

Micro- and Nanoplastic Water Contamination: Comparative Review of Classical Analytical Methods and Advanced Sensors

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Abstract

Micro- and nanoplastics have emerged as persistent, mobile water contaminants capable of carrying co-pollutants such as phthalates, polycyclic aromatic hydrocarbons and antibiotic-resistant pathogens, yet detection methods for this size range remain fragmented across classical analytical chemistry and a rapidly diversifying set of electrochemical, optical and biosensor platforms. This review compiles and classifies 53 original research articles published between 2021 and mid-2026, identified through a PRISMA-guided search of Google Scholar and ScienceDirect, covering detection principle, sensor type, sample matrix, target polymer, water source and particle size class. Optical techniques account for over half of the studies, mass spectrometry and electrochemical sensors divide most of the remainder, and biosensors show the fastest growth in recent years. Water dominates as the tested matrix, but seven in ten water-based studies rely on laboratory-spiked or unspecified samples rather than named environmental water, and nearly half now address the nanoplastic fraction. These findings are interpreted alongside contamination pathways, including atmospheric deposition, and against current EU and WHO frameworks, which place microplastics and antimicrobial resistance indicators on a watch list pending harmonised testing rather than binding limits. The review identifies matrix realism, nanoplastic capability and monitoring frequency as the central bottlenecks separating current detection science from field-ready water monitoring.

Keywords: microplastics; nanoplastics; water contamination; electrochemical sensors; optical sensors; biosensors; detection methods; systematic review; water regulation; PRISMA

1. Introduction

Although water covers about 71% of the Earth's surface, the fraction available for direct human use is very limited. Approximately 96.5% of global water resources are saline, while only about 2.5% are freshwater, and of this freshwater nearly 69% is locked in glaciers and polar ice caps [1]. Human use relies on about 30% of freshwater, which occurs mainly as groundwater, leaving less than 1% readily accessible in rivers, lakes and other surface waters [2]. This limited fraction is under increasing pressure from human demand, with agriculture accounting for roughly 70% of global freshwater withdrawals [3]. The urgency of safeguarding water quality is further underscored by World Health Organization (WHO) and UNICEF, which warned in 2025 that around 2.1 billion people still lacked access to safely managed drinking water, including 106 million who relied directly on untreated surface water sources [4]. In parallel with this quantitative



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scarcity, water resources are increasingly affected by chemical and particulate contamination, among which microplastics and nanoplastics have emerged as contaminants of major concern [5].

Since Thompson et al. introduced the term microplastics [6], plastic pollution has evolved from a largely visible litter problem into a complex particulate contaminant issue spanning aquatic, terrestrial and atmospheric compartments. Since the 1950s, global plastic utilisation has increased dramatically, and a substantial fraction of the material produced has accumulated in landfills or dispersed into the environment [7]. This trajectory has been confirmed by more recent tracking: annual global plastics production and use rising from 234 million tonnes in 2000 to 460 million tonnes in 2019 and projected to reach 736 million tonnes by 2040 in a business-as-usual scenario [8]. Unlike biodegradable organic matter, plastics persist and progressively fragment rather than fully mineralize, generating smaller particles with higher surface area, enhanced reactivity and greater potential for transport and biological interaction [9]. At the nanoscale, these particles behave more like colloids and may more readily cross biological barriers, increasing both their environmental mobility and their toxicological relevance [9,10]. Their occurrence has been documented in surface water, groundwater, drinking water, rainwater, seawater and atmospheric deposition, confirming that water is both a major transport medium and a direct exposure route [11].

Beyond their physical presence, microplastics and nanoplastics can act as carriers of co-contaminants and microbial hazards. For instance, plastic surfaces efficiently adsorb phthalates and polycyclic aromatic hydrocarbons (PAHs) while also serving as vectors for pathogens and antibiotic resistance genes, which increases the ecological and public health significance of these particles [12,13]. A growing body of toxicological evidence links such exposure to oxidative stress, inflammatory responses and other adverse cellular effects across ingestion, inhalation and dermal routes, although a rapid systematic review of digestive, reproductive and respiratory outcomes found that the direct human evidence base still rests mainly on animal studies rather than human observational data [14–16]. The WHO reached a broadly similar conclusion in its first dedicated assessment of microplastics in drinking water, finding no evidence of risk at then-current exposure levels but explicitly flagging the poor quality and inconsistency of the underlying occurrence data as the main limitation on that conclusion [17]. In regulatory and hazard-based discussions, microplastics and nanoplastics are increasingly considered alongside the concept of persistent, mobile and toxic substances, a classification originally developed for organic water contaminants that slip through conventional treatment. This reflects how durability and transport potential can matter as much as acute toxicity in long term exposure risk [18].

Analytical detection remains one of the central challenges in the field of water contamination with micro- and nanoplastics. However, standardised workflows are still lacking, especially for nanoplastics and for microplastics smaller than 10 μm [19,20]. These limitations together with the need for harmonised protocols and consistent data reporting across laboratories have driven the rapid development of advanced analytical sensing platforms.

This systematic review presents both classical analytical methods and advanced sensor-based approaches for detecting micro- and nanoplastics in water. Studies published between 2021 and mid-2026 were classified according to detection principle, including mass spectrometry, vibrational spectroscopy, electrochemical sensing, plasmonic sensing, fluorescence, hyperspectral imaging, biosensors and AI-assisted approaches. The resulting dataset shows that optical methods remain the dominant group, while electrochemical sensors are among the fastest growing categories, particularly in the most recent years, a pattern

consistent with recent syntheses of the sensor literature [19,21]. In particular, we compare established analytical workflows and emerging sensor technologies in terms of their sensitivity, selectivity, matrix compatibility, portability and readiness for field deployment. In addition, we highlight a cross-cutting bottleneck framework derived from the classification of matrix realism, nanoplastic capability, and monitoring frequency gaps that recur across otherwise very different platforms.

