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PFAS Occurrence and Bioaccumulation in Marine Organisms and Implications for Coral Reef-Associated Food Webs and Biomonitoring

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Abstract. Per- and polyfluoroalkyl substances (PFAS) have been measured in diverse marine organisms, but differences among compounds, sample matrices, food webs, and analytical methods complicate interpretation of their environmental significance. This targeted, literature-based comparative review organized evidence from 29 distinct peer-reviewed primary studies published from 2011 to 2026 for which full-text articles were available. The studies covered marine and marine-associated coastal organisms; three directly investigated coral reef fishes or reef-building corals. Study characteristics and reported endpoints were coded separately for occurrence, bioaccumulation, trophic transfer, physiological associations, risk assessment, and biomonitoring relevance. All 29 studies reported measured biological PFAS, whereas 10 provided explicit accumulation-factor evidence and 11 investigated trophic transfer or predator–prey magnification. PFOS was prominent in numerous vertebrate and food-web investigations, but analyte profiles differed substantially: PFHxS accounted for 43% of measured PFAS in one reef-coral study, whereas trifluoroacetic acid was detected in all 54 coral samples in another. Compound-specific trophic patterns also differed among food webs; for example, reported PFOS trophic magnification factors included 1.75 in the Beibu Gulf and 0.78 in a Bohai Sea investigation. Differences in tissue, wet- versus dry-weight reporting, analyte coverage, and ecological context precluded quantitative pooling. Biological-effect evidence was more limited and should not be inferred from concentrations or dietary-risk estimates. The reviewed studies support taxon- and matrix-specific biomonitoring but do not yet establish how PFAS move through complete coral reef food webs. Standardized, co-located sampling of reef organisms, surrounding media, and biological endpoints would strengthen future interpretations.

Keywords: per- and polyfluoroalkyl substances, PFOS, marine organisms, bioaccumulation, coral reefs

1. Introduction

Per- and polyfluoroalkyl substances (PFAS) comprise a chemically diverse class of anthropogenic organofluorine compounds used in numerous industrial and commercial applications. Their

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persistence, coupled with the environmental mobility of certain compounds and continued use or formation of replacements and precursors, has produced widespread contamination of aquatic systems [1, 2]. The marine environment receives PFAS via rivers, wastewater discharges, coastal runoff, atmospheric inputs, and exchanges among suspended particles and sediments. Individual compounds nevertheless differ in environmental partitioning and biological uptake, so concentrations measured in seawater do not directly predict those in organisms [2, 3].

Marine monitoring must therefore encompass biological as well as environmental matrices. Studies in the South China Sea and East China Sea have measured PFAS in seawater, sediments, and multiple organism groups, illustrating the potential for distinct chemical profiles within connected compartments [4, 5]. A study in a tidally connected Hong Kong coastal wetland extended this approach across plankton, invertebrates, fishes, and waterbirds, although its brackish setting should not be treated as equivalent to an open-ocean or coral reef food web [6]. Across coastal regions, organisms may reflect cumulative waterborne, sediment-associated, and dietary exposure, together with compound-specific uptake and elimination. However, movement, age, seasonality, and habitat use also influence how well a sampled organism represents a particular locality or exposure interval.

Marine organisms can encounter PFAS through water, sediments, and food; the relative contribution of these routes' changes with ecology and life history [1]. Many ionic PFAS interact with proteins and membrane phospholipids, unlike conventional neutral lipophilic contaminants. Consequently, tissue concentrations need not follow lipid content, and sample types such as liver, muscle, plasma, whole body, and invertebrate soft tissue are not interchangeable [3]. In Belgian North Sea fishes, Byns et al. [7] reported markedly different PFOS levels in liver and muscle. Similarly, Fair et al. [8] reported total PFAS concentrations in whole fish approximately two to three times those in fillets in a coastal estuarine study. Measurements from white sharks and marine mammals provide additional evidence of matrix-specific chemical distributions [9-11].

Bioaccumulation concerns uptake and retention in an organism relative to its environment, whereas trophic magnification concerns concentration change along an explicitly defined food web. These are related but distinct questions. Studies from the South China Sea and adjacent waters have reported increasing accumulation for several longer-chain PFAS, together with trophic magnification for selected legacy and emerging compounds [5, 12-14]. Other analyses have not found a consistent trophic pattern or have estimated trophic magnification factors below one for specific PFAS [7, 15]. Thus, measured tissue contamination should not be treated as proof of trophic magnification, and neither endpoint alone establishes biological damage.

