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Detection of long chain *per-* and polyfluoroalkyl substances (PFAS) in the benthic Golden tilefish (*Lopholatilus chamaeleonticeps*) and their association with microscopic hepatic changes

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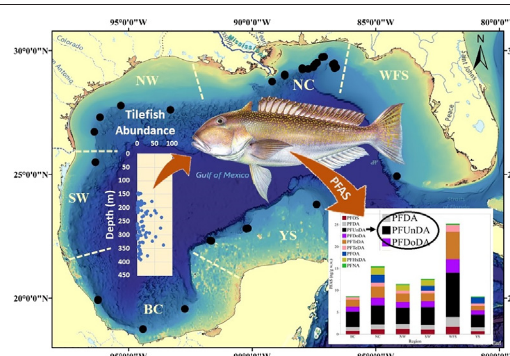
HIGHLIGHTS

- First report of PFAS in deep (≤400 m) waters in the Gulf of Mexico (GoM)
- Nine long-chain PFAS were detected in Golden Tilefish
- Tilefish had a unique PFAS profile dominated by PFUnDA
- Regional differences within the GoM were detected in Tilefish PFAS concentrations
- Significant associations were found between PFAS and microscopic hepatic changes

GRAPHICAL ABSTRACT



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ABSTRACT

Oceans are major sinks for anthropogenic pollutants, including *per-* and polyfluoroalkyl substances (PFAS). Although PFAS have been detected in surface waters globally, this is the first report of PFAS in a deep (170–400 m) demersal species in the Gulf of Mexico (GoM). Golden Tilefish (*Lopholatilus chamaeleonticeps*) plasma extracts ($n = 185$) were investigated for the presence of PFAS using ultra-high performance liquid chromatography-tandem mass spectrometry. A subset of liver tissues ($n = 51$) were also analyzed for microscopic hepatic changes (MHCs). Overall, nine of the 110 PFAS targeted were detected in Tilefish plasma at relatively high frequencies. Plasma concentrations of total PFAS (Σ_9PFAS) ranged from below the detection limit to 27.9 ng g^{-1} w.w. Significant regional differences were observed with the highest concentrations of PFAS detected in the north central region of the GoM, where substantial industrialization and discharges from the Mississippi River occur. Compared to most wildlife and matrices analyzed globally, the PFAS profiles in Tilefish were unique as they are dominated by PFUnDA. Profile differences are hypothesized to be the result of Tilefish's distinctive lifestyle, habitat, diet, and partitioning characteristics of long-chain PFAS. Several MHCs were identified in this

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chromatography – tandem mass spectrometry
Demersal
Fish
Long-chain PFCA

subset of Tilefish that could be detrimental to their health. Significant correlations between PFAS concentrations and biometric indices and MHCs were evident, however, additional research is needed to investigate the role PFAS and PFAS combined with chemical admixtures may play in inducing observed hepatic changes and other physiological effects in Tilefish. These findings give insight into the fate of PFAS at depth in aquatic ecosystems and are cause for concern regarding the health of other deep water benthic biota in GoM and other deepwater sinks for PFAS.

1. Introduction

Per- and polyfluoroalkyl substances (PFAS) are a highly complex family of more than 9000 anthropogenic chemicals in which all hydrogens on at least one (polyfluoroalkyl) or all (perfluoroalkyl) carbons, have been replaced by fluorine (Buck et al., 2011; Flaherty et al., 2005; Mueller and Yingling, 2017). PFAS exist as short-chain, typically containing up to seven carbons, and long-chain compounds, typically containing six or more carbons for perfluoroalkane sulfonates and seven or more carbons for the perfluoroalkyl carboxylic acids, with a chain length of eight being the most common, for example perfluorooctane sulfonic acid (PFOS)(Buck et al., 2011; Taniyasu et al., 2008; USEPA, 2009). Due to the strength of the carbon-fluorine bond and diverse chemical moieties, PFAS have a wide range of chemical and physical properties including oil, water, stain, and soil repellency, high chemical and thermal stability, enhanced surfactant characteristics, and the ability to provide friction reduction to a range of products (Beecher et al., 2018; Flaherty et al., 2005; Kissa, 2001; Mueller and Yingling, 2017). Since the 1950s, these properties have led to PFAS being applied in a broad range of applications, including: aviation, aerospace, emergency response, and military via aqueous film forming foams (AFFFs), biocides, cosmetics/personal care products, electronics, food processing, household and medical products, metal plating, oil production, mining, paper and packaging, photolithography, construction, semiconductors, and textiles (Beecher et al., 2018; Calafat et al., 2007; Mueller and Yingling, 2017).

The chemical stability of PFAS largely due to the strong carbon-fluorine bond renders them relatively metabolically inert, nonbiodegradable, environmental persistent and easily absorbed by organisms (Lau et al., 2007). As a result, a broad range of PFAS are detected in the environment, wildlife, and humans, and many PFAS exhibit a myriad of toxicological effects – posing great concerns for adverse health effects (Beecher et al., 2018; Buck et al., 2011; Calafat et al., 2007; Flaherty et al., 2005; Mueller and Yingling, 2017). Moreover, the long-chain PFAS have been found to be more bioaccumulative than their short-chain analogues (Buck et al., 2011; Palmer et al., 2019). In contrast to legacy lipophilic contaminants (e.g., polychlorinated biphenyls), PFAS are considered proteinophilic due to their high affinity to bind to proteins, particularly in plasma, and their phospholipid partitioning (Conder et al., 2008; Pizzurro et al., 2019). Research has demonstrated the highest levels of PFAS are generally measured in blood/serum/plasma, followed by liver and kidneys (Bangma et al., 2017; Pizzurro et al., 2019; Shi et al., 2020). It as has been shown for humans and most wildlife species (aquatic and terrestrial) examined to date across various habitat types (Flaherty et al., 2005; Forsthuber et al., 2020; Houde et al., 2006a; Jones et al., 2003; Mueller and Yingling, 2017; Ng and Hungerbühler, 2013). However, it is the physiochemical properties (e.g., chain-length, solubility, log Kow) and species-specific pharmacokinetics that likely influence their environmental fate and bioaccumulation potential.

More importantly, PFAS exposure/burden in humans and animals have been linked to hepatic, immunological, developmental, endocrine, reproductive, and cardiovascular effects, as well as several types of cancer (Beecher et al., 2018; Bell et al., 2021; Fenton et al., 2021; Gagliano et al., 2020; Mueller and Yingling, 2017; Pelch et al., 2019; Sunderland et al., 2019). In vitro studies have revealed PFAS exposure can induce cytotoxicity, apoptosis, oxidative stress, DNA damage, modulated immune function, inflammations and interfere with liver enzyme activities (as reviewed in Ojo et al., 2020).

From an aquatic perspective, the major sources of PFAS release are from industrial facility effluents (that produce or utilize PFAS), entities that employ AFFFs during emergency response or training (e.g., airports, military, fire training facilities), waste management effluent and residuals, and biosolids production and application (Mueller and Yingling, 2017). Once released into the aquatic environment, PFAS exposure is typically the result of direct uptake by organisms in the water column or via diet (where bioaccumulation can occur trophically)(Martin et al., 2003). Previous studies in the Gulf of Mexico (GoM) have documented PFAS in the plasma or tissues of Bottlenose Dolphins (*Tursiops truncates*), West Indian Manatees (*Trichechus manatus*), Hawksbill Sea Turtles (*Eretmochelys imbricata*), Leatherback Sea Turtles (*Dermochelys coriacea*), and tissues from Striped Mullet (*Mugil cephalus*), Pigfish (*Orthopristis chrysoptera*), Sheepshead (*Archosargus probatocephalus*), Pinfish (*Lagodon rhomboides*), and Spotted Seatrout (*Cynoscion nebulosus*) (Houde et al., 2006a; Houde et al., 2005a; Keller et al., 2012; Lynch et al., 2019; Palmer et al., 2019). Across these marine species, six PFAS (i.e., PFOS, PFNA, PFDA, PFDoDA, PFHxS, FOSA) consisting of long chains were consistently detected in different matrices (e.g., plasma and whole fish) across all studies (Table S1; full chemical nomenclature can be found in Table S2). These previous studies have demonstrated that secondary consumers (e.g., omnivores and carnivores, such as marine fishes and Leatherback Sea Turtles) and primary consumers (e.g., herbivores, such as West Indian Manatees), generally have lower concentrations of PFAS compared to tertiary consumers (e.g., top level carnivores, such as dolphins) (Houde et al., 2006a; Houde et al., 2005a; Keller et al., 2012; Lynch et al., 2019; Palmer et al., 2019). The detection and confirmation of PFAS exposure in wildlife in Florida coastal waters has largely focused on samples from estuaries along the west coast, including Sarasota Bay and Crystal River (Houde et al., 2006b; Houde et al., 2005b; Palmer et al., 2019).

But what of the fate and effects of PFAS chemicals that may be transported offshore and deposited on the continental shelves and beyond? There are currently no reports of PFAS in the sediments along the continental shelf of the GoM. If PFAS have reached the offshore areas of the GoM, it might be expected to find higher concentrations of long-chain PFAS in the sediments based on their partitioning characteristics (Zhao et al., 2012; Zhao et al., 2016). Scientific data regarding offshore demersal species in deep waters on the continental shelf of the GoM are nonexistent to our knowledge, with the exception of Leatherback Sea Turtles, dwelling in the upper depths with periodic deep dives for short durations (Houde et al., 2006a; Keller et al., 2012). Previous studies have demonstrated that other organic pollutants, such as polychlorinated biphenyls (PCBs), organochlorine pesticides (OCPs) and polycyclic aromatic hydrocarbons (PAHs), have reached the offshore habitats of the GoM (Carr, 2020; Kennicutt, 2017; Kennicutt et al., 1988; Pulster et al., 2020b; Snyder, 2020).

