
1–2 times/week	134	4.14 (3.60–4.76)*	1.13 (1.01–1.28)*	0.42 (0.37–0.47)*	1.11 (0.95–1.29)	0.07 (0.07–0.08)	0.08 (0.07–0.08)	10.41 (9.56–11.33)*
3–4 times/week	41	4.03 (2.90–5.60)*	1.08 (0.82–1.43)*	0.39 (0.32–0.49)*	0.97 (0.69–1.35)	0.07 (0.07–0.08)	0.07 (0.07–0.08)	10.38 (8.68–12.43)*
5+ times/week	15	4.20 (2.78–6.33)*	1.09 (0.86–1.38)*	0.36 (0.29–0.46)*	1.01 (0.61–1.67)	0.08 (0.07–0.08)	0.07 (0.07–0.08)	10.09 (7.71–13.20)*

*$p < 0.05$, **$p < 0.001$.
[a]CI = Confidence Interval.
[b]GED = General Education Development.
[c] Land use proportions: proportion of land use within 1 mile of geocoded residence

study which examined PFAS by race among the US population using the NHANES sample. Non-Hispanic, White adults had significantly higher PFOA and PFHxS (Sonnenberg et al., 2023). In NHANES, Asian adults had the highest concentrations of PFNA and PFOS when compared to other racial groups, with non-Hispanic White adults following as next highest group. Due to the lack of geographical variation among those who self-identified as non-White in our study sample, it is difficult to assess the difference in PFAS exposure seen by race and is a limitation of this study. Hence, we were unable to examine Asian adults separately due to low sample sizes. Racial segregation in Wisconsin, particularly in Milwaukee (Wisconsin's largest city), could be driving this association (Loyd and Bonds, 2018). The higher PFAS serum levels among participants who self-identified as non-Hispanic White (not in combination with any other race) compared to those who self-identify as non-White could also be due to a variety of differences in dietary or consumer product exposure that stem from social, cultural, and other behavioral differences. While our study found White adults had higher PFOS levels, another study among NHANES adults found non-Hispanic Black participants had the highest adjusted mean of % n-PFOS, followed by Asian adults, significantly higher than those of non-Hispanic White adults (Kang et al., 2023). White women in SWAN-MPS were also found to have lower % n-PFOS compared to Black, Chinese, and Japanese women. PFHxS could be a tracer for exposure to PFAS in consumer products and some differences seen by PFHxS (and other PFAS and they are known to co-occur), could be partly driven by differences in exposure to consumer products (Chang et al., 2021; Hu et al., 2018; Sonnenberg et al., 2023). More research is warranted to examine differences in PFAS exposure by race and ethnicity and requires population-based samples which have larger sample sizes among different racial populations.

We found a small but consistent positive association between income and serum PFAS, which was significant after adjustment for other factors like race and land use near the home. A previous meta-analysis also found a consistent association between higher income and higher internal exposure to PFOS and PFOA (Buekers et al., 2018), though this runs counter to the environmental justice hypothesis, that those with lower income would have higher levels of environmental exposures. This may be explained by possible differences in diet or consumer product use.

We examined whether land cover near participants' residences was associated with PFAS exposure. This was a crude proxy measure for exposure via water supply contamination. We hypothesized there may be higher PFAS exposure living near agricultural fields that have the potential for PFAS-laden sludge as fertilizer, and hypothesized those living near high urban development may also have higher PFAS exposure due to the potential of living near industrial use and contamination via drinking water. While we did not observe large univariate differences in serum PFOA between Census urban and rural designations, after adjustment for covariates the percentage of urban development within 1 mile of the residence was a significant predictor of serum PFOA. Because of the use of PFCAs like PFOA in industrial applications, fast food packaging, and firefighting foams (Prevedouros et al., 2006), PFOA may be more prevalent in urban areas compared to rural areas, where industry and PFOA products are concentrated (e.g., fast food, airports, etc.). We did not find associations between PFAS serum levels and living near land designated as agriculture. Our findings align with a recent study that measured PFAS prevalence and source-tracing from shallow groundwater used for drinking in Wisconsin, where they found PFAS detections were more common in developed, urban areas (Silver et al., 2023), which is reflected in our findings. However, local septic system effluence was also significantly associated with PFAS detection in rural areas as opposed to agricultural sources, which we were unable to account for in our study. Furthermore, among the n = 450 samples they tested, only four samples (1%) had one or more PFAS above a Wisconsin public health value, and 19 (4%) had one or more PFAS above an EPA proposed Maximum Contaminant Level (Silver et al., 2023). This suggests groundwater may not always significantly contribute to population exposure. Our measure of environmental exposure to PFAS may have resulted in misclassification bias, inaccurately categorizing people's true exposure for both agricultural and urban land uses. More granular land use measures which can differentiate local septic system sources from other environmental sources may be necessary to elucidate geographic associations with PFAS exposure.