Increasing analytical attention to replacement and ultrashort-chain PFAS adds a further source of heterogeneity. In an East China Sea fish food web, Cheng et al. [5] investigated emerging ether-based PFAS, while Wang et al. [14] combined target measurements, suspect screening, and isomer analysis in a subtropical marine food web. These studies indicate that chemical coverage and analytical selectivity can alter which compounds appear most prevalent or trophically persistent. At the organism level, biological outcomes have been addressed less directly. Ask et al. [16] documented sex-dependent associations between selected PFAS and circulating thyroid hormones in Arctic kittiwakes, but observational associations are not equivalent to a demonstrated causal toxic effect.

Coral reef-associated systems should not be presented as entirely unstudied. Pan et al. [17] measured PFAS in South China Sea reef fishes. Xu et al. [18] investigated 22 target compounds in reef-building corals, identifying seasonal and interspecific differences, while Zhou et al. [19] expanded attention to ultrashort-chain substances including trifluoroacetic acid in reef-building corals. Yet these studies address different components of the reef ecosystem and employ different

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compound panels and analytical endpoints. Observing PFAS in coral tissues or reef fish does not identify their transfer through an integrated coral reef trophic network, nor does it establish an adverse effect on reef organisms.

Existing marine PFAS syntheses already summarize broad occurrence and accumulation patterns [2, 20]. The contribution sought here is therefore not a claim of being the first marine PFAS review, but a documented comparison that explicitly separates the kinds of inference supported by individual studies. Variation in analytical panels, detection limits, weight bases, tissue selection, food-web definitions, and sampling location makes direct numerical comparisons difficult. Similarly, an exposure association, a tissue concentration, a trophic magnification factor, and a human dietary-risk estimate answer different scientific questions. Organizing this information by taxa, matrices, compounds, and evidence types can identify practical monitoring priorities while avoiding unsupported generalization from broader marine settings to coral reef ecosystems.

This review aims to compare published primary measurements of PFAS in marine and marine-associated coastal organisms, emphasizing the ecological and analytical context of those measurements. Specifically, it examines (i) the taxa, tissues, ecosystems, and geographic regions represented in the assembled evidence; (ii) which PFAS compounds and accumulation patterns are reported in different biological matrices; (iii) what evidence exists for food-web transfer, physiological associations, and risk-related endpoints; and (iv) how the available evidence can inform biomonitoring of marine and coral reef-associated organisms. Throughout, observations from reef organisms are distinguished from extrapolations based on non-reef marine studies.

2. Methods

2.1. Literature identification and study eligibility

This study used a targeted, literature-based comparative review design. Sources were assembled from an existing candidate bibliography, publicly accessible scholarly and publisher records, reference-list tracing from relevant marine PFAS publications, and full-text studies collected and checked during the review development process through 11 October 2026. Working discovery terms combined PFAS, PFOS, PFOA, or "per- and polyfluoroalkyl substances" with marine, coastal, reef, coral, fish, crustacean, mollusk, seabird, marine mammal, occurrence, bioaccumulation, biomagnification, trophic transfer, tissue, biomonitoring, and biological effect. These are topic-search terms; they do not represent a reproducible search executed across all commercial bibliographic databases.

Primary studies were eligible for the core dataset if they were peer-reviewed articles published in the preferred period 2010–2026, had an accessible main text, and reported measured PFAS in marine or marine-associated coastal organisms. Studies of estuarine and tidally influenced brackish food webs were eligible when their organisms or ecological setting had a clear marine connection; these environments were labeled separately from open-marine and coral reef systems. For mixed marine–freshwater datasets, only identifiable marine taxa or marine subsets were used for marine-specific comparisons. Only direct sampling of reef-associated fish or reef-building corals was classified as direct reef evidence. Excluded from the core primary dataset were freshwater-only studies, purely abiotic monitoring without measured marine biota, reviews, policy documents, theses lacking journal peer review, duplicate publications, and source records for which a main paper was not available for the same full-text checking procedure.

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2.2. Data extraction and provenance

For each study, we recorded author, publication year, DOI, habitat and geographic setting, organism group, sampling matrix, PFAS targets, and the evidence types represented. When supported by the main text or supporting information, we extracted sample or species counts, detection frequencies, concentration ranges or summary statistics, bioaccumulation measures (e.g., BCF, BAF, BSAF), food-web measures (BMF and TMF), exposure or physiological associations, and human dietary-risk estimates. Study identifiers and the original literature source were retained with each record. Unreported variables were not inferred, and samples, species, tracks, and composite pools were not treated as interchangeable counting units.