2. Methodology

To systematically review analytical methods used for micro- and nanoplastic assessment in water, published articles were retrieved from Google Scholar and ScienceDirect following the PRISMA guideline [22]. Combinations of the terms “microplastics in water”, “nanoplastics in water”, “sensors” and “detection method” were used to select publications between January 2021 and mid-2026. Of them, original articles that met the inclusion criteria of full-text availability, a reported plastic type, a described detection method, a defined water source or other sample matrix, and an original, non-review study design were selected. In addition, detailed detection technique, sensor category, target polymer, working matrix, water source and target particle size class were included in the selection. The complete study-level dataset is provided in Table S1 (Supplementary Materials). On ScienceDirect, results were restricted using the platform’s built-in Article Type filter to exclude review articles, consistent with the original, non-review inclusion criterion stated above. Google Scholar does not offer an equivalent filter, so review-type records returned by Google Scholar were removed manually during title and abstract screening. Because the original search was not separately logged at the time it was run, the identification-stage counts reported here (Google Scholar $n = 224$, ScienceDirect $n = 109$ after excluding 53 review-tagged records, total $n = 333$) were confirmed by re-running the same search terms on both platforms on 9 July 2026; of these 333 records, 124 were duplicates or otherwise overlapping between the two sources, leaving the 209 records taken forward to screening reported above. Screening against the inclusion criteria was carried out on title, abstract and full text; individual reasons for the 156 records excluded at this stage were not retained in the original review records and are not reconstructed here. The quantitative patterns emerging from this classification are woven into the sections that follow alongside the narrative literature, integrating classical methods with the latest sensor developments. This systematic review is not a formal quantitative meta-analysis: no effect-size synthesis or heterogeneity statistics were computed. Therefore, the absence of per-record exclusion reasons and formal risk-of-bias scoring, while noteworthy, does not affect any quantitative pooled estimate. The completed checklist and flow diagram are provided in Figure 1 and Supplementary Table S2. The review protocol was not formally preregistered in a public registry (e.g., OSF, PROSPERO, or Inplasy) before data extraction.

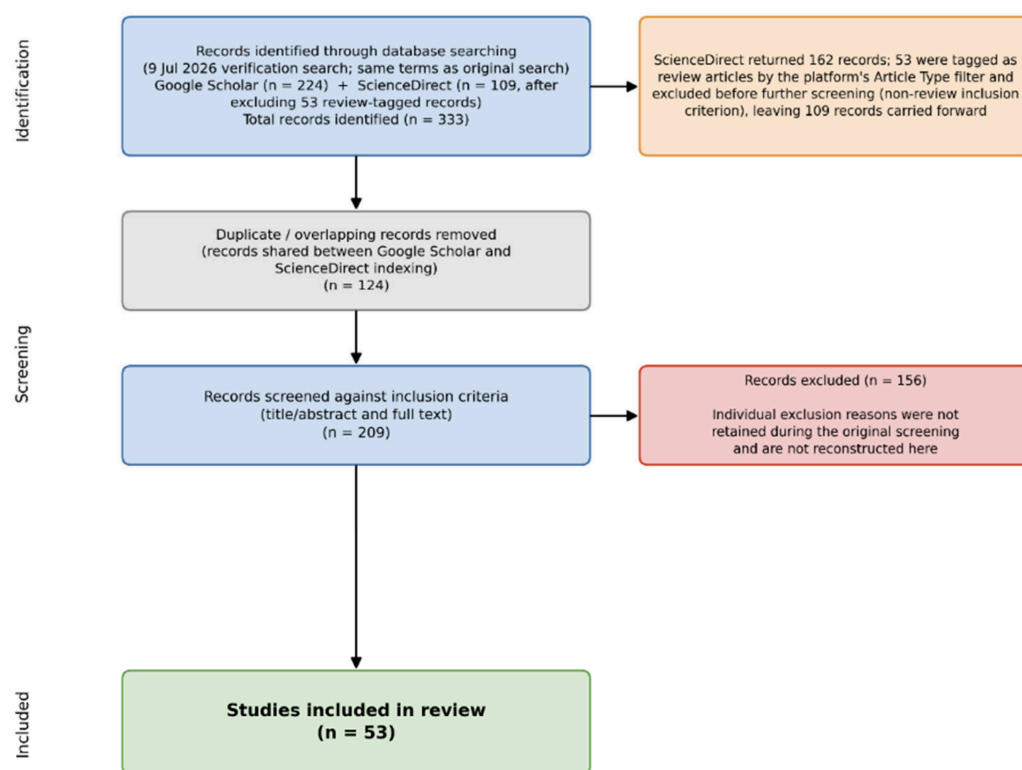


Figure 1. PRISMA 2020 flow diagram of the record identification, screening and inclusion process. Identification-stage counts (Google Scholar $n = 224$, ScienceDirect $n = 109$, total $n = 333$) were confirmed by a verification search run on 9 July 2026 using the same search terms as the original search; the original search was not separately logged. Duplicate/overlap removal ($n = 124$), screening ($n = 209$) and inclusion ($n = 53$) figures are as reported in the main text. Individual reasons for the 156 records excluded during screening were not retained in the original review records and are not reconstructed here.

3. Micro- and Nanoplastic Diversity

Micro- and nanoplastics represent a persistent and toxicologically relevant group of contaminants in aquatic environments, particularly due to their ability to act as carriers of organic pollutants such as phthalates and PAHs. Microplastics, generally defined as plastic particles smaller than 5 mm, and nanoplastics, typically considered to be below 1 μm , originate from the fragmentation of larger plastic debris under the influence of ultraviolet (UV) radiation, mechanical abrasion, oxidation and biological processes [23].

Across the 53 studies compiled for this review, eight polymer types account for essentially all targeted particles: polyethylene (PE), polypropylene (PP), polystyrene (PS), polyvinyl chloride (PVC), polyethylene terephthalate (PET), polyamide (PA, nylon), poly(methyl methacrylate) (PMMA) and polycarbonate (PC) (Figure 2, Table 1). These polymers differ substantially in backbone chemistry: PE and PP are simple polyolefins formed by addition polymerisation of ethylene and propylene, PS carries a bulky aromatic side group, while PVC, PET, PA, PC and PMMA are heteroatom-containing polymers produced by radical or step-growth (condensation) polymerisation. This chemical diversity translates directly into different densities, surface polarities and degradation pathways [24].

None of these polymers occurs naturally: all originate from large-scale petrochemical manufacturing and enter the aquatic environment secondarily, through the fragmentation of packaging, textiles, tyres, construction materials and other consumer products [23,24]. Once released, a polymer's density relative to water is the single most important property governing its subsequent environmental cycling: the low-density polyolefins PE and PP ($\rho \approx 0.89\text{--}0.96 \text{ g/cm}^3$) remain buoyant and are transported predominantly at the water

surface by wind and surface currents, whereas denser polymers such as PVC, PET, PA and PC ($\rho \approx 1.14\text{--}1.45 \text{ g/cm}^3$) sink and accumulate preferentially in sediments [25]. PS occupies an intermediate position, with the solid resin close to neutral buoyancy while its expanded polystyrene (EPS) foam form remains highly buoyant.