Previous literature found those who frequently ate microwave popcorn had higher levels of serum PFAS, and PFAS have been used in numerous types of food packaging (Susmann et al., 2019; Zabaleta et al., 2017). In our study, popcorn consumption frequency was associated with higher serum PFAS as well, but only in univariate analyses, suggesting a true association may be confounded by other demographic or behavioral factors. We also found Wisconsin adults who never eat fast food had higher levels of some PFAS, though these were not significant in multivariate models, also suggesting confounding by other factors, or due to the imperfect measurement of exposure. We relied on self-reported frequency of fast-food consumption without any information of the specific type of fast food or food packaging. The food frequency questionnaire asked about one's typical diet and was not a recent dietary recall, which may have resulted in misclassification bias. Dental floss is another product found to contain PFAS (Gaines, 2023). For instance, Oral-B glide, and other wax-coated floss, has been found to have PFHxS, and users of Oral-B glide had elevated serum levels of PFHxS (Boronow et al., 2019). In this study, more frequent dental floss use was also associated with elevated levels of several PFAS but was not a significant predictor of exposure for any PFAS. The survey questions asked in this study were not specific enough to examine different brands of wax vs. non-wax coated floss use, which may have resulted in bias towards the null. Self-reported flossing also may have been subject to social-desirability bias, also biasing results toward the null.

In univariate analyses, we found a positive association between creatinine levels and PFAS, which agrees with previous literature (Jain and Ducatman, 2022), which may indicate decreased kidney function (Blake et al., 2018). However, the association with creatinine was not significant after adjustment for covariates and was not included in the final model, which may indicate that the association between creatinine and serum PFAS is largely confounded by other factors in this sample.

Our study did not find statistically significant associations between smoking status and PFAS in adjusted models. Reporting of smoking had the most missing data in our study, and hence we retain them as their own category to reduce selection bias. However, when participants with missing smoking status were removed from analysis, statistically significant inverse associations were found with PFAS and smoking. It is possible a true association exists, but we were not able to capture it with our sample size and inability to correctly classify those with missing data. There have been mixed results regarding the association between smoking and PFAS. Several longitudinal studies have found inverse associations between smoking and serum PFOA (Fisher et al., 2016), PFOS (Kato et al., 2014; Fábelová et al., 2023; Tsai et al., 2018; Lauritzen et al., 2016), and PFUnDA (Fábelová et al., 2023; Shu et al., 2018), while others have found positive or null associations with smoking (Barton et al., 2020; Nyström et al., 2022). The reduction of thyroid-stimulating hormone (TSH) that occurs in smokers may play a role in how PFAS exposure is metabolized in the body (van Gerwen et al., 2020). Yet, the opposite association was found in midlife women highly exposed to firefighting foams, where current smokers had the highest serum PFAS, and never smokers had the lowest (Park et al., 2019; Barton et al., 2020). However, this could be explained by PFAS exposure being high enough among the smokers to override any thyroid hormone mechanistic offset in how the PFAS is being metabolized and processed. The missing data on smoking and our inability to adequately examine smoking and PFAS is a limitation of our study.

Study findings further support that fish consumption among this population-based sample living in and around the Great Lakes region is a significant route of PFAS exposure in humans. These findings are important in driving policy towards protecting higher risk anglers and