2.3. Harmonization and evidence coding

Concentrations were preserved in their originally reported units and basis, including ng/g wet weight (ww), ng/g dry weight (dw), ng/mL plasma, and μg/kg wet weight; no blanket wet/dry conversion or cross-matrix pooled concentration was attempted. PFAS analyte abbreviations were standardized only when chemical identity was explicit. Analyte-specific data were distinguished from study-specific sums because the compounds contributing to ΣPFAS vary among papers. BCF, BAF, BSAF, BMF, and TMF were kept as separate operational endpoints; point estimates above one was not automatically interpreted as statistically significant magnification without the source's uncertainty or test result.

For descriptive evidence mapping, each article could be assigned multiple codes: O = biological occurrence or concentration; B = an explicit organism/environment accumulation factor; T = a trophic-transfer model or predator–prey magnification comparison; E = a measured organismal physiological or body-condition association; R = modeled human dietary or ecological risk; M = an explicit tissue comparison or biomonitoring application; and Y = a reported temporal, seasonal, or migratory-exposure pattern. These are review-level classifications, not mutually exclusive study designs. Human dietary risk and threshold screening were not counted as direct proof of health effects on sampled wildlife.

2.4. Synthesis and quality checks

We used descriptive tabulation and a structured narrative comparison because taxonomic groups, analyte panels, reporting bases, sampling designs, and statistical methods were too heterogeneous for defensible pooled-effect estimation. The evidence table was checked against the uploaded primary manuscripts. When source-level ambiguities were identified, the original reporting was retained and flagged. Diao et al. [4] report different PFOS TMF values in their results table and conclusion, while a reported risk-reference value in the abstract of Zang et al. [21] differs from the value used later in the article. Other recorded qualifications concern mixed freshwater and marine market samples and studies comparing different tissues or samples collected at different sites or times. These limitations were incorporated into the synthesis rather than treated as missing observations to be filled by inference.

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3. Results

3.1. Characteristics of the assembled literature

The source-audited comparison comprised 29 independent full-text primary studies published from 2011 through 2026. Each contained measured PFAS in at least one eligible biological group. The studies were geographically diverse but unevenly distributed: 15 focused on Chinese study settings, including Hong Kong's tidally influenced wetland and Zhejiang's market-based study; additional settings included the European North Sea and adjacent waters, southern Africa, the Mediterranean, Australia, the northwestern Atlantic, and Arctic or sub-Arctic waters. This distribution describes the assembled papers and should not be interpreted as the global geographic distribution of PFAS research. Three investigations directly sampled reef-associated biota: coral reef fishes [17] and reef-building corals [18, 19].

Table 1. Characteristics of the 29 full-text primary studies in the curated comparative corpus

Study	Study setting	Organism or matrix	Evidence
Loi et al.[6]	Hong Kong; Mai Po coastal tidal wetland	Tidal brackish food web incl. birds	O, T, R
Baduel et al. [22]	Australia; east coast	Stingrays; liver	O, M
Braune et al.[23]	Hudson Bay	Twelve marine prey fish; seabird comparator	O, T
Lam et al. [24]	South China Sea	Dolphins and porpoises; liver	O, R, Y
Fernandes et al. [25]	UK and nearby European seas	Edible marine fishes	O, M
Pan et al. [17]*	South China Sea	Coral-reef fish; muscle	O, R
Fair et al. [8]	Charleston Harbor	Six fish species; whole fish and fillet	O, R, M
Ask et al. [16]	Svalbard	Black-legged kittiwakes; plasma	O, E
Du et al. [12]	Qinzhou Bay	Multi-taxon coastal food web	O, B, T
López-Antia et al. [26]	Belgium; coastal mixed foraging	Lesser black-backed gull chicks; plasma	O, E, M
Pan et al. [13]	Beibu Gulf	Mixed marine food web; biota and abiotic media	O, B, T, R
Savoca et al. [27]	Italy; Palermo coast	Sea urchins; coelomic fluid, cells, gonads	O, B, M
Byns et al. [7]	Belgium; Belgian North Sea	Fish liver and muscle; crustacean whole body	O, B, T, R, M
Cheng et al. [5]	East China Sea	Fish (TMF subset), snails, crabs	O, B, T
Diao et al. [4]	South China Sea	Fish, mollusks, crustaceans; muscle	O, B, T, R
Meng et al. [28]	Bohai Sea	Seven mollusk taxa; freeze-dried tissue	O, R, M, Y
Xu et al. [18]*	northern South China Sea	Reef-building corals; coral tissue	O, M, Y
Wang et al. [14]	northern South China Sea	Fish, crustaceans, cetaceans; whole biota vs liver	O, T, M
Alygizakis et al. [29]	Europe; multiple waters	Marine predator/fish subset, different tissues	O, M
Khalid et al. [10]	St. Lawrence Estuary/Gulf	Marine mammals; muscle and liver	O, M
Léandri-Breton et al. [30]	Svalbard	Migratory black-legged kittiwakes; blood	O, M, Y