A common demersal fish along the western Atlantic and GoM continental shelf is the Golden Tilefish (*Lopholatilus chamaeleonticeps*), a nonmigratory, secondary consumer that opportunistically feeds on a wide variety of invertebrates and fishes (Freeman and Turner, 1977). In the GoM, Tilefish have been caught along the interface of the continental shelf and slope in depths of 150–400 m with water temperatures between 9 and 18 °C (Murawski et al., 2018b; Nelson and Carpenter, 1968). Adult Tilefish build characteristic, large funnel-shaped burrows in the seafloor by oral excavation of sediment, which they use as long-term habitat and protection from predators (Able et al., 1982; Grimes et al., 1986). Some of the highest levels of PAHs and mercury have been documented in northern Golden Tilefish which is a suspected result of their burrowing lifestyle and contaminated sediments (Perrot et al., 2019; Snyder, 2020; Snyder et al., 2020; Snyder et al., 2019; Snyder et al., 2015). Tilefish are considered sedentary and non-migratory, foraging close to their burrow communities and demonstrate high site fidelity (Grimes et al., 1986). Due to their specific habitat requirements and sedentary nature, Tilefish are unable to alter their distribution making them highly vulnerable to exploitation, environmental disturbances, and anthropogenic pollution (Grimes and Turner, 1999).

Based on the life history characteristics of Golden Tilefish, combined with the partitioning characteristics and toxic effects of PFAS, we hypothesize that Golden Tilefish may have elevated levels of long-chain PFAS with potential associations with microscopic hepatic changes (MHCs). This study aimed to investigate these hypotheses by determining the presence and concentration of PFAS in the plasma of Golden Tilefish in an effort to determine if PFAS have reached deep waters on the continental shelf of the GoM. Additionally, strengths of relationships between plasma concentrations and previously reported MHCs were evaluated. Plasma from Golden Tilefish collected in the GoM was analyzed using a newly developed 96-well plate extraction technique coupled to a ultra-high performance liquid chromatography-tandem mass spectrometry (LC-MS/MS) workflow (Da Silva et al., 2020).

2. Materials and methods

2.1. Golden tilefish plasma

Golden Tilefish (herein referred to as Tilefish) were collected during comprehensive Gulf-wide surveys conducted by the Center for the Integrated Modeling and Analysis of the Gulf Ecosystem (C-IMAGE). Demersal surveys were conducted from 2011 to 2017 using 5-nautical mile sets of demersal long lines previously described in detail (Murawski et al., 2018a). Tilefish were sampled at depths ranging from 170 to 400 m, with a mean depth of 260 m. As soon as Tilefish were landed, biometrics recorded and sacrificed following Institutional Animal Care and Use Committee (IACUC) approved protocols (IACUC protocol # W IS00000515). Within 15 min of landing, 2–4 mL of whole blood was collected in 4 mL lithium heparin tubes (BD Vacutainer, Franklin Lakes, NJ) from each specimen via caudal dissection, and tubes were placed on ice. Within one hour of collection, plasma was separated from the whole blood by centrifuging at 1600 ×g for 15 min and the supernatant stored at −20 °C, prior to long-term storage at −80 °C (Deak, 2020). Tilefish plasma samples ($n = 185$) collected between 2015 and 2017 were used in this study. The Tilefish in this study were caught six different regions in the Gulf of Mexico, defined a priori: West Florida Shelf (WFS), north central (NC), northwest (NW), southwest (SW), Bay of Campeche (BC) and the Yucatan Shelf (YS; Fig. 1). The percent lipid for individual liver tissues were quantified gravimetrically using a modified Folch method (Matyash et al., 2008) previously described in detail (Snyder et al., 2019).

2.2. PFAS extraction (96-well plate) methodology

The extraction of PFAS from Tilefish plasma used 96-well plate techniques adapted from Da Silva et al. (2020). Prior to extraction, plasma samples were thawed to room temperature and weighed before and after adding into each well of the 96-well plate to determine the mass of the sample. Weights were used to normalize the PFAS concentrations to wet weight (w.w.) of plasma. Extracts were performed by adding 100 μL of plasma to the well-plate, followed by spiking with 10 μL of an internal standard mixture with a target final concentration ranging between 1 and 4 ng mL^{-1} (depending on the PFAS). Samples were then mixed on the plate for 5 min at 400 rpm using a microplate vortexer (Fisher Scientific, Waltham, MA, United States), 390 μL of methanol (Optima LC-MS, Fisher Scientific, Waltham, MA, United States) were added and vortexed on the plate for 5 min at 450 rpm, the well plate was centrifuged for 15 min at 500 ×g, followed by the placement of a micromat on the well plate. The total volume in each

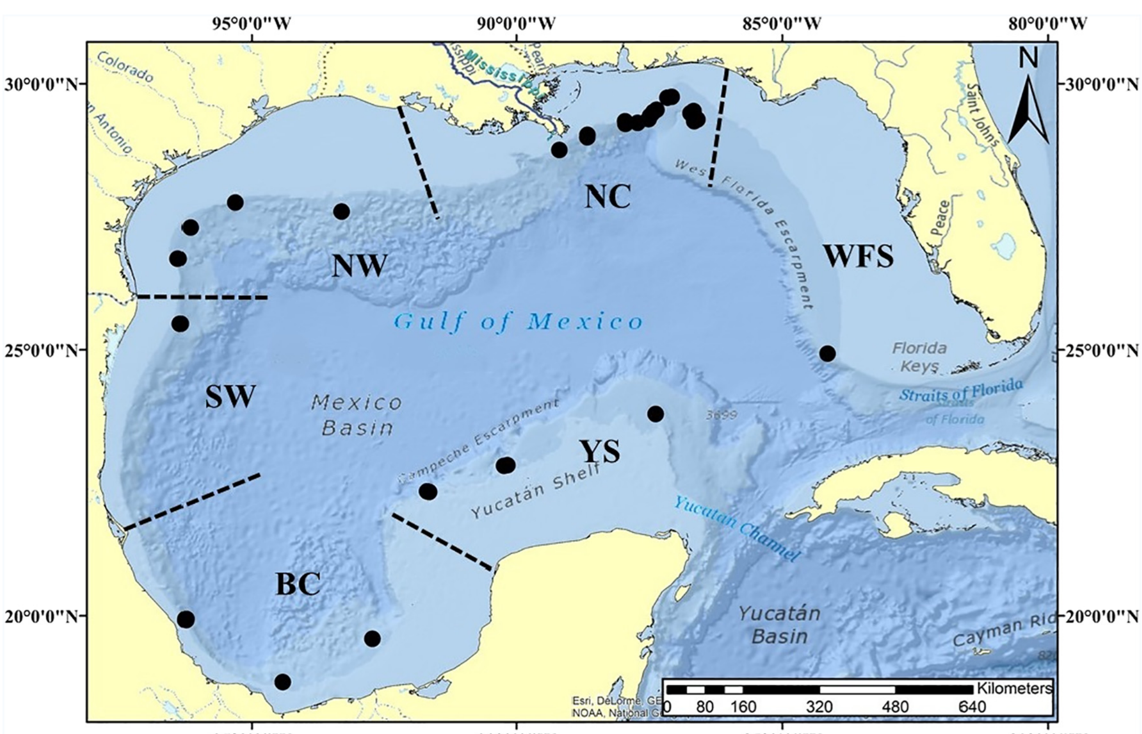

Fig. 1. (Use color in print). Golden Tilefish collection locations (●) in the Gulf of Mexico, 2015–2017.

well was 500 μL. The supernatant was transferred to a clean well plate and then directly seated in the autosampler of a Vanquish ultra-high performance liquid chromatograph (UHPLC, Thermo Scientific, San Jose, CA, United States) autosampler and the injection depth was set to 5 mm (Da Silva et al., 2020).

2.3. Target PFAS analysis

A Thermo Scientific Vanquish UHPLC coupled to a TSQ Quantis triple quadrupole mass spectrometer (UHPLC/MS/MS) was operated using electrospray ionization (in negative mode) to scan for 110 target PFAS analytes (Table S2) in Tilefish plasma extracts. Prior to analysis, the instrument was fitted with a delay column (Acclaim™ 120 C18, 2.3 × 50 mm, 5 μm, 120 Å), and PFAS-free peek tubing to eliminate PFAS artifacts. The UHPLC/MS/MS operating conditions and acquisition parameters can be found in Tables S3–S4.

2.4. Quality control (QC)

Xcalibur Quan Browser software V4.1 (Thermo Fisher Scientific) was used to perform data acquisition and peak integration. Detected PFAS were quantified by integrating the area under the curve defined by each peak related to the most intense transition (quantifier). A sum of isomers is presented for perfluorohexyl sulfonate (∑PFHxS), *N*-methylperfluoro-1-octanesulfonamidoacetic acid (∑N-MeFOSAA), and N-ethylperfluoro-1-octanesulfonamidoacetic acid (∑N-EtFOSAA). These three compounds were targeted as isomeric mixtures. For those compounds for which a related mass-labeled internal standard was not available, the closest IS in retention time was selected for quantitation (Da Silva et al., 2020). A linear regression model and the calibration curve for each identified compound was manually generated (the intercept, slope, correlation coefficient, limit of detection, LOD, and limit of quantification, LOQ, are described in Table S5). Additionally, the calibration curve and internal standard concentrations of all target PFAS (mass-labeled and non-labeled) that were targeted in this study can be found in Table S6. A total of 14 calibration curve levels, ranging from 0.0073 to 109.21 ng mL^{-1}, were prepared via successive dilution with three primary stock solutions and spiked internal standard (IS). This study used serial dilution of calibration curve levels to obtain the LOD (signal-to-noise >3) and LOQ (signal-to-noise >10) values; the values obtained confirmed by replicates of seven injections for each concentration observed.