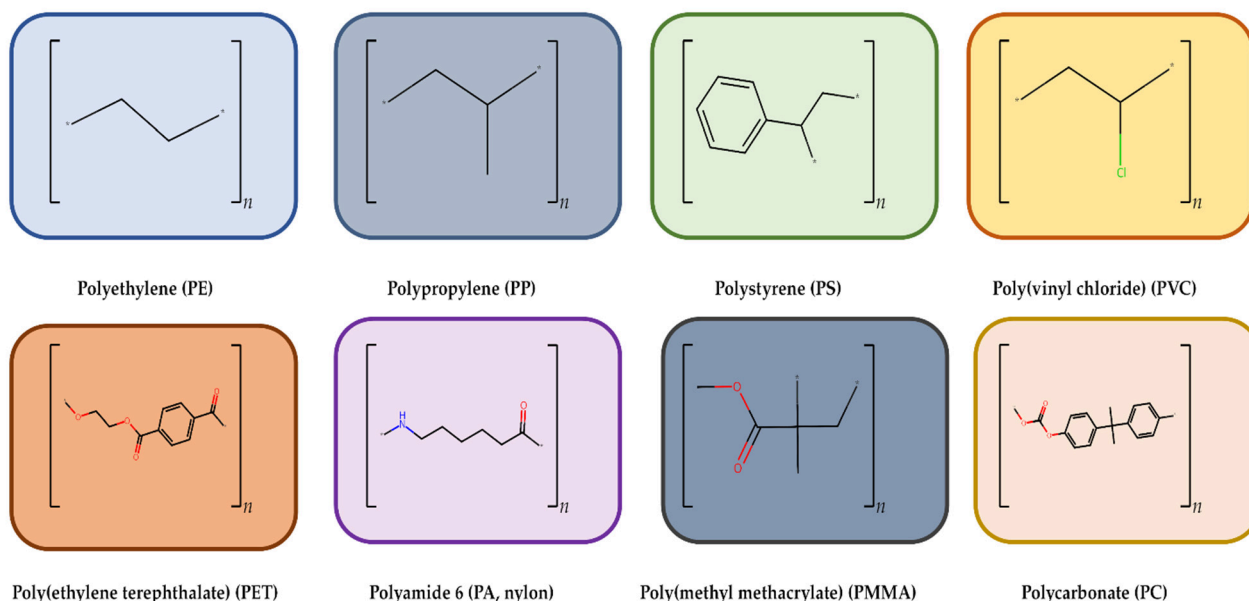


Figure 2. Structural formulas of the repeating unit of the eight polymer types most frequently targeted across the reviewed studies. Structures were generated with RDKit (2D coordinates); square brackets and the subscript n denote the repeating unit of each polymer chain.

Table 1. Main physicochemical properties, structure and typical sources of the eight polymer types most frequently targeted across the reviewed studies.

Polymer	Abbr.	Repeating Unit/Monomer	Density (g/cm ³)	Buoyancy in Water	Main Sources/Products
Polyethylene	PE	$-(\text{CH}_2-\text{CH}_2)_n-$; ethylene	0.89–0.96	Floats	Packaging films, bags, bottles, pipes
Polypropylene	PP	$-(\text{CH}_2-\text{CH}(\text{CH}_3))_n-$; propylene	0.90–0.92	Floats	Food containers, textiles, ropes, bottle caps
Polystyrene	PS	$-(\text{CH}_2-\text{CH}(\text{C}_6\text{H}_5))_n-$; styrene	0.96–1.06 (foam ~0.02–0.05)	Near-neutral/foam floats	Disposable cutlery, foam packaging, insulation
Polyvinyl chloride	PVC	$-(\text{CH}_2-\text{CHCl})_n-$; vinyl chloride	1.16–1.58	Sinks	Pipes, cable insulation, flooring
Polyethylene terephthalate	PET	Ester of terephthalic acid + ethylene glycol	1.37–1.45	Sinks	Beverage bottles, textile fibres (polyester)
Polyamide (nylon)	PA	Amide-linked diamine + dicarboxylic acid (or lactam)	1.02–1.16	Near-neutral/sinks	Textile fibres, fishing lines and nets
Poly(methyl methacrylate)	PMMA	$-(\text{CH}_2-\text{C}(\text{CH}_3)(\text{COOCH}_3))_n-$; methyl methacrylate	1.15–1.19	Sinks	Acrylic glass, coatings, cosmetic microbeads
Polycarbonate	PC	Carbonate-linked bisphenol A units	1.20–1.22	Sinks	Optical media, electronics housings, bottles

This idealised float-sink pattern is progressively modified over time by biofouling and aggregation, whereby microbial biofilms and adhering organic matter increase a particle's

effective density and drive even initially buoyant PE and PP towards the sediment [25]. Colonisation by microorganisms simultaneously converts each particle into a distinct, transportable microhabitat known as the plastisphere, allowing micro- and nanoplastics to redistribute both themselves and their attached microbial communities well beyond their point of release [26]. Beyond the aquatic compartment itself, the same fragmentation processes generate airborne fibres and fragments that undergo atmospheric transport and are subsequently returned to surface waters through wet deposition [27].

Determining polymer identity is not equally reliable across these eight types. Spectroscopic misclassification is a documented risk between PVC and PET specifically: an inter-system comparison of Fourier-transform infrared and Raman spectra found that PVC's highest spectral hit-quality index was against the PET reference rather than its own, an effect attributed to overlap between the phthalate plasticisers commonly blended into PVC and the ester carbonyl bonds intrinsic to PET [28]. Environmental weathering compounds this unevenly across polymers: a systematic comparison of polystyrene, PET and nylon 6 under controlled weathering conditions showed that each polymer's Raman and infrared signature drifts away from its pristine reference spectrum at a different rate and along a different chemical pathway, so a spectral library calibrated on virgin resin loses reliability progressively, and to a different extent for each polymer, as particles age in the environment [29]. A parallel limitation affects thermal methods: comparing decomposability with detectability across a wider polymer set showed that some polymers resist thermal breakdown under pyrolysis-GC-MS conditions, producing false negatives specifically for those polymers even though the same method performs well on more readily pyrolysed resins [30]. The practical consequence is that no single method offers uniform confidence across the eight-polymer taxonomy summarised in Table 1, and reported detection frequencies for the least commonly identified polymers in this dataset (Table 1, Figure 2) should be read as a lower bound on their true environmental occurrence rather than a precise estimate.