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Table 1. (continued)

Marciano et al. [9]	USA/northwestern Atlantic	White sharks; blood plasma and muscle	O, M
Khalid et al. [11]	St. Lawrence Estuary/Gulf	Seven marine mammal taxa; brain	O, M
Liu et al. [31]	Laizhou Bay	Marine food-web organisms	O, B, T, R
Zhang et al. [32]	Zhejiang markets	Bivalves and marine crustaceans; edible tissue	O, R
Estment et al. [33]	South Africa; False Bay	Five intertidal invertebrate taxa	O, B, M
Li et al. [15]	Bohai Sea	60 species; mixed marine food web	O, T, R
Zang et al. [21]	western Yellow Sea	Nine marine species; edible tissue	O, B, T, R
Zhou et al. [19] *	Beibu Gulf	Reef-building corals; tissue and seawater	O, B, M

An asterisk (*) marks direct coral or coral-reef fish sampling. O = occurrence, B = accumulation factor, T = trophic analysis, E = organismal association, R = risk estimate, M = monitoring/matrix, Y = temporal/seasonal/migratory pattern. Mixed and brackish settings remain separately identified.

On the basis of the evidence definitions in Section 2.3, all 29 papers reported organismal occurrence or concentrations; ten reported explicit accumulation-factor evidence; eleven involved trophic-transfer or predator–prey comparisons; two reported organismal physiological or body-condition associations; twelve included dietary or ecological risk estimates; sixteen contributed explicit tissue-comparison or biomonitoring evidence; and four had a primary seasonal, multi-year, or migration-exposure component (Table 2). These categories overlap, and their frequencies are conditional on the purposively assembled corpus.

Table 2. Descriptive evidence-classification counts in the assembled corpus (n = 29)

Code	Classification	Studies
O	Biological occurrence or concentration	29
B	Explicit BCF, BAF or BSAF	10
T	Trophic-transfer model or predator–prey magnification comparison	11
E	Measured physiological/body-condition association	2
R	Modeled dietary or ecological risk	12
M	Matrix comparison or bioindicator/monitoring interpretation	16
Y	Seasonal, multi-year, or migration-linked pattern	4

Codes are not mutually exclusive; this is a source-audit summary, not a systematic-review prevalence estimate.

3.2. Taxonomic and geographic coverage

Taxonomic coverage extended from reef-building corals and intertidal invertebrates to fish, sharks and rays, seabirds, and marine mammals. Direct reef evidence consisted of seven sampled reef-fish speciesin Pan et al. [17]; 68 coral samples representing 12 reef-building speciesin Xu et al. [18]; and 54 coral tissue samplesin Zhou et al. [19]. The latter two studies sampled coral tissue rather than a multi-trophic reef food web, and therefore supplied occurrence, organism-level accumulation, and potential monitoring information but not reef-wide trophic magnification factors. Together, these studies provide direct evidence of PFAS occurrence in coral reef organisms, although their focus remains largely on individual taxa rather than complete reef food webs.

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Non-reef invertebrate studies included Mediterranean Sea urchins [27], False Bay intertidal mussels, limpets, periwinkles, sea urchins, and anemones [33], and 372 mollusk samples from Bohai Sea coastal settings collected over multiple years [28]. Zhang et al. [32] studiedmarket-sourced seafood including 170 bivalve and 69 marine crustacean samples, alongside a separately identified freshwater crustacean subgroup; only marine subgroups were considered in marine-specific comparisons. Mixed-food-web analyses included 60 marine speciesin Li et al. [15], 38 fish/crustacean/cetacean speciesin Wang et al. [14], and nine western Yellow Sea speciesin Zang et al. [21]. Because these values refer variously to individual organisms, species or composite samples, they were not summed into a synthetic count of animals examined. These studies extend the taxonomic and geographic coverage of PFAS research while highlighting differences in sampling approaches and ecological contexts.