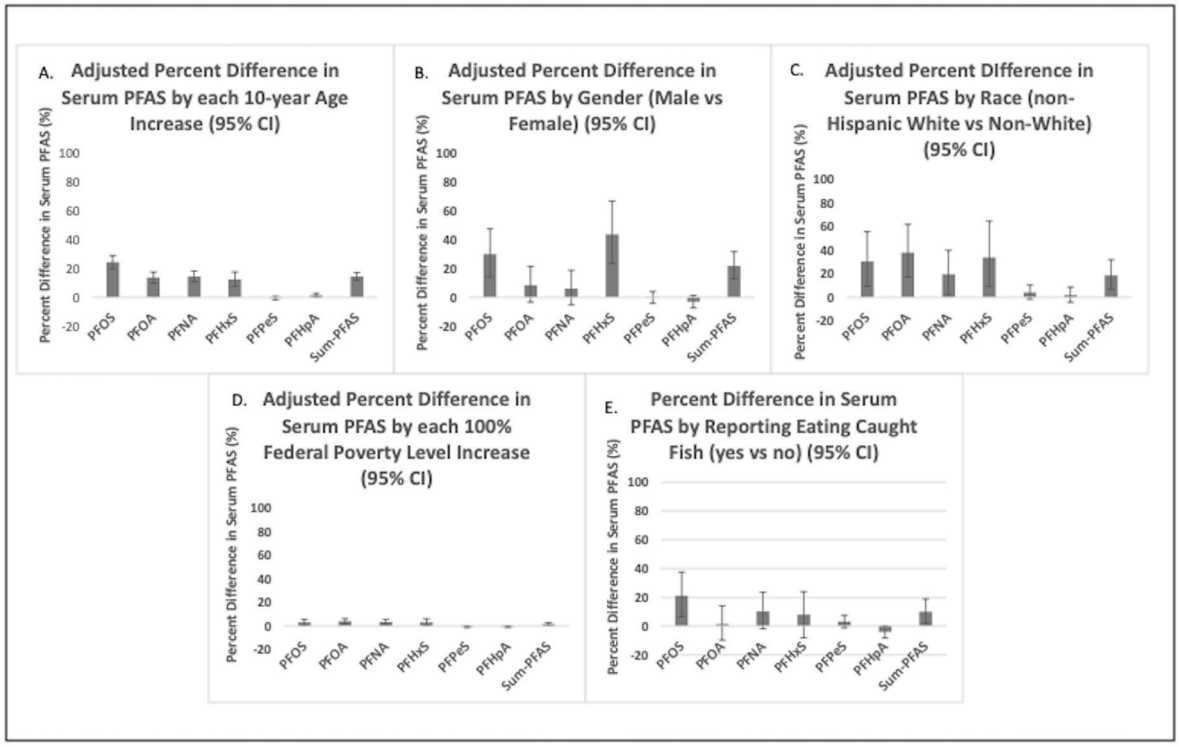

Fig. 1. Mutually adjusted differences in PFAS serum concentrations compared to the listed reference group (estimates and 95% confidence intervals) by demographics and characteristics (n = 577) from multivariate linear regression models for (A.) Age (10-year increase) (B.) Gender (Male vs. Female) (C.) Race (Non-Hispanic White vs. Non-White) (D.) Household Income (100% Federal Poverty Level increase) (E.) Eat Caught Fish (yes vs. no).