Micro- and nanoplastics are water contaminants not only due to their high persistence but also because they are carriers of other organic micropollutants. Recent studies show that rainwater can contain measurable concentrations of PAHs, phthalate esters, and BTEX compounds (benzene, toluene, ethylbenzene, and xylene), originating from atmospheric washout processes [27]. Beyond the organic compounds rainwater can carry micro- and nanoplastic particles that function as a distinct microbial habitat, commonly referred to as the plastisphere [26]. Typically, microorganisms form biofilms on plastic particles, within which they are more resistant to environmental stresses than when they are freely suspended in water [31–33]. Biofilms associated with microplastic particles have consistently been shown to enrich human pathogenic bacteria alongside antibiotic resistance genes at concentrations significantly exceeding those in the surrounding water column, a risk further amplified by horizontal gene transfer among colonising species [34–36]. Because these plastic particles remain largely intact as they migrate from diverse sources, such as wastewater outfalls, agricultural runoff, or urban stormwater drains, into aquatic ecosystems, they serve as highly mobile reservoirs. This mobility facilitates the environmental persistence and long-range dissemination of both pathogens and resistance genes far beyond their original point of release.

4. Pathways of Water Contamination with Micro- and Nanoplastics

The presence of micro- and nanoplastics in water systems is primarily linked to the fragmentation of larger plastic debris but also to emissions originating from textile use and industrial activities [37]. In addition, atmospheric transport has emerged as an important, though still insufficiently addressed, contamination mechanism.

Contamination via atmospheric pathways is strongly influenced by local meteorological conditions and rainfall intensity, as demonstrated in urban environments [27,38]. Airborne microplastic fibres have been identified in atmospheric deposition, with early evidence from urban areas such as Central London, where fibres accounted for up to 92% of detected particles [39]. Deposition rates of fibres were shown to be significantly higher than at remote sites, a pattern since confirmed across a wider range of urban and remote locations and linked to inhalation as a direct human exposure route [40]. These particles are removed from the atmosphere through precipitation and subsequently introduced into surface waters and stormwater systems. Indeed, atmospheric deposition is increasingly recognised as a relevant pathway of water contamination by micro/nanoplastics, alongside direct discharges and surface runoff. Importantly, atmospheric deposition can affect water bodies that are not directly exposed to point source pollution. Remote lakes, protected aquifers recharged by precipitation, and harvested rainwater used for drinking purposes may all be subject to contamination. Evidence shows that airborne microplastic fibres can be transported over long distances, reaching even remote regions such as the Southern Ocean and Antarctica [39,41].

Atmospheric precipitation may increase the concentration of organic micropollutants in water bodies, where the washout of compounds such as BTEX correlates with rainfall intensity and meteorological conditions [38]. Consequently, a single precipitation event can simultaneously introduce both particulate and dissolved contaminants into aquatic systems. Similar temporal variability has been reported for microplastics in urban stormwater, where concentrations, particle size distribution, and morphology differ between rainfall events and dry-weather conditions. Higher concentrations and a greater proportion of smaller particles are typically observed during and immediately after storm events [42]. This variability further complicates an already significant methodological issue in microplastic research. A meta-analysis of over one hundred studies showed that sampling methodology alone can lead to differences in reported concentrations spanning several orders of magnitude [43]. Specifically, small-volume grab samples and large-volume net sampling often yield substantially different results even when conducted at the same location and time. These limitations suggest that conventional monitoring approaches, particularly those relying on periodic grab sampling, are not well suited to capturing short-term contamination peaks associated with rainfall events. Consequently, there is a considerable risk that current assessments underestimate actual exposure levels and peak contaminant loads.

5. Methods for Water Contaminant Monitoring

Conventional analytical techniques, such as atomic absorption and atomic fluorescence spectroscopy, inductively coupled plasma mass spectrometry (ICP-MS), gas chromatography–mass spectrometry (GC-MS), and liquid chromatography–tandem mass spectrometry (LC-MS/MS), remain the standard reference methods for regulatory water quality monitoring, relied upon for their high accuracy and low detection limits [44,45]. Although accurate and sensitive, these methods consistently depend on expensive, bench-top instrumentation, specialised personnel, and lengthy sample preparation. Consequently, while well-suited to scheduled compliance testing, they lack the temporal resolution needed to capture transient, event-driven contamination, such as storm-flushed pollutant pulses or intermittent industrial discharges.

This gap between the rapid dynamics of real-world contamination and the low temporal resolution of traditional laboratory methods has driven the development of chemical and biosensor platforms. By translating recognition of a target water contaminant into an electrochemical or optical signal, these devices enable rapid, low-cost, and potentially continuous, field-deployable detection [46,47]. Biosensors are primarily categorised by

their transducer type and operating principle, as reliable transduction is essential for signal reproducibility. Optical biosensors exploit properties such as absorbance, fluorescence, and chemiluminescence, employing fibre optics or optoelectronic transducers alongside bioreceptors that provide specificity and sensitivity of detection [47]. A specialised subset, plasmonic biosensors, including surface plasmon resonance (SPR), localised surface plasmon resonance (LSPR), and surface-enhanced Raman scattering (SERS), relies on subwavelength light interactions with nanostructured conducting interfaces or metallic nanoparticles [48]. Ultimately, all optical biosensing platforms require optoelectronic conversion to translate optical responses into digital or electronic outputs for processing. Electrochemical biosensors use this principle via a transducer that translates bioreceptor-analyte binding events into quantifiable electrical signals caused by electron transfer variations. In these systems, the bioreceptor acts as the recognition unit, while the transducer converts the molecular interaction into a readable signal [49]. Signal detection relies on four primary electrochemical techniques: potentiometry (measuring cell potential with zero or constant current), amperometry (measuring current at a fixed potential), electrochemical impedance spectroscopy (EIS, measuring the frequency-dependent alternating current response to a sinusoidal potential), and voltammetric methods, such as cyclic voltammetry (CV), differential pulse voltammetry (DPV), and square wave voltammetry (SWV), analysing current output across a dynamic potential range. By translating recognition of a target water contaminant into an electrochemical or optical signal, these devices enable rapid, low-cost, and potentially continuous, field-deployable detection [45,46].

Portable biosensors offer a practical way to monitor pollution dynamics that scheduled grab sampling entirely misses. Biosensors yield rapid signal responses (can be seconds or minutes) and enable continuous monitoring, whereas classical spectroscopy requires lengthy mapping, single-particle scanning, or long chromatographic run times. Moreover, traditional laboratory-based techniques require rigorous, multi-step sample treatment to isolate particles from environmental matrices. Biosensors can often operate directly in water media with reduced sample preparation.

Across the 53 recent studies compiled for this review, optical techniques account for 28 studies (52.8%), mass spectrometry for 12 (22.6%) and electrochemical sensors for 11 (20.8%), with the remaining two studies (3.8%) built around mechanical or gas-sensing principles that fit neither category (Figure 3).