Blank extractions were used to investigate background PFAS levels in the 96-well plate. None of the target PFAS were detected in any of the seven wells used as blanks. Additionally, other QC measures followed those by Da Silva et al., 2020. Briefly, method accuracy was investigated using National Institute Standards and Technology (NIST) Standard Reference Material 1950 (SRM-1950) – Metabolites in Frozen Human Plasma. Experimental values for six PFAS were within the range for acceptable accuracy (85–126%) compared to the SRM 1950 reference values (Table S7).

2.5. Liver histopathology

Histological evaluation was performed by a board-certified veterinary pathologist (Fishhead Labs, LLC, Stuart, FL) on liver tissues from Tilefish collected in the GoM. These samples were previously evaluated for relationships between microscopic hepatic changes (MHCs) and hepatic concentrations of PAHs, biometrics (i.e., age, sex, length, weight, organ weights, somatic indices and condition factor) and spatiotemporal patterns (Snyder, 2020). These relationships are not presented in this manuscript. The plasma from a subset of these same fish from the NC ($n = 29$), NW ($n = 5$) and BC ($n = 17$) regions were included in this study and analyzed for PFAS ($n = 51$, 2015–2016). Relationships between the previously described MHCs and plasma PFAS concentrations are explored herein. Comprehensive details on the histological evaluation are provided in Snyder, 2020.

2.6. Statistical analysis

All statistical analyses were performed using JMP Pro Version 16.0.0 (SAS Institute Inc.) and MATLAB R2021a (MathWorks) with the Fathom Toolbox (Jones, 2017). Regional and temporal differences in mean Σ_9PFAS in plasma extracts were assessed using a modified permutational multivariate analysis of variance (PERMANOVA) and pair-wise modified PERMANOVA with 1000 iterations (Anderson et al., 2017). The level of significance was set at $\alpha = 0.05$ and adjusted using the Bonferonni correction to protect against Type I error. To evaluate associations between PFAS concentrations and biometric parameters, permutation-based Pearson's correlations were calculated using biometrics and Σ_9PFAS or individual PFAS concentrations. To determine if PFAS influenced body condition of Tilefish, the strength of the relationship was evaluated between Σ_9PFAS or individual PFAS concentrations and Fulton's Condition Factor (K, K = 100 × total body weight /

Table 1
Mean (standard deviation) concentrations (ng g^{-1} wet weight) by region and collection year for the total of nine PFAS analytes ($\sum_9$PFAS) and individual compounds.

Gulf of Mexico region	Collection year	Sample size (n)	$\sum_9$PFAS	PFOS	PFDA	PFUnDA	PFDoDA	PFTrDA	PFTeDA	PFOA	PFHxDA	PFNA
Bay of Campeche (BC)	2015	18	8.05 (2.60)	0.77 (0.40)	0.88 (0.12)	3.51 (0.89)	1.13 (0.29)	1.59 (0.92)	0.58 (0.13)	<LOD	<LOD	0.20 (0.02)
North central (NC)	2015	49	13.0 (6.09)	1.16 (0.58)	1.12 (0.26)	4.86 (2.11)	1.96 (0.89)	2.93 (2.18)	0.95 (0.45)	<LOD	1.31 (0.02)	0.19 (0.06)
	2017	53	10.9 (4.33)	1.07 (0.81)	1.09 (0.26)	3.95 (1.55)	1.60 (0.64)	2.29 (1.85)	0.82 (0.31)	1.79 (1.15)	2.06 (0.99)	0.42 (0.81)
	2015–2017	120	11.9 (5.30)	1.11 (0.71)	1.10 (0.26)	4.37 (1.87)	1.75 (0.78)	2.59 (2.02)	0.88 (0.39)	1.79 (1.15)	1.61 (0.65)	0.31 (0.58)
Northwest (NW)	2016	21	9.78 (4.55)	1.21 (0.69)	1.10 (0.26)	3.75 (1.67)	1.30 (0.49)	1.91 (1.52)	0.66 (0.23)	<LOD	1.29	0.23 (0.11)
Southwest (SW)	2016	21	10.1 (3.30)	1.05 (0.58)	1.06 (0.22)	4.06 (1.38)	1.38 (0.41)	1.75 (0.99)	0.61 (0.13)	1.12	1.36	0.25 (0.19)
West Florida Shelf (WFS)	2017	1	25.3	1.69	2.25	10.1	3.11	6.24	1.59	<LOD	<LOD	0.29
Yucatan Shelf (YS)	2015	12	6.72 (2.83)	0.63 (0.19)	0.93 (0.05)	3.10 (0.89)	1.08 (0.20)	1.29 (0.64)	0.57 (0.06)	<LOD	<LOD	0.16 (0.01)
	2016	10	5.29 (3.17)	0.77 (0.59)	0.89 (0.18)	2.55 (1.13)	0.91 (0.29)	0.72 (0.68)	0.51 (0.13)	1.36 (0.49)	<LOD	0.26 (0.15)
	2015–2016	22	6.07 (3.01)	0.68 (0.35)	0.92 (0.12)	2.85 (1.02)	1.02 (0.25)	1.03 (0.71)	0.55 (0.09)	1.36 (0.49)	<LOD	0.22 (0.12)
Gulf-wide	2015–2017	185	10.4 (5.06).	1.11 (0.63)	1.11 (0.25)	3.92 (1.59)	1.54 (0.69)	2.13 (1.77)	0.78 (0.35)	1.54 (0.82)	1.51 (0.56)	0.23 (0.19)

total length[3])(Froese, 2006). The strength of relationships was considered very weak ($r < 0.30$), weak ($r = 0.30$ to 0.50), strong ($r = 0.50$ to 0.70), or very strong ($r > 0.70$).

3. Results

3.1. PFAS levels in tilefish plasma

Nine of the 110 targeted PFASs were detected in Tilefish plasma, therefore concentrations are reported as the sum of the nine detected PFAS ($\sum_9$PFAS ng g^{-1} w.w.). Nine long-chain PFAS were detected above the method LOQ in Tilefish plasma samples, these included one perfluoroalkyl sulfonic acid (PFSAs), known as PFOS, and eight perfluoroalkyl carboxylic acids (PFCAs). The remaining 101 PFAS analytes were not detected above instrument sensitivity in any of the Tilefish plasma samples. Five of the 185 Tilefish plasma samples analyzed did not have detectable levels (<LOD) of any target PFAS.

The PFCAs dominated Tilefish plasma profiles (69–100%), whereas the PFOS, accounted for a relatively minor amount of profile compositions (0–31%). Detection rates were highest for PFUnDA (97%) and PFTrDA (97%), followed by PFDoDA (92%), PFOS (87%), PFTeDA (82%), PFDA (72%), PFNA (41%), PFHxDA (4%) and PFOA (3%). PFAS ($\sum_9$PFAS) concentrations ranged from <LOD to 27.9 ng g^{-1} w.w. with a Gulf-wide mean (± SD) of 10.1 ± 5.28 ng g^{-1} w.w. (Table 1, Fig. 2). Plasma concentrations of individual PFAS ranged from < LOD to 9.30 ng g^{-1} (w.w.), with PFUnDA having the highest mean concentration followed by PFTrDA > PFOA > PFDoDA = PFHxDA > PFDA > PFOS > PFTeDA > PFNA (Table 1, Fig. 3). Concentrations generally increased with carbon chain length until PFUnDA, with a carbon chain length of 11 (C11) and then concentrations declined (Fig. 4).

3.2. Relationships between PFAS and biometrics

The Σ_9PFAS and the majority of the individual PFAS detected (with the exception of PFOS and PFNA) had statistically significant but weak correlations with total length, total weight and organ weights (Table S8). There was a very weak correlation between PFDA and liver lipid fraction. There were no significant correlations between individual PFAS or Σ_9PFAS and condition factor. A weak correlation was detected between PFNA and gonadosomatic index (GSI, $r = 0.38$, $p = 0.01$, Table S8). There were no significant differences in Σ_9PFAS or individual PFAS concentrations by sex.

3.3. Spatiotemporal differences

Regional differences Gulf-wide were detected in the Tilefish plasma concentrations (Fig. 5). Mean (±SD) plasma concentrations of Σ_9PFAS (ng g^{-1} w.w.) in Tilefish collected in the NC region were significantly higher than those collected in the BC ($p = 0.030$) and on the YS ($p = 0.015$). Plasma concentrations in Tilefish collected in the SW region were also higher than those collected on the YS ($p = 0.015$).

A general linear model ($p < 0.0001$) evaluating the effect of region, year and sex on PFAS concentrations indicated region ($p = 0.0015$) was the only significant parameter. Regional differences between individual PFAS concentrations (ng g^{-1} w.w.) were also detected (Table S9). Concentrations of PFDA in Tilefish plasma from the NC region were higher than those collected in the BC ($p = 0.015$) and YS ($p = 0.015$) regions. Tilefish collected on the WFS had PFUnDA levels up to 3.5 times higher than all other regions; however, the single fish collected on the WFS precluded statistical comparisons. The next highest levels of PFUnDA were found in Tilefish collected in the NC region which were significantly higher than those measured in Tilefish from the YS ($p = 0.015$). Tilefish from the NC region also had significantly higher PFDoDA compared to the BC ($p = 0.015$) and YS ($p = 0.015$) regions. Plasma concentrations of PFDoDA in Tilefish from the SW region were also significantly higher than those collected along the YS ($p = 0.030$). Plasma concentrations of PFTrDA in the NC region were also significantly higher than from the YS ($p = 0.015$). PFTeDA concentrations were significantly higher in Tilefish collected in the NC region than from the SW ($p = 0.015$), BC ($p = 0.015$) and YS ($p = 0.015$) regions.