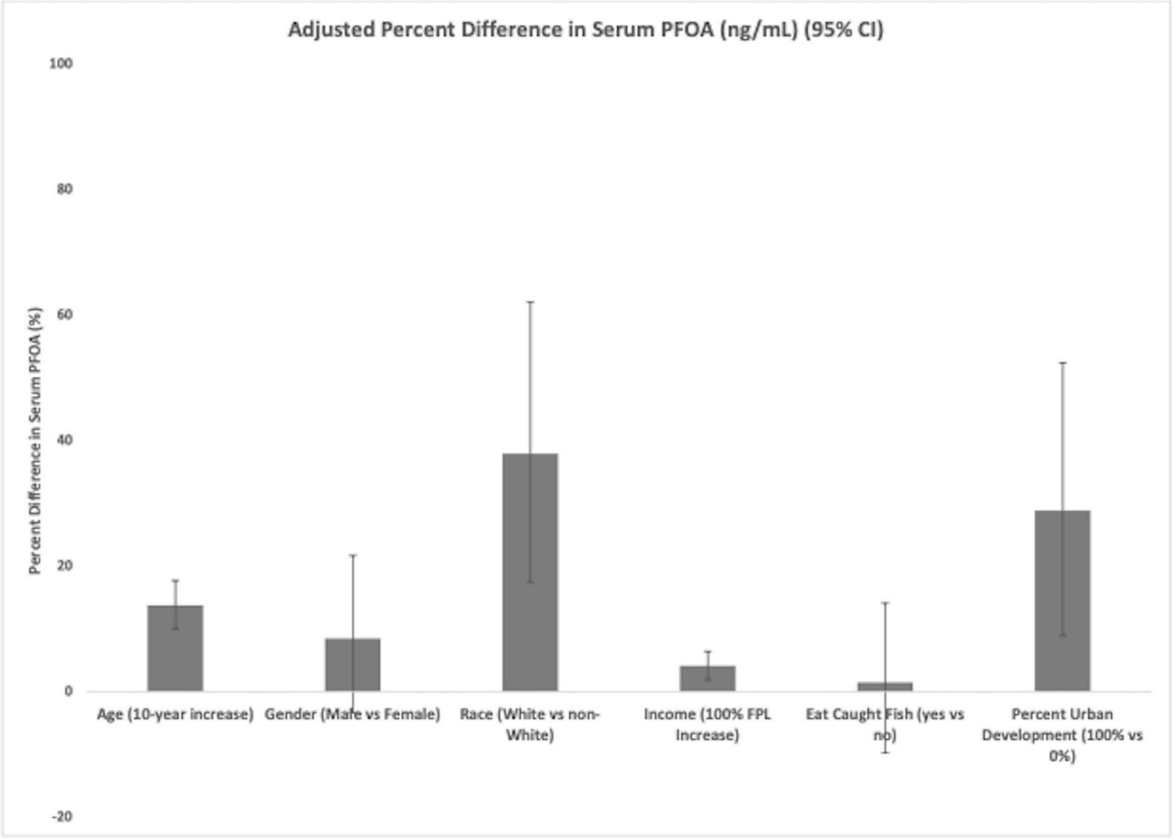

Fig. 2. Mutually adjusted percent differences in PFOA serum concentrations (estimates and 95% confidence intervals) by age, sex, race, eating caught fish, federal poverty level to income ratio, and percent urban development within one mile of the home (n = 577) from multivariate linear regression models. Comparisons are based on the listed reference group on the x-axis.

reflect the influence of bioaccumulation of PFAS in waterways that pose a potential threat to public health. Those who reported eating caught fish in the past year had elevated levels of several PFAS, even after adjustment for covariates, with the difference being the greatest for PFOS – 20% higher when compared to those who have not eaten caught fish in the last year. Several studies have found associations between fish consumption and serum PFAS, both commercially bought and self-harvested seafood and freshwater fish (Liu et al., 2022; Christensen et al., 2017; Denys et al., 2014). Both NHANES and SWAN-MPS populations showed a positive association with fish consumption and relative proportion of serum n-PFOS, although the fish consumption reported in these studies was not limited to caught fish or Great Lakes fish but included all commercially bought fish and shellfish consumption (Kang et al., 2023). Similar findings were seen among anglers in Milwaukee, Wisconsin, where median PFOS serum levels were two times higher than those seen by a comparable subpopulation within NHANES (Christensen et al., 2016a,b). A study of PFAS among 28 fish species from seven river systems and Lakes Michigan and Superior found PFOS to be present in the highest concentrations and in the highest number of samples (Williams and Schrank, 2016), affirming our current understanding of bioaccumulation PFOS presents compared to other PFAS compounds (Remucal, 2019) throughout the state and region. Taken together, anglers may present an effective group for public health intervention in Wisconsin when it comes to mitigating PFAS exposure, and exposure to other harmful chemicals. In 2021, the Wisconsin Department of Natural Resources issued 1.43 million fishing licenses (Smith, 2022), which reflects the strong culture of fishing in the state. Wisconsin anglers have been found to have elevated levels of other persistent contaminants in their blood, including PCBs (polychlorinated biphenyls) and mercury (Anderson et al., 2004; Christensen et al., 2016a,b; Peterson et al., 1994; Schantz et al., 2010). Additionally, subsistence fishing among indigenous tribes and other groups in Wisconsin may pose a public health concern (Peterson et al., 1994).

We compared PFAS serum levels among those whose primary water source is a private well versus municipal water supply. We did not observe significant differences in serum PFAS by water source. We were unable to account for differences in treatment mitigation efforts across private well users and municipal water supplies, because samples were collected prior to statewide PFAS regulations, which may explain why we did not see an association. A recent study by DeLuca et al. (2023) observed differences by drinking water source between tap and bottled water (DeLuca et al., 2023) but did not differentiate between those using private wells and community water supplies. Additionally, their study was only conducted among pregnant women and was not a statewide representative sample. The geographic distribution of our study sample did not include adults residing in very close proximity to known contamination sites in Wisconsin (i.e. Truax Military Site, JCI/Tyco plant, etc.) (PHMDC, 2022; MJS, 2024). Due to the variability in geographic location among both rural and urban residents in the sample, and the resulting variability in nearby land uses, it is not surprising to find minimal differences in serum PFAS by drinking water source.

This study has several strengths. First, the rich survey data is a particular strength, as relatively few health surveys include questions specifically asking about fish consumption, a known exposure route. Additionally, nuanced measurement of both long- and short-chain serum PFAS paints a richer picture of the PFAS exposure profile of this sample. Because SHOW does not have a specific exposure or outcome it investigates, social desirability bias related to exposures is less of a concern. Finally, this statewide subsample adds to our understanding of the determinants of human exposure to PFAS in a general population.