Marine vertebrate studies expanded the spatial and ecological range of the evidence.Marciano et al. [9] sampledplasma from 27 white sharks and muscle from 12 of those animals in the northwestern Atlantic.Khalid et al. [10, 11] investigateddistinct tissues from marine mammals in Canada's St. Lawrence Estuary and Gulf, whereasLam et al. [24] investigatedarchived cetacean livers from the South China Sea across 2002–2014. The two kittiwake studies assessed different questions: endocrine associations at breeding sites [16] and relationships between wintering geography and prelaying PFAS burden [30]. The latter analysis included 64 individually tracked kittiwakes and 71 wintering tracks, not 71 different birds. Overall, the available literature provides broad coverage of marine taxa, but direct coral reef evidence remains limited to a small number of fish and coral studies.

3.3. Compound profiles and biological matrices

The analyzed PFAS profiles differed considerably among studies. In the South China Sea reef-fish study, six of sixteen targeted compounds were detected, and PFOS was the predominant analyte, reaching a reported maximum of 27.05 ng/g ww [17]. The coral studies differed in chemical focus: Xu et al. [18] targeted 22 PFAS and reported PFHxS as 43% of the total measured PFAS, whereas Zhou et al. [19] explicitly investigated ultrashort-chain analytes and detected trifluoroacetic acid in all 54 sampled corals, with a reported median of 23 ng/g dw. These are differences between targeted analyte panels and sampling designs; they do not establish a temporal replacement of one dominant coral contaminant by another.

The contrast was also apparent outside reefs.Zang et al. [21] foundPFOA contributed 24.7–61.1% of measured total PFAS across nine western Yellow Sea species; PFOS was important in numerous fish and marine-mammal investigations [7, 10], whereas PFAS composition varied with tissue and species. Emerging ether-based PFAS featured in the East China Sea analysisby Cheng et al. [5], andsuspect screeningin Wang et al. [14] revealedadditional chemical and isomer-specific analytical questions. Broad panels, including the 80 targets investigated in marine-mammal tissuesby Khalid et al. [10, 11], arenot directly comparable with single-analyte studiessuch as Savoca et al. [27], whichinvestigated PFOA in sea urchins.

Biological matrix selection created further variability.Byns et al. [7] reportedranges of mean PFOS values from below quantification to 107 ng/g ww in fish liver and to 24 ng/g ww in muscle across sampled taxa and stations.Fair et al. [8] reportedmean ΣPFAS levels of 12.7–33.0 ng/g ww in whole fish and 6.2–12.7 ng/g ww in fillets across species.Marciano et al. [9] reportedsummed concentrations of frequently detected PFAS from 0.56 to 2.9 ng/mL in white-shark plasma and from 0.20 to 0.84 ng/g ww in muscle; the different physical units rule out direct numerical ratio comparisons. In marine mammals,Khalid et al. [10] reportedmean total PFAS in white-sided

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dolphins of 334 ng/g ww in liver compared with 39 ng/g ww in muscle.Lam et al. [24] reportedmuch larger dry-weight liver concentrations in some South China Sea cetaceans, but these cannot be directly ranked against the St. Lawrence wet-weight values without analytical and matrix comparability.

3.4. Bioaccumulation, trophic transfer, and risk-related endpoints

Field-derived factors provided different answers depending on chemical properties and food web. In the Beibu Gulf,Pan et al. [13] reportedtrophic magnification factors of 1.75 for PFOS and 1.69 for the alternative F-53B.Cheng et al. [5] reportedfactors of approximately 2.3 for 6:2 Cl-PFAES and 1.8 for PFNOBS in a fish-based trophic analysis.Wang et al. [14] foundsignificant magnification for multiple targeted legacy and alternative compounds, including a reported TMF of 4.88 for linear PFOS, while highlighting potential analytical interference in the identification of some candidates. These examples show why an aggregate label of "PFAS biomagnification" is insufficient to describe individual compounds.

Not all study-specific PFOS estimates were above one.Li et al. [15] reporteda PFOS TMF point estimate of 0.78 in a Bohai Sea network, although the same analysis found TMF values exceeding one for several alternative or longer-chain compounds.Zang et al. [21] reporteda PFOS TMF point estimate of 2.02 in a western Yellow Sea food web, but a PFBA value of 0.25.Byns et al. [7] didnot identify a consistent trophic-transfer pattern in the Belgian North Sea sample and cautioned that a narrow trophic range limited interpretation.Diao et al. [4] reportedPFOS magnification, but a numerical discrepancy between its results table (1.78) and conclusion (1.18) was recorded in the evidence audit; the conflicting result was not used in pooled calculations.Braune et al. [23] presentedpredator–prey magnification comparisons, but whole-fish and seabird-liver matrices must not be reinterpreted as same-tissue TMFs. Taken together, these results support compound- and system-specific assessments, not an assumption of universal trophic magnification.