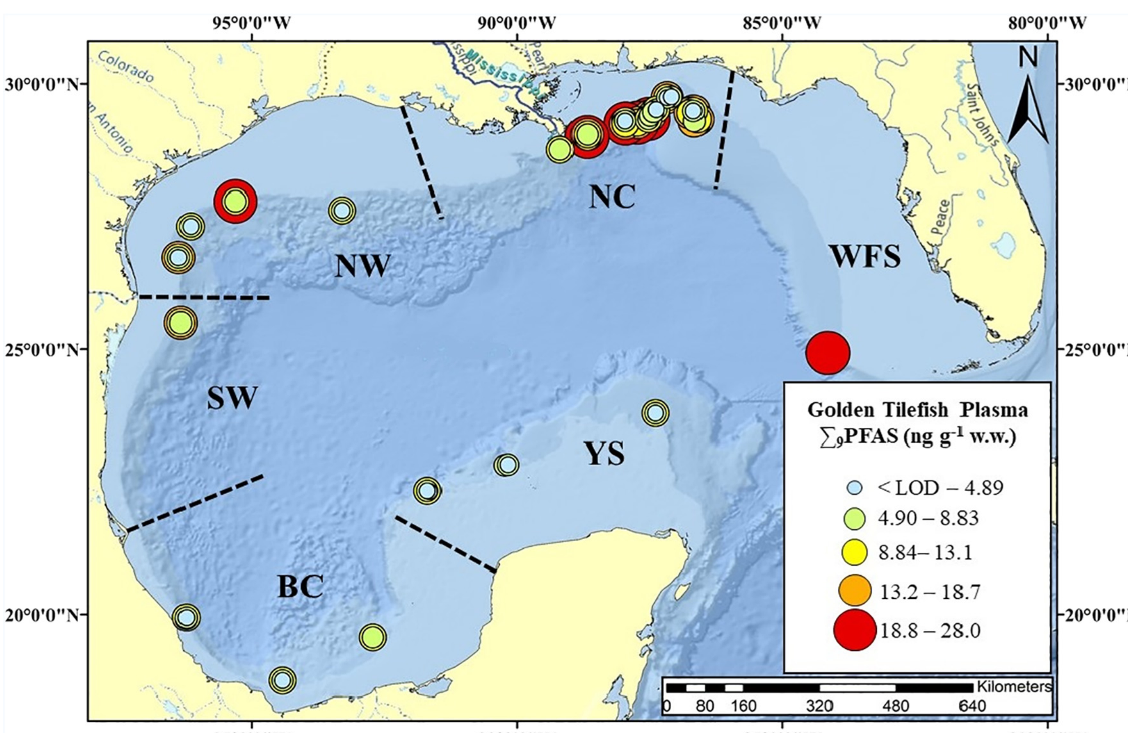

Fig. 2. (Use color in print). Mean concentrations of total PFAS (Σ_9PFAS ng g^{-1} w.w.) in Golden Tilefish plasma collected in the Gulf of Mexico, 2015–2017.

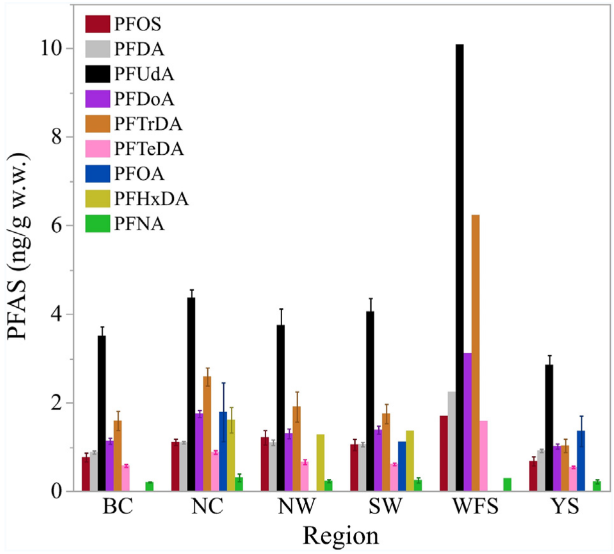

Fig. 3. Mean concentrations of individual PFAS analytes (ng g^{-1} w.w.) by region in Golden Tilefish plasma collected in the Gulf of Mexico, 2015–2017. WFS: West Florida Shelf, NC: North Central, SW: Southwest, NW: Northwest, BC: Bay of Campeche, YS: Yucatan Shelf. Each error bar is constructed using 1 standard error from the mean.

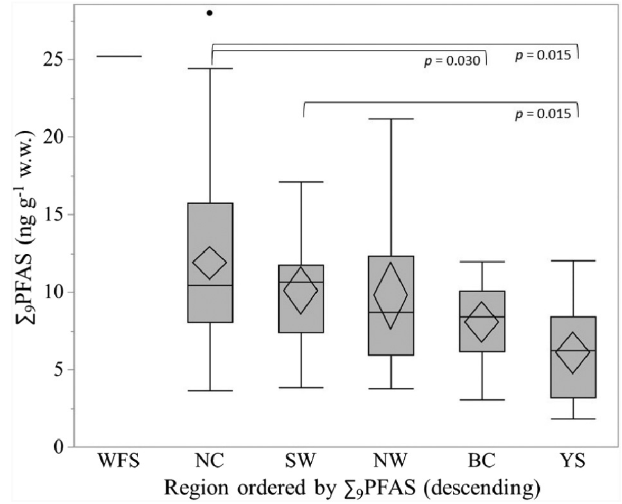

Fig. 5. The blood plasma PFAS concentrations (ng g^{-1} w.w.) by region in Golden Tilefish (n = 185) collected in the Gulf of Mexico, 2015–2017. WFS: West Florida Shelf, NC: North Central, SW: Southwest, NW: Northwest, BC: Bay of Campeche, YS: Yucatan Shelf. The horizontal line for WFS reflects a single observation from this region. The solid line and diamonds in the box plots are the mean and the confidence intervals (excluding outliers), respectively.

Repeat sampling was conducted in the NC region in 2015 and 2017, but there were no significant differences (p = 0.122) in the mean Σ_9PFAS (ng g^{-1} w.w.) in plasma samples between collection years. However, mean concentrations of PFUnDA (p = 0.044) and PFDoDA (p = 0.021) were significantly higher in 2015 compared to those collected in 2017.

3.4. Relationships between concentrations of PFAS and MHCs

Previously described MHCs identified in this subset of Tilefish (n = 51) included lipid-type and glycogen-type hepatocellular vacuolar change, bile duct fibrosis, edema, pigmented macrophage aggregates, granulomas, parasites, Foci of Cellular Alterations (FCAs) and atrophy (Table 2, for complete sample set see Snyder, 2020). In this subset of specimens, the majority (>80%) of the Tilefish evaluated had microvesiculation consistent with hepatocellular intracytoplasmic glycogen-type accumulation. Whereas, hepatocellular macrovesiculation was primarily associated with intracytoplasmic lipid-type changes and interstitial edema. The highest prevalence of glycogen-type microvesiculation and lipid-type macrovesiculation were in the NC and NW regions, respectively. Bile duct fibrosis and edema were also frequently detected (>70%) in all the Tilefish. Pigmented macrophage aggregates (PMA) and PMAs with hemosiderin, lipofuscin or ceroid were highest in the BC. Foci of cellular alteration (FCA) were only detected in the NC region.

Regression models detected both negative and positive relationships between MHCs and Σ_9PFAS in Golden Tilefish for all regions combined as well as in the NW and BC regions. (Tables S9–S12). In Tilefish collected Gulf-wide (Table S9), weak negative correlations were detected between hepatocellular glycogen-type macrovesiculation and Σ_9PFAS, PFOS, PFUnDA, PFDoDA and PFTrDA. Very weak to weak positive correlations were detected between interstitial edema and Σ_9PFAS, PFDoDA, PFTrDA and PFTeDA. Regression models did not detect any other relationships between PFAS concentrations and MHCs in Tilefish collected Gulf-wide (Table S9).

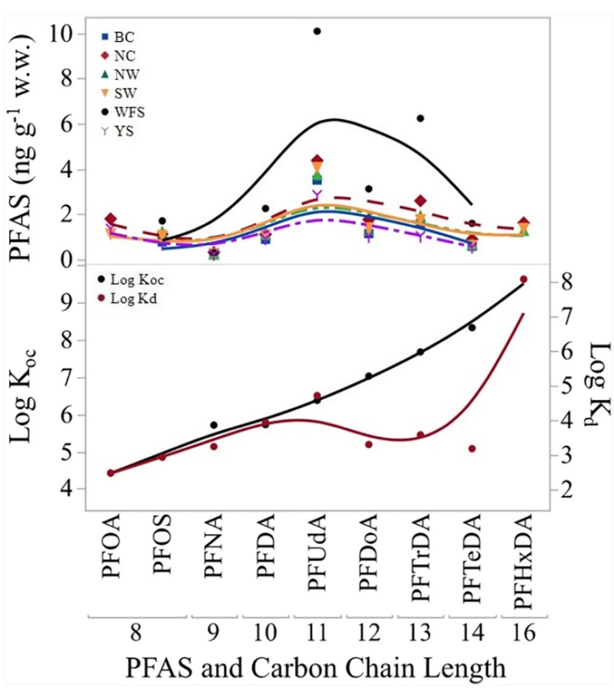

Fig. 4. (Use color in print). Concentrations of PFAS analytes (ng g^{-1} w.w.) in Golden Tilefish plasma by region, carbon chain length and partitioning properties (Log K_{oc}: organic carbon-normalized distribution coefficient; Log K_d: sediment/water distribution coefficient). Values for partitioning coefficients are from (Gałęzowska et al., 2021).

Table 2
Prevalence of microscopic hepatic changes (MHC) identified in liver tissues of a subset of Golden Tilefish (n = 51) collected in the north central (NC), northwest (NW) and Bay of Campeche (BC) regions of the Gulf of Mexico. Sample sizes (n) by region are indicated and regions with the highest prevalence of each MHC are in bold.