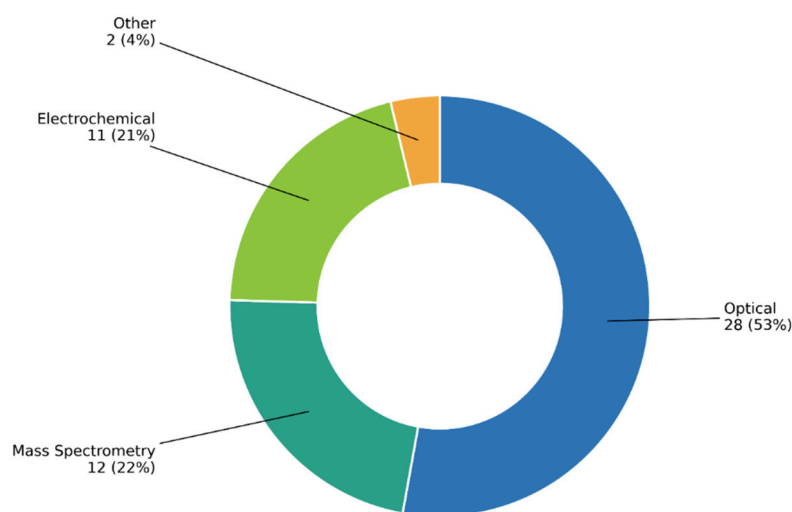


Figure 3. Distribution of the 53 reviewed studies by detection method, grouped by underlying transduction principle. The whole-number percentages are rounded using the largest-remainder method and therefore sum exactly to 100%.

Electrochemistry and optical sensing together already outnumber mass spectrometry by more than three to one, a reversal from the field's origins, when thermal and chromatographic techniques were essentially the only tools available. The pace of that reversal matters as much as its scale: the electrochemical share is concentrated almost entirely in papers from 2023 onward, indicating that this is an active, ongoing shift rather than a settled distribution.

Across the full dataset, not only the classical subset, polystyrene appears in twenty-nine of the fifty-three studies (54.72%), roughly twice as often as polyethylene, the next most common target (Figure 4). This is a matter of convenience rather than environmental relevance. Commercial PS nanospheres are monodisperse, cheap and easy to source at defined sizes, which makes them an obvious calibration standard. However, they poorly represent real samples that contain mixed, weathered, irregularly shaped debris. That mismatch between what is convenient to test and what is actually present in the environment will be discussed below.

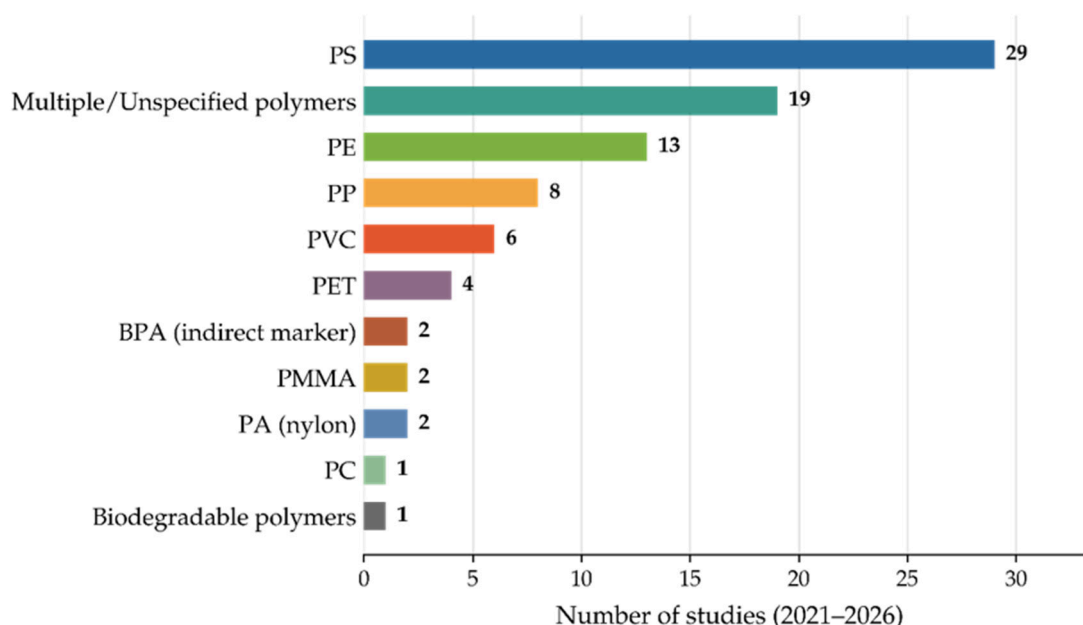


Figure 4. Frequency of target polymer types across the reviewed studies. A study may target more than one polymer, so the sum of mentions exceeds the number of studies (53).

6. Classical Analytical Methods

Classical methods such as pyrolysis gas chromatography mass spectrometry (Py-GC/MS) and thermal extraction desorption GC/MS (TED-GC/MS) were originally developed to characterise organic polymers in real sediment and water samples. These methods provide polymer-specific, mass-based quantification and are often treated as reference approaches, particularly in complex matrices such as sediments, soils and biological tissues [30,50]. Py-GC/MS breaks a filtered particle down into characteristic pyrolysate fragments that a mass spectrometer can match against a reference library, providing a genuinely polymer-specific quantitative result rather than merely a count of particles of uncertain composition. For instance, PE pyrolyses into characteristic, matrix-interfering sequences of n-alkanes and n-alkenes (commonly measured from C16 to C26); polystyrene yields styrene monomers, dimers, and highly specific styrene trimers (5-hexene-1,3,5-triyltribenazene); PP produces specific pyrolysis products such as 2,4-dimethyl-hept-1-ene; while PET and polycarbonates under direct thermal degradation conditions produce diverse polar compounds, leading to poor chromatography and lower sensitivity. Adding derivatisation agents like tetramethylammonium hydroxide (TMAH) in thermochemolysis induces ester- and ether-

cleavage followed by methylation, producing highly specific, readable derivatives. Finally, styrene-butadiene rubber (SBR) breaks down into specific indicators like butadiene dimer and 4-phenylcyclohexene.