Selected quantitative measures are presented without conversion or pooling in Table 3. They are illustrative examples with different denominators, analyte panels, statistics, and measurement units, rather than a ranking of contaminated ecosystems.

Table 3. Selected original-study quantitative results retained without cross-study pooling

Study / matrix	Analyte / measure	Source-reported result	Qualification
Pan et al. [17], reef fish	PFOS maximum	27.05 ng/g ww	One measured concentration in coral reef fish; not a species-wide mean
Xu et al. [18], coral tissue	ΣPFAS mean ± SD	9.7 ± 5.6 ng/g dw	22-compound coral panel; not directly comparable with Zhou panel
Zhou et al. [19], coral tissue	TFA median	23 ng/g dw	54 coral samples; 100% TFA detection
Byns et al. [7], fish	PFOS liver / muscle	<LOQ–107 / <LOQ–24 ng/g ww	Ranges of mean values; different tissues
Fair et al. [8], fish	ΣPFAS whole / fillet	12.7–33.0 / 6.2–12.7 ng/g ww	Range of species means; distinct sample preparations
Marciano et al. [9], white shark	Frequently detected PFAS total	0.56–2.9 ng/mL (plasma)	N = 27; target-panel-specific total
Marciano et al. [9], white shark	Frequently detected PFAS total	0.20–0.84 ng/g ww (muscle)	N = 12; no direct numerical plasma-to-muscle ratio

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Table 3. (continued)

Khalid et al. [10], white-sided dolphin	ΣPFAS mean, liver / muscle	334 / 39 ng/g ww	Two tissues; group mean, not paired effect estimate
Khalid et al. [11], mammals	PFOS detection frequency	97.5% (brain)	Brain tissue: high prevalence does not establish neurotoxicity
Lam et al. [24], dolphins	ΣPFAS concentration range	136–15,300 ng/g dw	Dolphin liver; cannot compare directly with ww
Meng et al. [28], mollusks	Σ33PFAS range	2.51–1351 ng/g dw	372 mollusk samples; temporal and geographical variation
Savoca et al. [27], sea urchins	PFOA median, coelomocytes / gonads	153 / 195 ng/g	At more contaminated site; original matrix-specific units
Pan et al. [13], marine food web	TMF, PFOS / F-53B	1.75 / 1.69	Compound-specific food-web estimates
Cheng et al. [5], fish food web	TMF, 6:2 Cl-PFAES / PFNOBS	2.3 / 1.8	Fish-based TMF; not all sampled taxa in regression
Wang et al. [14], SCS food web	TMF, linear PFOS	4.88	Isomer-specific trophic regression
Li et al. [15], Bohai Sea	TMF, PFOS	0.78	Point estimate < 1; statistical inference requires uncertainty
Zang et al. [21], Yellow Sea	TMF, ΣPFAS / PFOS / PFBA	2.25 / 2.02 / 0.25	Point estimates; different compounds have different slopes
Léandri-Breton et al. [30], kittiwakes	linPFOS mean ± SD	21.29 ± 9.17 ng/g ww	64 tracked individuals; wintering history related to burden

Examples use distinct source denominators, analyte panels, and original concentration bases. TMF = trophic magnification factor; LOQ = limit of quantification; ww = wet weight; dw = dry weight. Differences among rows must not be interpreted as controlled comparisons.

Measures labeled as ecological or human-health risk were generally modeled exposures or comparisons with screening thresholds, not direct adverse-health observations.Pan et al. [17], Fair et al. [8], and Zhang et al. [32] assessed possible implications of seafood consumption;Zang et al. [21] modeledexposure using local consumption assumptions. In contrast,Ask et al. [16] directlymeasured thyroid hormones and reported sex-dependent PFAS associations in kittiwakes.López-Antia et al. [26] relatedgull-chick PFAS concentrations to maternal transfer, diet and body condition, but did not establish PFAS-induced harm. These endpoints answer different questions and were therefore coded separately.

3.5. Patterns relevant to biomonitoring

Several studies directly addressed the suitability or interpretation of biological monitoring matrices.Savoca et al. [27] reportedmedian PFOA concentrations of 21, 153 and 195 ng/g, respectively, in coelomic fluid, coelomocytes and gonads of sea urchins at the more contaminated Palermo site, supporting tissue-specific rather than undifferentiated whole-organism monitoring.Estment et al. [33] identifiedspecies- and location-dependent PFAS accumulation in South African intertidal invertebrates; their results do not imply all five taxa are equally informative.Meng et al. [28] provideda multi-year mollusk series with a reported decline in legacy PFAS after 2015, illustrating the potential for repeated monitoring while also retaining uncertainty about causal source attribution. For wide-ranging wildlife,Léandri-Breton et al. [30] foundthat the

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prelaying PFAS profile of Arctic-nesting kittiwakes was associated with wintering latitude and time since return, indicating that blood collected at a breeding site may reflect earlier, geographically distant exposure.