Microscopic hepatic change	NC (n = 29)	NW (n = 5)	BC (n = 17)
Lipid-type change	24%	**40%**	18%
Glycogen-type change	86%	80%	**88%**
Bile duct fibrosis	76%	**100%**	71%
Edema	83%	**100%**	76%
Pigmented macrophage aggregates (PMA)	69%	60%	**88%**
PMA with hemosiderin/lipofuscin/ceroid	62%	40%	**82%**
Granuloma(s)	**38%**	20%	18%
Parasite(s)	24%	**40%**	18%
Foci of cellular alteration (FCA)	**17%**	0%	0%
Atrophy	7%	**20%**	6%

In the Bay of Campeche, weak negative correlations were detected between hepatocellular glycogen-type macrovesiculation and Σ_9PFAS, PFUnDA, PFDoDA and PFTrDA (Table S10). Whereas, significant strong correlations were found between lipid-type microvesiculation and PFTeDA. Significant strong correlations were also detected between granulomas and PFTeDA and PFTeDA. Parasites appeared to have a strong positive association with PFNA and weak negative association with PFTeDA. Additionally, strong positive relationships were detected between PFDA and PMAs with hemosiderin, lipofuscin or ceroid. In Tilefish collected in the NW region, PFDA was found to have a very strong negative association with lipid-type vacuolar change and macrovesiculation (Table S11). In contrast, there were no significant associations found between MHCs and PFAS in Tilefish collected in the NC region (Table S12).

4. Discussion

This is the first report of significant PFAS concentrations in a benthic burrowing species located in deep waters (140–400 m) along the continental shelf of the GoM. Of the 110 targeted PFAS, nine long-chain PFAS, mostly PFCAs, were 315detected in Tilefish collected in 2015–2017. Tilefish plasma concentrations of individual PFAS ranged from <LOD to 9.30 ng g^{-1} (w.w.) while total PFAS (Σ_9PFAS) concentrations ranged from <LOD to 27.9 ng g^{-1} (w.w.). These levels are 2.5 to 16 times lower than $\sum_4$PFAS levels measured in the plasma of Smallmouth Bass *(Micropterus dolomieu)* collected in 2018 from the Chesapeake watershed (Blazer et al., 2021). Additionally, the mean total PFCA concentrations in Northern Pike *(Esox lucius,* $\sum_{11}$PFCAs 104 ± 50 ng g^{-1} w.w.) collected in 2011 from the St. Lawrence River Estuary (De Silva et al., 2016) are up to an order of magnitude higher than mean concentrations in Tilefish ($\sum_8$PFCAs 9.46 ± 4.76 ng g^{-1} w.w.) from this study. Lower concentrations in the demersal, offshore Tilefish were not unexpected since the gradient of pollution concentrations generally decline with increasing distance from the coast and anthropogenic sources.

In both deep and coastal environments, PFOS (a PFSA) was the predominant PFAS measured in livers and plasma of shallow water fishes (Blazer et al., 2021; De Silva et al., 2016; Nakata et al., 2006), livers of deep-diving whales (Hart et al., 2008a), oceanic tuna (Hart et al., 2008b), and in Arctic environments (Butt et al., 2010; Carlsson et al., 2016; Muir et al., 2019). PFOS is generally the predominant PFAS found in all species, tissues and locations analyzed globally (Houde et al., 2011). In contrast, PFUnDA (a PFCA) was the predominant PFAS followed by PFTrDA in the offshore demersal Tilefish plasma in this study.

The unique compositional profile of PFAS in Tilefish compared to published research may be explained by their benthic lifestyle, diet and the partitioning characteristics which define the bioavailability of these compounds. Tilefish are sedentary and non-migratory as juveniles and adults, displaying high site fidelity to their self-excavated burrows that are used as long-term habitat and protection from predators. These burrows are formed and maintained by frequent oral excavation of sediments. As such, Tilefish are intimately coupled with fine-grained sediments, increasing their likelihood of exposures to anthropogenic pollutants as observed in Snyder et al. (2015).

Experimental studies have revealed the membrane/water (log $K_{mem/w}$) and protein (albumin)/water (log $K_{p/w}$) coefficients both increase incrementally with chain length, indicating increasing cellular permeability and affinity for proteins (albumin), respectively (Allendorf et al., 2019; Ebert et al., 2020). However, the log $K_{mem/w}$ and log $K_{p/w}$ increase with increasing chain length until PFDA [C10: $(M\text{-}H)^- = C9$] and then decreased for PFUnDA [C11: $(M\text{-}H)^- = C10$] suggesting reduced diffusibility and uncertainty for longer chain (>C10) compounds (Allendorf et al., 2019; Ebert et al., 2020). In this study, the concentrations of PFAS in plasma extracts increased for those compounds with carbon chain lengths C8 [$(M\text{-}H)^- = C7$] through C11 [$(M\text{-}H)^- = C10$] with corresponding log K_{ow} values between 4.5 and 6.82, followed by a linear decrease in concentrations for compounds with higher carbon chain lengths (≥C12; log $K_{ow} > 7$). Concentrations increased with increasing chain length starting at C8 and then declined after C11 (Fig. 3).

Higher concentrations of long-chain PFAS tend to be found in sediments due to the partitioning properties between water and sediment, while the water column tends to be predominated by short-chain PFAS (Zhao et al., 2016). Experimental studies have determined the predominance of long-chain PFCAs tend to associate with carbonaceous materials by hydrophobic interaction (Zhao et al., 2012). The longer carbon chain increases the hydrophobicity of PFAS and enhances the interaction with carbonaceous materials (Higgins and Luthy, 2006; Zhao et al., 2012). Desorption of PFAS from sediment particles is another major process dictating the fate of these compounds. The desorption of PFAS decreases with increasing chain length therefore while short-chain PFAS (C ≤ 7) are more frequently detected in the water column and pore-water, the long-chain PFAS (C ≥ 11) prefer to be bound to sediments (Ahrens and Bundschuh, 2014; Ahrens et al., 2011; Zhao et al., 2012). The organic carbon-normalized distribution coefficient (Log K_{oc}) and the sediment/water distribution coefficient (Log K_d) are commonly used parameters to predict the partitioning behavior in aquatic environments. Although the Log K_{oc} continues to increase with increasing chain length, the PFAS concentrations in Tilefish increase with increasing Log K_{oc} and K_d and then exponentially decline after PFUnDA (Log K_{oc}: 6.4; Log K_d:4.7) suggesting the bioavailability of PFAS in sediments decreases with carbon chain length (Fig. 3), supporting hypotheses from experimental research by Zhao et al. (2012).

The unique profile observed in Tilefish may also be explained by pharmacokinetics. Research has demonstrated the preferential distribution between blood and tissues can be both compound and species-specific (Pizzurro et al., 2019). Yet overall, PFBS, PFOA, and PFOS preferentially distribute to the liver, while PFBA and PFHxS appear to preferentially distribute to serum (Pizzurro et al., 2019; Savoca and Pace, 2021). The low levels of PFOS and PFOA, as well as the non-detectable levels of other short-chain PFAS and precursor compounds in Tilefish plasma may be the result of these compounds distributing in other matrices. Furthermore, PFOS precursors appear to exist as temporary decomposition intermediates of their own precursors before being transformed into the final end product, PFOS (Buck et al., 2011; Lee et al., 2020). However, further evaluation of additional tissues in Tilefish would be needed to determine the species-specific preferential distribution of these compounds. Therefore, we conclude the unique compositional profile of PFASs in Tilefish is likely in part due to the ingestion of high amounts of bioavailable, long-chain PFAS compounds bound to sediments due to the fish's burrowing lifestyle, the ingestion of both contaminated sediments and diet, as well as species-specific pharmacokinetics.

Regional differences in GoM PFAS concentrations were also noted in Tilefish plasma. Tilefish collected near major riverine inputs (e.g., Rio Grande, Mississippi, Coatzacoalcos, Grijalva-Usumacinta, Papaloapan) tended to have the highest levels of PFASs. Substantial riverine emissions and discharge of long-chain (C7–12) PFCAs have been reported for other regions globally (Muir and Miaz, 2021). The high levels in one Tilefish at the southern end of the WFS is suspected to be primarily the result of its location near the Loop Current which flushes the entire GoM through the Florida Straits. Overall, the NC region near the mouth of the Mississippi River and DeSoto Canyon has significantly higher PFAS levels than those collected off Texas and Mexico. A similar spatial pattern was observed for PAH exposure and hepatic accumulation in Tilefish (Snyder et al., 2020). The Mississippi River drains ~40% of the continental U.S. into the GoM transporting elevated levels of nutrients, agricultural and industrial chemicals, including PFAS which then tend to sink into the DeSoto Canyon (Newsted et al., 2017).

Additionally, two compounds mainly originating from fluorotelemer alcohols or other fluorotelemer products, PFUnDA and PFDoDA (Zhang et al., 2016), were found to be significantly higher in the plasma of

Tilefish collected in the north central region of the GoM during 2015 compared to 2017. Although there were no significant differences in Tilefish length between these two years ($p = 0.0524$), Tilefish collected in the north central GoM during 2015 were significantly heavier ($p = 0.039$). Therefore, the lower concentrations in plasma of Tilefish collected in 2017 may be the result of biodilution.

Exposure to PFAS can have serious health implications for both humans and animals, including hepatic, immunological, developmental, endocrine, reproductive, and cardiovascular effects, as well as liver, kidney and bladder cancers (Beecher et al., 2018; Bell et al., 2021; Fenton et al., 2021; Gagliano et al., 2020; Mueller and Yingling, 2017; Pelch et al., 2019; Sunderland et al., 2019). Experimental studies have demonstrated that PFOS in particular may interfere with the homeostasis of DNA metabolism and disrupt hepatocyte membrane integrity (Hoff et al., 2003). The PFAS with the highest concentration in Tilefish, was PFUnDA, a potent developmental toxicant and teratogen in amphibian and avian species (Kim et al., 2013; O'Brien et al., 2013). Concentrations of PFAS in Tilefish collected both Gulf-wide had weak relationships with biometrics suggesting PFAS may have a negligible influence on morphometrics and health condition. However, when evaluating individual regions, there were strong relationships detected between Σ_9PFAS and MHCs in the BC and NW regions, while there were no relationships detected in the NC region. This is somewhat surprising considering the NC region had the highest levels of Σ_9PFAS. Snyder et al. (2020) detected strong significant associations between several MHCs and total polycyclic aromatic hydrocarbons (PAHs) measured in the hepatic tissues of Tilefish collected in the NC region, a region that suffers from acute and chronic oil pollution. These data might suggest that PAHs have more of an impact on hepatic changes in Tilefish from the NC region compared to PFAS. However hepatic concentrations of PFAS need to be determined to identify which of these compound groups had a stronger effect on MHCs in Tilefish.