Mass spectrometry, in its various pyrolysis and desorption forms, remains the single largest technique category in the dataset compiled for this review, accounting for 20 of the 53 studies, a dominance that is not accidental given this historical grounding in polymer characterisation. Py-GC/MS therefore offers an advantage repeatedly highlighted in broader assessments of chemical analysis methods for micro- and nanoplastics [51]. Li et al. [52] combined GC/MS with a TMAH digestion and dichloromethane extraction step to recover microplastics and nanoplastics from sediment and soil, matrices where organic and mineral interference would otherwise swamp the pyrolysate signal. Ccancapa-Cartagena et al. [49] adapted the same protocol for detection of seven common polymers in tap water by streamlining the sample preparation to a twenty-three-minute run and reporting linearity above 0.996. This quantitative approach was extended with a soil-specific validated Py-GC/MS workflow targeting synthetic and environmental soils [53]. Low detection limits of 0.02–0.44 µg were achieved and high recovery rates across sandy, loamy and sandy-loam soil types, although extensive pre-cleaning remained necessary for organic-rich samples.

TED coupled to GC/MS, which pyrolyses an entire filter in a tube furnace, was developed specifically to avoid the sampling bias that comes from cutting a filter into pieces before analysis. Sorolla-Rosario et al. [50] showed that this whole-filter approach gave more consistent results across common polymers, including PE, PP, polystyrene (PS), polyethylene terephthalate (PET) and polyvinyl chloride (PVC). Kittner et al. [30] later used the same technique to compare decomposability against detectability across a wider polymer set. This study indicated that some polymers resist thermal breakdowns producing false negatives, a limitation that can be overlooked if a method is only validated on thermosensitive polymers. Among the mass spectrometry methods, matrix-assisted laser desorption/ionisation (MALDI)-based platforms have begun to extend detection into matrices that were previously out of reach for this type of protocols. For instance, optimised silver cluster ionisation improved signal intensity for PS nanoplastic detection [54], hollow laser desorption ionisation enabled high-throughput analysis of airborne nano- and microplastics collected on ordinary portable air samplers without any pre- or post-treatment [55], while a trapped ion mobility variant achieved spatial mapping of PS microspheres inside whole mouse tissue sections [56]. These studies illustrate how far mass spectrometry has been pushed beyond its original applications for micro- and nanoplastic detection in sediment and water.

In parallel, vibrational spectroscopic techniques, especially Fourier-transform infrared spectroscopy (FTIR) and Raman microscopy, allow particle-level identification without fully destroying the sample. However, their performance is constrained by acquisition time, matrix interference and size-dependent detection limits [57,58]. Only three studies in the present dataset used FTIR or µFTIR. A direct comparison on the same set of drinking water filters found that µRaman resolved particles down to about 1.0 µm while µFTIR missed the majority of particles below 50 µm, illustrating how strongly the choice of instrument can shape the reported result on identical samples [57]. A related reflectance-based µFTIR method coupled with multivariate analysis has since improved the technique's ability to track photodegradation on weathered particles, although neither approach fully solves the problem of rapid, field adaptable monitoring [59].

Taken together, classical analytical methods share a common profile: high confidence in polymer identity but long run times, destructive sample handling, and bulky instrumentation.

7. Sensors for Micro- and Nanoplastic Detection

Once reference analytical instrumentation studies are set aside, the remaining 38 studies in our dataset describe sensor development for micro- and nanoplastic detection (Figure 5). In Figure 5 sensors are classified based on their recognition mechanism and not transduction principle. Consequently, one electrochemically transduced study is grouped under biosensors because its selectivity derives from an aptamer/CRISPR-Cas12a recognition element rather than from the electrode surface alone [60]. In contrast, in Figure 3 sensors are classified according to their transduction principle. Therefore, the total number of electrochemical reports differs between Figures 3 and 5.

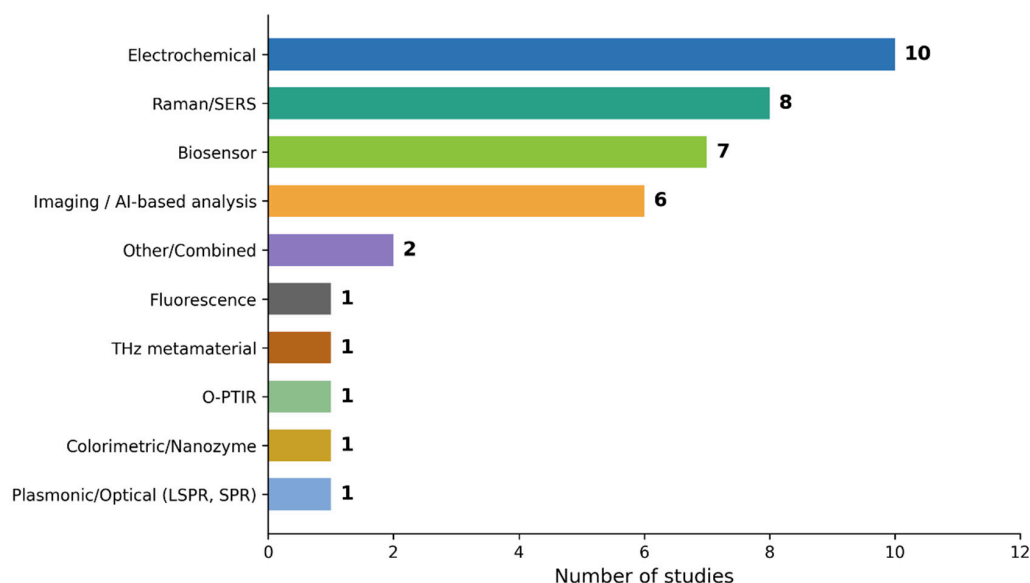


Figure 5. Detailed sensor type distribution among the 38 studies employing dedicated sensing platforms (mass spectrometry and classical bulk FTIR/uFTIR comparative studies excluded).

Polymer chemistry significantly influences sensor design, performance, and operational efficiency across several key aspects. Plastic particles encompass a wide range of chemical structures, such as PE, PET, PP, PS, PVC and PMMA, and their detection by biosensors requires a variety of specific bioreceptors (i.e., enzymes, antibodies, aptamers and peptides), which are selected or engineered to target specific polymer chemical structures at the molecular level. These bioreceptors interact with nano- and microplastics, providing specificity to sensors for different plastic types [61]. By distinguishing PE from PET or PP, biosensors prevent false positives or overestimation caused by non-plastic environmental matrices such as soil, algae or organic debris [62].