4. Discussion

4.1. Main patterns across the literature

The assembled evidence reveals a broad range of marine organisms with measurable PFAS burdens but considerable differences in the information available for interpreting those burdens. All 29 studies documented occurrence or concentrations, yet fewer directly quantified organism-to-environment accumulation or trophic transfer; studies with measured physiological or body-condition associations were scarcer still. This should not be interpreted as a definitive numerical description of the global literature, because the corpus was curated and not obtained from a complete systematic search. It does, however, indicate why the distinction among evidence categories is useful within a heterogeneous research collection.

PFOS was prominent in many fish, seabird, and marine-mammal investigations, but it was not uniformly dominant across the studies. Panels emphasizing ultrashort-chain substances, alternative ethers, or single target compounds necessarily yielded different profiles[5, 19, 27]. Similarly, reporting a higher ΣPFAS value in one paper than another can reflect panel composition, tissue selection or the reporting basis, as well as different environmental exposure. The opposing PFOS TMF point estimates reported in separate Bohai/Yellow Sea and South China Sea food webs reinforce the importance of examining where and how a trophic relationship was measured instead of treating magnification as an intrinsic, constant property of every PFAS in every marine ecosystem[13, 15, 21].

4.2. Mechanisms and methodological explanations

Differences in environmental availability, species ecology, and physiological handling plausibly contribute to the reported heterogeneity. Binding to proteins and phospholipids, rather than preferential accumulation exclusively in lipids, provides a basis for observed liver–muscle and plasma–muscle differences[3]. Differences in prey composition, benthic or pelagic habitat, trophic position, animal size, and elimination kinetics may contribute to contrasting accumulation profiles.Baduel et al. [22], for example, reported a negative association between stingray size/age and liver PFAS in one sampled group. Changes in sources and food-web structure may further influence compound-specific trophic slopes across different regions.

Analytical choices can be equally important. Panel size and the inclusion of ultrashort-chain or replacement PFAS determine whether those substances can be reported at all. Suspect screening can extend coverage but may yield lower-confidence chemical identifications, as demonstrated by the interference problems examinedby Wang et al. [14]. Differencesin detection limits, isotope-based trophic-level assignment, and whether species were pooled into composite samples constrain direct comparisons. Study-specific PFAS sums are not chemically identical when contributing analytes differ; nor are wet-weight and dry-weight observations automatically convertible without measured tissue water content. These methodological distinctions are not simply statistical inconveniences: they affect the substantive interpretation of what biomonitoring can detect.

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4.3. Implications for coral reef-associated food webs

Direct reef-associated measurements provide a defensible starting point for monitoring design.Pan et al. [17] showedPFAS exposure in reef fishes;Xu et al. [18] documentedaccumulation and variation among reef-building coral taxa and seasons; andZhou et al. [19] demonstratedthat ultrashort-chain PFAS can be measured in coral tissues. Together, these studies support including both animal taxa and broader compound panels in reef-oriented surveillance. However, they cannot presently be combined into a single reef trophic magnification model: they involve different organisms, collection strategies, target lists, and exposure metrics rather than matched consumers and prey from the same connected food web.

Findings from non-reef marine ecosystems offer hypotheses, not transferable reef effect sizes. Food-web studies have shown that certain legacy and alternative PFAS can magnify across trophic positions, while others exhibit dilution or no consistent trend [7, 13, 15]. Reef monitoring would be strengthened by coordinated sampling of water, sediment or particles, reef-building corals, invertebrate prey, and fish across measured trophic positions, using common analyte panels and time points. Because coral assemblages and reef-dependent consumers do not all share the same habitat use or tissue kinetics, their data should remain taxonomically explicit. Finally, existing coral PFAS occurrence studies should not be presented as demonstrating declines in coral health without direct paired biological measurements.

4.4. Biomonitoring and bioindicator implications

An appropriate monitoring organism depends on the specific question. Relatively site-associated invertebrates may be useful for describing local contamination, provided their tissue-specific responses and exposure conditions are characterized. The sea-urchin findings ofSavoca et al. [27], themulti-species intertidal comparison ofEstment et al. [33], andthe Bohai Sea mollusk time series ofMeng et al. [28] illustratecomplementary possibilities for local, taxonomic, and temporal monitoring. Market-collected seafood can provide consumer-exposure information, as inZhang et al. [32], butcannot always be assigned to a local marine pollution source without verified harvest origin.