The liver is the primary target organ for exposure effects and mounting evidence suggests acute toxicity of PFAS increase with chain length (Jensen and Leffers, 2008; Tartu et al., 2014). Blood (and plasma) interact with all organs, including the liver, therefore we explored the relationships between Tilefish plasma levels of PFAS and histological hepatic changes. Significant, albeit weak relationships, were found between PFAS and hepatocellular macrovesiculation and interstitial edema. Vacuolar change is the intracytoplasmic cellular accumulation of glycogen or lipids in the hepatocytes (Cullen, 2009; Wolf and Wheeler, 2018). In fish, vacuolar change can be an indicator of nutritional status, age related processes, reproductive status, cellular mechanism alteration, hepatocyte damage, and exposure to toxicants. The size and type of hepatocellular vacuole formation can be influenced by the mechanism behind the accumulation and may suggest a specific pathogenesis. In this study, macrovesiculation was primarily associated with glycogen-type vacuoles. The inverse association between PFAS and glycogen-type macrovesiculation suggests prevalence of glycogen storage increases with decreasing PFAS concentration. Microvesiculation was primarily associated with lipid-type changes in this study. Furthermore, a higher detection frequency was seen in the subset of Tilefish collected in the NC region of the GoM. Although there were no correlations detected with PFAS, microvesiculation have been documented in cases with mitochondria injury however, in small animals, severe liver dysfunction is typically seen in cases of lipid-type macrovesiculation (Cullen, 2009). More importantly, this suggests that increased PFAS concentrations may be altering liver metabolism and energy storage which can have severe implications for the health of Tilefish populations. Similar effects were found for other organic compounds (i.e., PAHs) in Red Snapper (*Lutjanus campechanus*) and Tilefish (Pulster et al., 2021; Snyder, 2020). Although the depletion or accumulation of hepatic glycogen and lipid can be the result of common toxicological responses following pollutant exposures, these changes can also be the result of normal physiological processes or secondary responses to dietary excess or starvation (Wolf and Wolfe, 2005).

Pigmented macrophage aggregates (PMAs) and hemosiderin, lipofuscin or ceroid laden PMAs were frequently detected in all Tilefish, but the highest prevalence was detected in the BC region. In fish, PMAs are common indicators of chronic stress, underlying disease or parasitic infestations, however, they are also useful indicators of water quality since the size and abundance of PMAs have been associated with stressful environmental conditions, such as deoxygenation and chemical pollution (Agius and Roberts, 2003). Increased hemosiderin laden PMAs may be the result of increased metabolic destruction of damaged erythrocytes or as a protective mechanism. Increased PMAs containing hemosiderin have been observed following hemolytic anemia and prolonged starvation. Lipofuscin or ceroid pigments may accumulate in the PMAs as a result of dietary deficiencies, autophagy, cellular breakdown, and have been observed in fish numerous pathological conditions, including toxins, nutritional deficiencies, bacterial and viral diseases.

Several of the MHCs described also had significant but weak relationships with polycyclic aromatic hydrocarbons (PAHs), a component of crude oil, in Tilefish collected Gulf-wide (Snyder, 2020). The GoM is well known for its oil and gas productivity and elevated levels of PAHs in several species of fish, including Tilefish (Pulster et al., 2020b; Snyder, 2020). The mixture of PAHs and PFAS have been shown to disrupt the endocrine system and the combination of the two have demonstratable effects on the reproduction and behavior in juvenile Atlantic Cod (*Gadus morhua*) by altering the dopaminergic pathway (Khan et al., 2019). Future research should include measuring PFAS in liver tissues to more definitely evaluate relationships and roles in the induction of hepatic changes. In addition, the combined and interactive toxic effects of multiple contaminants and stressors needs to be further explored in fishes from the GoM as the health of commercially and ecologically important fish populations has been found to be threatened (Pulster et al., 2021; Pulster et al., 2020a; Pulster et al., 2020b; Snyder, 2020; Struch et al., 2019).

5. Conclusion

Oceans are a major sink for anthropogenic pollutants, including PFAS which have been widely detected in global surface waters for the past two decades (Muir and Miaz, 2021). To our knowledge, this is the first report of PFAS in a deep (≤400 m) demersal species along the continental shelf of the GoM, and in fact anywhere globally. A unique compositional profile of PFAS, consisting of nine long-chain compounds, dominated by PFCAs were detected at relatively high frequencies in Tilefish collected throughout the GoM, with highest plasma concentrations in the NC region. These findings give insight into the fate of PFAS at depth in our aquatic ecosystems. In addition, this research supports previous findings that indicate regions adjacent to major riverine systems (i.e., Mississippi River) are at risk for higher inputs of anthropogenic pollution, including PFAS. The relationships between PFAS and MHCs, albeit weak, suggest PFAS may play a minor role in liver changes that could be detrimental to their overall health. The PFAS with the highest concentration in Tilefish, was PFUnDA, a potent developmental toxicant and teratogen in amphibian and avian species (Kim et al., 2013; O'Brien et al., 2013). Sublethal effects can initiate a variety of physiological responses that can impair an individual fish (e.g., behavioral effects) with possible lethal consequences, and also have an impact at the population level (e.g., reproductive and body condition effects). However, additional research is needed to investigate the definitive role PFAS play in inducing hepatic changes in fish as well as the combined toxic effect of PFAS with other chemicals and environmental stressors. Tilefish accounted for nearly 400,000 pounds and $1.2 million in commercial fishery landings in the GoM during 2019 (https://fisheries.noaa.gov). Considering dietary ingestion and drinking water are the two major PFAS exposure pathways in humans, future studies should include measuring the PFAS levels in edible muscle tissue for assessing potential human health risks from consumption. The presence of PFAS, in addition to other pollutants (e.g., PAHs) in Tilefish has severe implications for population health as well as the environmental and economic health of the GoM.

CRediT authorship contribution statement

The manuscript was written through contributions from all authors. **Erin** L. **Pulster:** Data analysis, original manuscript preparation, sample acquisition; **Amanda E. Wichterman:** Analytical chemistry, original manuscript preparation; **Susan M. Snyder:** Sample acquisition; **Susan Fogelson:** Conducted the histopathology; **Bianca F. Da Silva:** Analytical chemistry, method development and validation; **Kaylie A. Costa:** Analytical chemistry; **Joe Aufmuth**: Data analysis, map development; **Kristina Deak:** Sample acquisition; **Steven A. Murawski:** Investigation, sample acquisition; **John A. Bowden:** Conceptualization, method validation, resources, project supervision, project administration, funding acquisition. All authors contributed to the final writing, editing, review and approval of the manuscript.

Declaration of competing interest

The authors have no conflicts of interest to declare.

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Appendix A. Supplementary data

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References

Able, K.W., Grimes, C.B., Cooper, R.A., Uzmann, J.R., 1982. Burrow construction and behavior of tilefish, Lopholatilus chamaeleonticeps, in Hudson submarine canyon. Environ. Biol. Fish 7, 199–205.

Agius, C., Roberts, R.J., 2003. Melano-macrophage centres and their role in fish pathology. J. Fish Dis. 26, 499–509.

Ahrens, L., Bundschuh, M., 2014. Fate and effects of poly- and perfluoroalkyl substances in the aquatic environment: a review. Environ. Toxicol. Chem. 33, 1921–1929.

Ahrens, L., Yeung, L.W.Y., Taniyasu, S., Lam, P.K.S., Yamashita, N., 2011. Partitioning of perfluorooctanoate (PFOA), perfluorooctane sulfonate (PFOS) and perfluorooctane sulfonamide (PFOSA) between water and sediment. Chemosphere 85, 731.

Allendorf, F., Berger, U., Goss, K.-U., Ulrich, N., 2019. Partition coefficients of four perfluoroalkyl acid alternatives between bovine serum albumin (BSA) and water in comparison to ten classical perfluoroalkyl acids. Environ. Sci. Process. Impacts 21, 1852–1863.

Anderson, M.J., Walsh, D.C.I., Robert Clarke, K., Gorley, R.N., Guerra-Castro, E., 2017. Some solutions to the multivariate behrens-fisher problem for dissimilarity-based analyses. 59, 57–79.

Bangma, J.T., Reiner, J.L., Botha, H., Cantu, T.M., Gouws, M.A., Guillette, M.P., et al., 2017. Tissue distribution of perfluoroalkyl acids and health status in wild Mozambique tilapia (Oreochromis mossambicus) from loskop dam, Mpumalanga, South Africa. J. Environ. Sci. 61, 59–67.

Beecher, N., Rainey, M., East, N., 2018. What are PFAS?

Bell, E.M., De Guise, S., McCutcheon, J., Lei, Y., Levin, M., Li, B., et al., 2021. Exposure, health effects, sensing, and remediation of the emerging PFAS contaminants–Scientific challenges and potential research directions. Sci. Total Environ. 146399.

Blazer, V.S., Gordon, S.E., Walsh, H.L., Smith, C.R., 2021. Perfluoroalkyl substances in plasma of smallmouth bass from the Chesapeake Bay watershed. Int. J. Environ. Res. Public Health 18, 5881.

Buck, R.C., Franklin, J., Berger, U., Conder, J.M., Cousins, I.T., De Voogt, P., et al., 2011. Perfluoroalkyl and polyfluoroalkyl substances in the environment: terminology, classification, and origins. Integr. Environ. Assess. Manag. 7, 513–541.