The high affinity of bioreceptors for direct interaction with specific functional groups within the plastic polymer structure enables the detection of ultra-low particle concentrations and smaller nanoplastics [63]. However, various factors may reduce binding affinity and thereby compromise biosensor specificity and selectivity. Pristine micro- and nanoplastics exhibit native recalcitrance and high hydrophobicity due to their polymer chemistry. Due to weathering, the chemical structure of the polymer may change through oxidation, chain scission, and molecular weight reduction [61]. Weathering naturally occurs because plastic particles are exposed to environmental factors such as solar UV radiation, moisture, microbial degradation. Consequently, new functional groups are formed on the polymer surface which becomes more hydrophilic. This shift changes the surface properties, affecting how the particles interact with sensor surfaces and bioreceptors. In addition to varying degrees of weathering oxidation, the same polymer can differ significantly in molecular structure because of substances intentionally or unintentionally added, such as pigments,

stabilisers, plasticisers, or flame retardants [62]. Therefore, it is difficult to design a single universal receptor even for the same type of micro- or nanoplastic. This diversity can be addressed by multi-recognition systems, in which sensors rely on multi-probe or multiplexed sensing platforms to detect diverse plastic types effectively in real-world scenarios [61]. Alternately, biosensors can detect plastic pollution by target-detecting leaching additives and adsorbed pollutants associated with specific polymer chemistries, such as bisphenol A and phthalates [64].

Electrochemical platforms form the largest group within the sensor subset (26.3%), followed by Raman/SERS-based approaches (21.1%) and biosensors (18.4%). AI/deep-learning image-classification studies are grouped here under imaging analysis rather than treated as a standalone sensor class; imaging-based and AI-assisted classification approaches together form a fourth category (six of 38 studies, 15.8%). This trend is consistent with the rapid expansion of machine-learning-assisted spectroscopic and image-based detection, as recently reviewed by Vidic et al. [65] (THz) metamaterial, optical photothermal infrared (O-PTIR) and fluorescence-based systems, were used in one or two studies. Their development indicates a long tail of technically distinctive approaches for micro- and nanoplastic quantification in water samples.

7.1. Electrochemical Sensors for Micro- and Nanoplastic Detection

In electrochemical sensors, the interaction of a target microplastic or nanoplastic particle with the electrode surface alters its electrical properties, such as current, impedance, or capacitance [65–67]. The simplest implementations rely on direct particle counting. Colson and Michel [68] developed a Coulter-type flow sensor in which individual PE and PS particles passing through a micro-orifice produce discrete impedance pulses, giving real-time, reagent-free particle counts. However, this sensor had limited chemical specificity and a size threshold close to 20 μm , which limits its applicability to real water samples.

More recent work has moved towards nanomaterial-modified electrode surfaces to push both sensitivity and size range downwards. Du et al. [69] showed that a graphene-modified electrode operating by electrochemical impedance spectroscopy could resolve mixed PE/PS microplastics across a wide concentration range with a single low-cost sensing layer. Wang et al. [70] went further by functionalising an electrode with a copper-based metal-organic framework combined with multiwalled carbon nanotubes. They achieved a label-free detection of polystyrene nanoplastics across three size classes (100 nm, 500 nm, and 1 μm) in real water samples. A related metal-labelling strategy reported by Zhao et al. [13] used electrochemical stripping voltammetry after tagging nanoplastics with a metal marker, enabling quantitative concentration analysis across several different water sample types.

The use of naturally sourced and biomimetic electrode materials represents another direction in the development of electrochemical sensors. Naik et al. [71] coated a carbon paste electrode with iron-doped polydopamine, designed to mimic the adhesive chemistry of mussel byssal proteins. This modified electrode detected PS nanoplastics with high selectivity, distinguishing PS from structurally similar polymers such as poly(ethylene glycol) (PEG) and poly(vinyl alcohol) (PVA) in spiked tap water. In another study in the dataset, biochar electrodes derived from starfish or aloe vera biomass enabled quantification of PS microparticles with detection limits of ~ 0.44 nM and ~ 0.52 nM, respectively [72]. Both electrodes were highly affordable but showed batch-to-batch variability that remains a practical obstacle to standardisation.

A persistent challenge in electrochemical sensor development is the trade-off between achieving high sensitivity and selectivity and validating performance under realistic water chemistry conditions. Interfering factors, such as electrode fouling by natural organic

matter, competing ion adsorption and the complex electrochemical background of environmental matrices, are rarely characterised alongside the lowest limits of detection. Nevertheless, electrochemical sensors have advantages over other sensor types because they can be readily miniaturised to a handheld format.

7.2. Optical Sensors for Micro- and Nanoplastic Detection

Optical sensing is the largest single detection-method category identified in this review (Figure 3). Optical methods exhibit the greatest methodological diversity among detection strategies, spanning surface-enhanced Raman scattering (SERS), surface plasmon resonance (SPR), fluorescence, colourimetry, and terahertz sensing. Rather than sharing a common instrumentation platform, these techniques are unified solely by their fundamental reliance on light–matter interactions.

SERS is the largest individual technique within this group (Figure 5) and remains the method most consistently able to achieve detection limits within the nanoplastic range. Ruan et al. [73] reported detection of PS nanoplastics down to 20 nm using a nanostructured SERS substrate. This particle size is among the smallest directly detected by any optical method in the dataset. Wang et al. [74] combined electrosorption pre-concentration with SERS to improve selectivity for trace-level nanoplastics, while Yang et al. [75] used a filtration-hyphenated silver nanowire membrane to achieve large-volume pre-concentration ahead of SERS detection. These reports show how sample preparation and spectroscopic readout are increasingly co-engineered rather than treated as separate steps, a trend also highlighted in a recent comprehensive review of SERS-based micro- and nanoplastic detection strategies [76].

A second major strand of optical methods in Figure 5 comprises plasmonic sensors which exploit shifts in localised or propagating surface plasmon resonance upon particle binding [77]. Wu et al. [78] demonstrated a dual-function plasmonic membrane that simultaneously filters low-micrometre microplastics from lake water and enables SERS-enhanced detection at the single-particle level, combining sample concentration and spectroscopic identification in one step. Wang et al. [79] reported a localised surface plasmon resonance optical microfibre sensor for detection of ultrasmall aromatic and hydrophobic nanoplastics at ultralow levels, extending plasmonic sensing towards fibre-optic, potentially field-deployable format.

Fluorescence-based approaches, often using dyes such as Nile Red or intrinsic polymer fluorescence, and colourimetric or nanozyme-based sensors occupy a smaller but practically important niche because they can, in principle, be read by eye or with a smartphone camera. Moreover, they can achieve the sensitivity of spectroscopic instrumentation while remaining field-portable [80–83]. Remarkably, Zhang et al. [82] developed a nano-arrayed Cu₂S@MoS₂ heterojunction substrate for highly sensitive, visual detection of polystyrene in environmental matrices, which provides both a visual colourimetric signal and a confirmatory SERS spectrum from the same measurement.