Mobile vertebrates answer different questions. Fish muscle may be appropriate for human dietary exposure estimates, while liver, blood, or whole-body sampling may offer different signals of exposure and accumulation [7-9]. Long-lived sharks and marine mammals can integrate exposure across broad spatial and temporal scales, which complicates their use as stationary local indicators [10, 11, 24]. For seabirds, the migratory history and breeding stage can affect how a blood or egg measurement relates to local contamination;Léandri-Breton et al. [30] provideempirical evidence that winter conditions may remain detectable in prelaying blood. Such animals can serve as sentinels of distributed marine exposure, but their movement must be included in interpretation.

A defensible monitoring program should therefore define its intended output before choosing taxa and matrices: local environmental levels, food-web transfer, long-term contamination trends, human seafood exposure, or actual animal physiological responses. The same species is not necessarily equally informative for all of these objectives. Biomonitoring recommendations should also identify which inferences are directly supported and which require validation through co-sampled water, sediment, prey, and biological-response measurements.

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4.5. Limitations and future research

This review has significant methodological limits that must frame its conclusions. First, its 29 studies form a targeted full-text-available collection rather than the output of a complete systematic search of Web of Science, Scopus, or a fully exported Google Scholar result set. The selection is therefore vulnerable to availability and discovery bias; frequencies in the evidence map are descriptive properties of this corpus and cannot be interpreted as prevalence estimates across the whole field. Second, the strong representation of Chinese coastal settings limits broad geographic inference. Third, article-specific analyte panels, detection limits, sampling dates, organ choices, and concentration units prevent formal meta-analysis without more detailed harmonized measurements.

Some individual articles also require caution. Diao et al. [4] contain inconsistent reported values for a PFOS TMF; results requiring that exact number should be checked with authors or corrected-source materials. Khalid et al. [10] contains ambiguity in the reporting of a tissue-correlation test, and sample-count descriptions warrant verification for some mammal datasets. In Zhang et al. [32], the freshwater subgroup must remain excluded from marine-only comparisons, while a risk threshold inconsistency appears in Zang et al. [21]. Finally, ecological risk-screening ratios and correlations with endocrine or body-condition measures do not demonstrate causal effects on the sampled organisms.

Future work should prioritize harmonized, co-located and preferably repeated sampling across seawater, sediments or particles, coral tissues, reef-associated invertebrates and fishes, together with measurement of trophic position. Reporting should include the complete analyte list, detection and quantification limits, individual versus pooled sample counts, original concentration units, and uncertainty intervals for BAF, BMF and TMF. Emerging and ultrashort-chain compounds deserve coverage alongside legacy PFAS, while targeted physiological endpoints should be measured separately from chemical burdens. A genuinely comprehensive systematic update would also require dated, reproducible searches in accessible bibliographic databases and full screening records before publication frequencies or global gaps could be quantitatively generalized.

5. Conclusion

The 29 primary studies assembled in this comparative review document PFAS occurrence across reef-building corals, fish, invertebrates, seabirds, and marine mammals, but they do not support a single uniform pattern of accumulation or trophic transfer. Concentration profiles depend on chemical coverage, biological matrix, geographic setting, and species ecology. Trophic magnification was reported for selected legacy and emerging PFAS in some marine food webs, while other studies reported dilution or inconsistent relationships. Evidence of chemical occurrence, organismal accumulation, trophic transfer, modeled human or ecological risk, and measured physiological associations must therefore remain analytically distinct.

Direct studies of coral reef fishes and reef-building corals establish that reef-associated organisms can contain both legacy and ultrashort-chain PFAS. However, they do not yet characterize transfer through complete, matched coral reef food webs or provide sufficient organism-level evidence to attribute adverse reef health outcomes to measured PFAS burdens. The next step is coordinated, compound-specific monitoring spanning reef organisms, environmental exposure media, trophic relationships, and clearly defined biological endpoints. More transparent sampling and reporting will improve the usefulness of marine organisms as PFAS sentinels without extending individual findings beyond what the original measurements can demonstrate.

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Data availability

Study-level descriptors, evidence classifications, selected original quantitative results, DOI links, and source-level limitations supporting this synthesis are provided in the accompanying evidence audit workbook. No original animal sampling or experimental PFAS measurements were performed for this review. The cited primary publications remain the authoritative sources for their underlying results.

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