Butt, C.M., Berger, U., Bossi, R., Tomy, G.T., 2010. Levels and trends of poly- and perfluorinated compounds in the arctic environment. Sci. Total Environ. 408, 2936–2965.

Calafat, A.M., Wong, L.-Y., Kuklenyik, Z., Reidy, J.A., Needham, L.L., 2007. Polyfluoroalkyl chemicals in the US population: data from the National Health and nutrition examination survey (NHANES) 2003–2004 and comparisons with NHANES 1999–2000. Environ. Health Perspect. 115, 1596–1602.

Carlsson, P., Crosse, J.D., Halsall, C., Evenset, A., Heimstad, E.S., Harju, M., 2016. Perfluoroalkylated substances (PFASs) and legacy persistent organic pollutants (POPs) in halibut and shrimp from coastal areas in the far north of Norway: small survey of important dietary foodstuffs for coastal communities. Mar. Pollut. Bull. 105, 81–87.

Carr, B.E., 2020. Polychlorinated biphenyls, organochlorine pesticides, and polycyclic aromatic hydrocarbons in snapper (Family Lutjanidae) from Cuba and the wider Gulf of Mexico. College of Marine Science. MSc. University of South Florida, Tampa, FL (USA).

Conder, J.M., Hoke, R.A., Wolf, Wd., Russell, M.H., Buck, R.C., 2008. Are PFCAs Bioaccumulative? A critical review and comparison with regulatory criteria and persistent lipophilic compounds. Environ. Sci. Technol. 42, 995–1003.

Cullen, J.M., 2009. Summary of the world small animal veterinary association standardization committee guide to classification of liver disease in dogs and cats. Veterinary clinics of North America-small animal. Practice 39, 395-+.

Da Silva, B.F., Ahmadireskety, A., Aristizabal-Henao, J.J., Bowden, J.A., 2020. A rapid and simple method to quantify per-and polyfluoroalkyl substances (PFAS) in plasma and serum using 96-well plates. MethodsX 7, 101111.

De Silva, A.O., Spencer, C., Ho, K.C.D., Al Tarhuni, M., Go, C., Houde, M., et al., 2016. Perfluoroalkylphosphinic acids in northern pike (Esox lucius), double-crested cormorants (Phalacrocorax auritus), and bottlenose dolphins (Tursiops truncatus) in relation to other perfluoroalkyl acids. Environ. Sci. Technol. 50, 10903.

Deak, K., 2020. A baseline health evaluation of Golden tilefish (Lopholatilus chamaeleonticeps) and red snapper (Lutjanus campechanus), post-Deepwater horizon college of marine science. Ph.D. University of South Florida, Tampa, FL.

Ebert, A., Allendorf, F., Berger, U., Goss, K.-U., Ulrich, N., 2020. Membrane/water partitioning and permeabilities of perfluoroalkyl acids and four of their alternatives and the effects on toxicokinetic behavior. Environ. Sci. Technol. 54, 5051–5061.

Fenton, S.E., Ducatman, A., Boobis, A., DeWitt, J.C., Lau, C., Ng, C., et al., 2021. Per-and polyfluoroalkyl substance toxicity and human health review: current state of knowledge and strategies for informing future research. Environ. Toxicol. Chem. 40, 606–630.

Flaherty, J.M., Connolly, P.D., Decker, E.R., Kennedy, S.M., Ellefson, M.E., Reagen, W.K., et al., 2005. Quantitative determination of perfluorooctanoic acid in serum and plasma by liquid chromatography tandem mass spectrometry. J. Chromatogr. B 819, 329–338.

Forsthuber, M., Kaiser, A.M., Granitzer, S., Hassl, I., Hengstschläger, M., Stangl, H., et al., 2020. Albumin is the major carrier protein for PFOS, PFOA, PFHxS, PFNA and PFDA in human plasma. Environ. Int. 137, 105324.

Freeman, B.L., Turner, S.C., 1977. Biological and fisheries data on tilefish, Lopholatilus chamaeleonticeps. Goode and Bean: Sandy Hook Laboratory, Northeast Fisheries Center, National Marine Fisheries.

Froese, R., 2006. Cube law, condition factor and weight-length relationships: history, meta-analysis and recommendations. J. Appl. Ichthyol. 22, 241–253.

Gagliano, E., Sgroi, M., Falciglia, P.P., Vagliasindi, F.G., Roccaro, P., 2020. Removal of poly- and perfluoroalkyl substances (PFAS) from water by adsorption: role of PFAS chain length, effect of organic matter and challenges in adsorbent regeneration. Water Res. 171, 115381.

Gałęzowska, G., Rogowska, J., Olkowska, E., Ratajczyk, W., Wolska, L., 2021. Environmental risk assessment resulting from sediment contamination with perfluoroalkyl substances. Molecules 26, 116.

Grimes, C.B., Turner, S.C., 1999. The complex life history of tilefish Lopholatilus chamaeleonticeps and vulnerability to exploitation. In: Musick, J.A. (Ed.), Life in the Slow Lane: Ecology and Conservation of Long-Lived Marine Animals. 23, pp. 17–26.

Grimes, C.B., Able, K.W., Jones, R.S., 1986. Tilefish, Lopholatilus chamaeleonticeps, habitat, behavior and community structure in mid-Atlantic and southern New England waters. Environ. Biol. Fish 15, 273–292.

Hart, K., Kannan, K., Isobe, T., Takahashi, S., Yamada, T.K., Miyazaki, N., et al., 2008a. Time trends and transplacental transfer of perfluorinated compounds in melon-headed whales stranded along the japanese coast in 1982, 2001/2002, and 2006. Environ. Sci. Technol. 42, 7132–7137.

Hart, K., Kannan, K., Tao, L., Takahashi, S., Tanabe, S., 2008b. Skipjack tuna as a bioindicator of contamination by perfluorinated compounds in the oceans. Sci. Total Environ. 403, 215–221.

Higgins, C.P., Luthy, R.G., 2006. Sorption of perfluorinated surfactants on sediments. Environ. Sci. Technol. 40, 7251.

Hoff, P., Van Dongen, W., Esmans, E., Blust, R., De Coen, W., 2003. Evaluation of the toxicological effects of perfluorooctane sulfonic acid in the common carp (Cyprinus carpio). Aquat. Toxicol. 62, 349–359.

Houde, M., Wells, R.S., Fair, P.A., Bossart, G.D., Hohn, A.A., Rowles, T.K., et al., 2005a. Polyfluoroalkyl compounds in free-ranging bottlenose dolphins (Tursiops truncatus) from the Gulf of Mexico and the Atlantic Ocean. Environ. Scie. Technol. 39, 6591–6598.

Houde, M., Wells, R.S., Fair, P.A., Bossart, G.D., Hohn, A.A., Rowles, T.K., et al., 2005b. Polyfluoroalkyl compounds in free-ranging bottlenose dolphins (Tursiops truncatus) from the Gulf of Mexico and the Atlantic Ocean. Environ. Sci. Technol. 39, 6591–6598.

Houde, M., Bujas, T.A., Small, J., Wells, R.S., Fair, P.A., Bossart, G.D., et al., 2006a. Biomagnification of perfluoroalkyl compounds in the bottlenose dolphin (Tursiops truncatus) food web. Environ. Sci. Technol. 40, 4138–4144.

Houde, M., Bujas, T.A.D., Small, J., Wells, R.S., Fair, P.A., Bossart, G.D., et al., 2006b. Biomagnification of perfluoroalkyl compounds in the bottlenose dolphin (Tursiops truncatus) food web. Environ. Sci. Technol. 40, 4138–4144.

Houde, M., De Silva, A.O., Muir, D.C.G., Letcher, R.J., 2011. Monitoring of perfluorinated compounds in aquatic biota: an updated review. Environ. Sci. Technol. 45, 7962–7973.

Jensen, A.A., Leffers, H., 2008. Emerging endocrine disrupters: perfluoroalkylated substances. Int. J. Androl. 31, 161–169.

Jones, D.L., 2017. Fathom Toolbox for MATLAB: Software for multivariate ecological and oceanographic data analysis. University of South Florida, College of Marine Science, St. Petersburg, FL.

Jones, P.D., Hu, W., De Coen, W., Newsted, J.L., Giesy, J.P., 2003. Binding of perfluorinated fatty acids to serum proteins. 22, 2639–2649.

Keller, J.M., Ngai, L., McNeill, J.B., Wood, L.D., Stewart, K.R., O'Connell, S.G., et al., 2012. Perfluoroalkyl contaminants in plasma of five sea turtle species: comparisons in concentration and potential health risks. Environ. Toxicol. Chem. 31, 1223–1230.

Kennicutt, M.C., 2017. Water quality of the Gulf of Mexico. In: Ward, C.H. (Ed.), Habitats and Biota of the Gulf of Mexico: Before the Deepwater Horizon Oil Spill: Volume 1: Water Quality, Sediments, Sediment Contaminants, Oil and Gas Seeps, Coastal Habitats, Offshore Plankton and Benthos, and Shellfish. Springer, New York, New York, NY, pp. 55–164.

Kennicutt, M.C., Brooks, J.M., Atlas, E.L., Giam, C.S., 1988. Organic compounds of environmental concern in the Gulf of Mexico: a review. Aquat. Toxicol. 11, 191–212.

Khan, E.A., Bertotto, L.B., Dale, K., Lille-Langøy, R., Yadetie, F., Karlsen, O.A., et al., 2019. Modulation of neuro-dopamine homeostasis in juvenile female Atlantic cod (Gadus morhua) exposed to polycyclic aromatic hydrocarbons and perfluoroalkyl substances. Environ. Sci. Technol. 53, 7036–7044.

Kim, M., Son, J., Park, M.S., Ji, Y., Chae, S., Jun, C., et al., 2013. In vivo evaluation and comparison of developmental toxicity and teratogenicity of perfluoroalkyl compounds using xenopus embryos. Chemosphere 93, 1153–1160.

Kissa, E., 2001. CRC Press.