This study is not without limitations. Importantly, this data was collected cross-sectionally, which prevents us from identifying causality. Self-reported diet and health behavior covariates may be vulnerable to social desirability bias, as participants may report “healthier” behaviors than they practice. Memory-based dietary reports are known to be somewhat inaccurate, and do not directly reflect actual nutrient and energy consumption (Archer et al., 2015). Therefore, some dietary aspects, especially those relating to frequency or volume of consumption, may be biased. In addition, this study did not ask about commercial fish consumption. While commercial fish have been found to have lower levels of PFAS than caught fish, high levels of PFOS have been found in some species, like walleye and whitefish (Ruffle et al., 2020). If those who eat caught fish are also more likely to eat commercial fish, associations of PFAS and caught fish may be inflated and partly explained by exposure to commercial fish. There may be other dietary routes of exposure uncaptured via the dietary questionnaire that may account for residual confounding in our models. Unfortunately, we were not able to assess determinants of all compounds analyzed by WSLH due to low detection rate, as any compound with less than 50% detection rate was excluded from analyses. However, all compounds were combined into a summed measure. This summed measure is an imperfect crude proxy estimate of total PFAS burden as the first step towards characterizing total body burden of 38 PFAS in a general population sample (Sznajder-Katarzyńska et al., 2019; Sinclair et al., 2020) It is still unclear how to model total body burden and whether there are interactive, additive, or multiplicative effects.

As mentioned earlier, lack of racial representation with variable geography led to limitations in our ability to fully understand associations with race, which may be confounded by environmental, cultural, or social differences in behavior that lead to varying PFAS exposure. Additionally, our ability to assess environmental exposure via land use near residences was limited. Our proxy measure of exposure via agricultural fertilizer or urban sources was crude and subject to misclassification bias. Lastly, in this study we do not account for county level clustering, and we are limited in statewide geographic representativeness due to the SHOW sampling frame in 2014–2016 which focused on 10 primary counties in the state. While one-third the sample resided in a rural area, defined by the U.S. Census, which proportionately aligns with overall percent rural of the state population, only 10% of the sample was from the North region of the state and only 10% from the South region of the state. This could have biased findings away from the null, potentially finding associations where there are none. However, since no strong associations were found based on differing geographies within the sample, we believe this had a minimal effect on our findings. However, future sampling should be conducted on more geographically representative samples in Wisconsin, including more northern counties. This study will increase awareness of PFAS determinants in Wisconsin and can inform public health interventions at the state level.

5. Conclusions

Overall, demographic, behavioral, and environmental predictors of PFAS exposure were similar among a statewide, population-based sample of adults in Wisconsin. Age, race, and gender, as well as modifiable risk factors like eating caught fish presented differences in serum PFAS and were associated with higher PFAS body burden. Our findings confirmed prior and continuing biomonitoring findings among anglers in the state (Christensen et al., 2016a,b; He et al., 2022). However, future targeted biomonitoring around sites of known groundwater contamination could better characterize persons at potentially high levels of exposure. It is important to consider the health implications of alternative food that would replace caught fish in a diet if one were to reduce their fish consumption to reduce their PFAS exposure. Future work should include total organic fluorine (TOF) as a measurement to better understand total body burden of PFAS, not only in association with potential routes of exposure, but potential adverse health outcomes. Additionally, longitudinal follow-up should be conducted with an eye toward sources of exposure that include additional dietary, consumer product, and occupational exposure that may help use better understand differences seen in PFAS levels by sex, income, and race, and those levels change over time.

Ethical Approval

The SHOW protocol and informed consent documents were approved by the Health Sciences Institutional Review Board of The University of Wisconsin-Madison (protocol code 2013-0251). Participants in SHOW gave consent to their information being used for research prior to this study.

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Manuscript conflict of interest declaration

The authors declare they have no conflicts of interest related to this work to disclose.

CRediT authorship contribution statement

Rachel Pomazal: Writing – review & editing, Writing – original draft, Visualization, Methodology, Formal analysis. **Kristen Malecki:** Writing – review & editing, Supervision, Resources, Project administration, Investigation, Funding acquisition, Conceptualization. **Noel Stanton:** Writing – review & editing, Data curation. **Brandon Shelton:** Data curation. **Meshel Lange:** Methodology, Data curation. **Roy Irving:** Writing – review & editing, Funding acquisition. **Jonathan Meiman:** Writing – review & editing, Funding acquisition. **Christina K. Remucal:** Writing – review & editing, Supervision. **Amy Cochran:** Supervision, Methodology. **Amy A. Schultz:** Writing – review & editing, Supervision, Methodology, Investigation, Funding acquisition, Conceptualization.

Declaration of competing interest

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

Data availability

The data that has been used is confidential.

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Appendix A. Supplementary data

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