Lau, C., Anitole, K., Hodes, C., Lai, D., Pfahles-Hutchens, A., Seed, J., 2007. Perfluoroalkyl acids: a review of monitoring and toxicological findings. Toxicol. Sci. 99, 366–394.

Lee, Y.-M., Lee, J.-Y., Kim, M.-K., Yang, H., Lee, J.-E., Son, Y., et al., 2020. Concentration and distribution of per- and polyfluoroalkyl substances (PFAS) in the Asan Lake area of South Korea. J. Hazard. Mater. 381, 120909.

Lynch, K.M., Fair, P.A., Houde, M., Muir, D.C., Kannan, K., Bossart, G.D., et al., 2019. Temporal trends in per-and polyfluoroalkyl substances in bottlenose dolphins (Tursiops truncatus) of Indian River lagoon, Florida and Charleston,South Carolina. 53, 14194–14203.

Martin, J.W., Mabury, S.A., Solomon, K.R., Muir, D.C.G., 2003. Bioconcentration and tissue distribution of perfluorinated acids in rainbow trout (Oncorhynchus mykiss). Environ. Toxicol. Chem. 22, 196–204.

Matyash, V., Liebisch, G., Kurzchalia, T.V., Shevchenko, A., Schwudke, D., 2008. Lipid extraction by methyl-tert-butyl ether for high-throughput lipidomics. J. Lipid Res. 49, 1137–1146.

Mueller, R., Yingling, V., 2017. Interstate Technology & Regulatory Council, USA.

Muir, D., Miaz, L.T., 2021. Spatial and temporal trends of perfluoroalkyl substances in Global Ocean and coastal waters. Environ. Sci. Technol. 55 (14), 9527–9537.

Muir, D., Bossi, R., Carlsson, P., Evans, M., De Silva, A., Halsall, C., et al., 2019. Levels and trends of poly- and perfluoroalkyl substances in the Arctic environment – an update. Emerging Contam. 5, 240–271.

Murawski, S.A., Peebles, E.B., Gracia, A., Tunnell Jr., J.W., Armenteros, M., 2018a. Comparative abundance, species composition, and demographics of continental shelf fish assemblages throughout the Gulf of Mexico. Mar. Coast. Fish. 10, 325–346.

Murawski, S.A., Peebles, E.B., Gracia, A., Tunnell, J.W., Armenteros, M., 2018b. Comparative abundance, species composition, and demographics of continental shelf fish assemblages throughout the Gulf of Mexico. Mar. Coast. Fish. 10, 325–346.

Nakata, H., Kannan, K., Nasu, T., Cho, H.-S., Sinclair, E., Takemura, A., 2006. Perfluorinated contaminants in sediments and aquatic organisms collected from shallow water and tidal flat areas of the Ariake Sea, Japan: environmental fate of perfluorooctane sulfonate in aquatic ecosystems. Environ. Sci. Technol. 40, 4916–4921.

Nelson, W.R., Carpenter, J.S., 1968. Bottom longline explorations in the Gulf of Mexico- a report on "Oregon. II's" first cruise. Commer. Fish. Rev. 30, 57–62.

Newsted, J.L., Holem, R., Hohenstein, G., Lange, C., Ellefson, M., Reagen, W., et al., 2017. Spatial and temporal trends of poly- and perfluoroalkyl substances in fish fillets and water collected from pool 2 of the upper Mississippi River. Environ. Toxicol. Chem. 36, 3138–3147.

Ng, C.A., Hungerbühler, K., 2013. Bioconcentration of perfluorinated alkyl acids: how important is specific binding? Environ. Sci. Technol. 47, 7214–7223.

O'Brien, J.M., Williams, A., Yauk, C.L., Crump, D., Kennedy, S.W., 2013. In vitro microarray analysis identifies genes in acute-phase response pathways that are down-regulated in the liver of chicken embryos exposed in ovo to PFUdA. Toxicol. in Vitro 27, 1649–1658.

Ojo, A.F., Peng, C., Ng, J.C., 2020. Combined effects and toxicological interactions of perfluoroalkyl and polyfluoroalkyl substances mixtures in human liver cells (HepG2). Environ. Pollut. 263, 114182.

Palmer, K., Bangma, J.T., Reiner, J.L., Bonde, R.K., Korte, J.E., Boggs, A.S., et al., 2019. Per-and polyfluoroalkyl substances (PFAS) in plasma of the West Indian manatee (Trichechus manatus). Mar. Pollut. Bull. 140, 610–615.

Pelch, K.E., Reade, A., Wolffe, T.A., Kwiatkowski, C.F., 2019. PFAS health effects database: protocol for a systematic evidence map. Environ. Int. 130, 104851.

Perrot, V., Landing, W.M., Grubbs, R.D., Salters, V.J.M., 2019. Mercury bioaccumulation in tilefish from the northeastern Gulf of Mexico 2 years after the Deepwater horizon oil spill: insights from hg, C, N and S stable isotopes. Sci. Total Environ. 666, 828–838.

Pizzurro, D.M., Seeley, M., Kerper, L.E., Beck, B.D., 2019. Interspecies differences in perfluoroalkyl substances (PFAS) toxicokinetics and application to health-based criteria. Regul. Toxicol. Pharmacol. 106, 239–250.

Pulster, E.L., Gracia, A., Armenteros, M., Carr, B.E., Mrowicki, J., Murawski, S.A., 2020a. Chronic PAH exposures and associated declines in fish health indices observed for ten grouper species in the Gulf of Mexico. Sci. Total Environ. 703, 135551.

Pulster, E.L., Gracia, A., Armenteros, M., Toro-Farmer, G., Snyder, S.M., Carr, B.E., et al., 2020b. A first comprehensive baseline of hydrocarbon pollution in Gulf of Mexico fishes. Sci. Rep. 10, 6437.

Pulster, E.L., Fogelson, S., Carr, B.E., Mrowicki, J., Murawski, S.A., 2021. Hepatobiliary PAHs and prevalence of pathological changes in red snapper. Aquat. Toxicol. 230.

Savoca, D., Pace, A., 2021. Bioaccumulation, biodistribution, toxicology and biomonitoring of organofluorine compounds in aquatic organisms. Int. J. Mol. Sci. 22, 6276.

Shi, Y., Song, X., Jin, Q., Li, W., He, S., Cai, Y., 2020. Tissue distribution and bioaccumulation of a novel polyfluoroalkyl benzenesulfonate in crucian carp. Environ. Int. 135, 105418.

Snyder, S.M., 2020. College of Marine Science. Doctor of Philosophy. University of South Florida, Tampa, FL, p. 191.

Snyder, S.M., Pulster, E.L., Wetzel, D.L., Murawski, S.A., 2015. PAH exposure in Gulf of Mexico demersal fishes, post-Deepwater horizon. Environ. Sci. Technol. 49, 8786–8795.

Snyder, S.M., Pulster, E.L., Murawski, S.A., 2019. Associations between chronic exposure to polycyclic aromatic hydrocarbons and health indices in Gulf of Mexico tilefish (Lopholatilus chamaeleonticeps) post-Deepwater horizon. Environ. Toxicol. Chem. 38 (12), 2659–2671.

Snyder, S.M., Olin, J.A., Pulster, E.L., Murawski, S.A., 2020. Spatial contrasts in hepatic and biliary PAHs in tilefish (Lopholatilus chamaeleonticeps) throughout the Gulf of Mexico, with comparison to the Northwest Atlantic. Environ. Pollut. 258, 113775.

Struch, R., Pulster, E.L., Schreier, A., Murawski, S.A., 2019. Evaluating persistent PAH exposures in Urophycis spp. Following the Deepwater Horizon Oil Spill. University of California, Davis.

Sunderland, E.M., Hu, X.C., Dassuncao, C., Tokranov, A.K., Wagner, C.C., Allen, J.G., 2019. A review of the pathways of human exposure to poly-and perfluoroalkyl substances (PFASs) and present understanding of health effects. 29, 131–147.

Taniyasu, S., Kannan, K., Yeung, L.W., Kwok, K.Y., Lam, P.K., Yamashita, N., 2008. Analysis of trifluoroacetic acid and other short-chain perfluorinated acids (C2–C4) in precipitation by liquid chromatography–tandem mass spectrometry: comparison to patterns of long-chain perfluorinated acids (C5–C18). Anal. Chim. Acta 619, 221–230.

Tartu, S., Gabrielsen, G.W., Blévin, P., Ellis, H., Bustnes, J.O., Herzke, D., et al., 2014. Endocrine and fitness correlates of long-chain perfluorinated carboxylates exposure in Arctic breeding black-legged kittiwakes. Environ. Sci. Technol. 48, 13504–13510.

USEPA, 2009. Long-chain perfluorinated chemicals (PFCs) action plan.

Wolf, J.C., Wheeler, J.R., 2018. A critical review of histopathological findings associated with endocrine and non-endocrine hepatic toxicity in fish models. Aquat. Toxicol. 197, 60–78.

Wolf, J.C., Wolfe, M.J., 2005. A brief overview of nonneoplastic hepatic toxicity in fish. Toxicol. Pathol. 33, 75–85.

Zhang, X., Lohmann, R., Dassuncao, C., Hu, X.C., Weber, A.K., Vecitis, C.D., et al., 2016. Source attribution of poly- and perfluoroalkyl substances (PFASs) in surface waters from Rhode Island and the New York metropolitan area. Environ. Sci. Technol. Lett. 3, 316–321.

Zhao, L.X., Zhu, L.Y., Yang, L.P., Liu, Z.T., Zhang, Y.H., 2012. Distribution and desorption of perfluorinated compounds in fractionated sediments. Chemosphere 88, 1390–1397.

Zhao, P., Xia, X., Dong, J., Xia, N., Jiang, X., Li, Y., et al., 2016. Short- and long-chain perfluoroalkyl substances in the water, suspended particulate matter, and surface sediment of a turbid river. Sci. Total Environ. 568, 57–65.