Analogous to trends in electrochemical sensing, optical platforms reporting the lowest detection limits, with several demonstrating capabilities below 100 nm diameter, rely almost exclusively on laboratory-spiked or synthetic water samples. Consequently, rigorous validation in chemically complex, real-world environmental matrices remains to be demonstrated.

A distinct and rapidly emerging strand of optical detection couples conventional image or spectral acquisition with deep-learning classification rather than relying on manual identification. Royer et al. [83] trained a convolutional neural network to segment and categorise microplastic debris directly from field photographs, reporting 96% classification accuracy and a substantial efficiency gain over manual counting, though the approach

depends on large, annotated training sets and provides no chemical confirmation of polymer identity. Devipriya et al. [84] compared several pretrained convolutional architectures, including MobileNetV3Large, ResNet50V2, ResNet101V2 and EfficientNetB7, combined with transfer learning for classifying microplastics into beads, fibres and fragments from digital microscope images, finding that transfer learning reduced the training data required but that irregular fragment shapes remained the most difficult class to resolve. Rather than replacing spectroscopic identification, Adi et al. [85] used a convolutional neural network to classify FTIR spectra while applying Shapley Additive exPlanations (SHAP) to reveal which spectral regions drove each classification decision, adding a layer of interpretability that addresses the black-box limitation of standard deep-learning classifiers at the cost of additional computational overhead. Collectively, these AI-assisted approaches illustrate a shift from artificial intelligence as a pattern-recognition add-on towards artificial intelligence as an interpretable, quantitatively validated component of the optical detection pipeline, though none has yet demonstrated field-deployable, real-time performance.

7.3. Biosensors for Micro- and Nanoplastic Detection

Biosensors are analytical tools in which a biological recognition element (bioreceptors) such as an enzyme, antibody, aptamer, or living microorganism, confers selectivity and sensitivity. Biosensors are less frequently reported than electrochemical or optical sensors (Figure 5) but are the fastest-growing analytical methods for micro- and nanoplastic detection. Six of the seven biosensor studies identified in this review (86%) were published in 2024 or later. This timing reflects a broader shift in the field from purely physical or chemical transduction towards biology-enabled recognition, aimed at improving selectivity for specific polymers or polymer-associated responses in complex matrices [86].

Enzyme-based recognition of PS, PMMA (poly(methyl methacrylate)) and PE were demonstrated using laccase immobilised on the surface of a plasmon resonance chip [87]. The biosensor was validated directly on real rainwater samples collected in Mexico. Although a wide detection range was achieved across the tested polymers, reduced performance was observed when metal ions were present in field samples. Aptamer-based recognition was used in two publications. Aptamers are artificial DNA or RNA oligomers that usually bind to their targets through their 3D structures [88]. A Systematic Evolution of Ligands by EXponential enrichment (SELEX)-derived DNA aptamer was shown to bind both PVC and PS despite their differing chemical structures, addressing a longstanding challenge in identifying plastic-binding recognition elements [89]. Shi et al. [60] combined the aptamer with a CRISPR/Cas12a signal-amplification cascade to construct a label-free electrochemical aptasensor capable of resolving PVC and PS at ng/mL concentrations. More expensive than aptamers, antibodies also provide a means for biological recognition of micro- and nanoplastics. In a single study within the dataset, the first polystyrene-specific polyclonal antibodies were raised and validated across in vitro and in vivo matrices [90], opening a path towards standard immunoassay formats. However, immunoassays such as enzyme-linked immunosorbent assay (ELISA) or lateral-flow strips have not yet been widely applied to microplastic detection despite their established role in other analyte classes.

An engineered environmental bacterium *Pseudomonas aeruginosa* expressing green fluorescent protein upon adhesion to plastic surfaces was used as a living biosensor for both biodegradable and conventional polymers in pretreated seawater [91]. Detectable fluorescence signals were observed within 3 h, with a detection limit of 1 ng/mL, showing sensitive comparable to or better than existing microplastic analytical methods. When coupled to Raman microspectroscopy the biosensor detected 100 µg/mL microplastics in real water samples.

The central promise of biosensors, selectivity derived from molecular recognition rather than bulk physical properties, comes with a correspondingly distinct set of limitations: reagent cost and stability of biological components under field conditions, and, for whole-cell biosensors, the viability of the living recognition element itself. These limitations remain less characterised in the literature than the analytical performance metrics that dominate the electrochemical and optical sensor literature reviewed above.

It is worth noting that micro- and nanoplastic aggregation can significantly impact biosensor performance, affecting operational mechanisms, quantification accuracy, and sensitivity [61]. When nanoplastics aggregate into larger clusters, their effective surface-area-to-volume ratio drops drastically, burying target functional groups within the aggregate and preventing biorecognition elements from binding efficiently. In digital, optical, or electrochemical single-entity sensing, an aggregate of dozens or hundreds of nanoplastics registers as a single count, leading to severe undercounting of the true particle number [92]. In fluorescent or surface-enhanced Raman scattering (SERS) biosensors, dense aggregation can trigger self-quenching or uneven local field enhancements, resulting in non-linear signal responses [93,94]. Moreover, micro- and nanoplastic particles can undergo heteroaggregation with inorganic colloids and natural organic matter in the environment [51,61]. In biosensors coupled with a microfluidic device, large aggregates can physically block microfluidic channels or passivate electrode surfaces, reducing repeatability and shortening sensor lifespan. Table 2 shows some principal advantages, limitations and detection-limit ranges of different detection methods from selected references in Supplementary Table S1. Reported values in the rightmost column are quoted as published by the source studies and use heterogeneous endpoints (mass concentration, particle concentration, molar concentration, particle size and, for AI-based methods, classification accuracy); they are not directly comparable across rows and are included as an indication of each category's typical operating range.

Table 2. Comparative summary of the sensor and classical analytical platforms identified in this review.

Platform	Detection Principle	Key Advantage(s)	Key Limitation(s)	Representative LOD/Range	Reference
Mass spectrometry (Py-GC/MS, TED-GC/MS, MALDI)	Pyrolysis/thermal desorption or laser desorption ionisation; mass-resolved fragment or ion analysis	Polymer-specific, quantitative; established reference method	Destructive; long run times; bulky instrumentation; not real-time	~0.02–0.44 µg (soil); ng–µg range typical	[30,50,52–56,95–99]
FTIR/µFTIR (classical)	Vibrational (IR) absorption spectroscopy; particle-level identification	Non-destructive; direct chemical identification	Slow acquisition; misses particles below ~50 µm; matrix interference	≥~1 µm (µRaman comparator resolves ~1 µm; µFTIR coarser)	[57,59,100]
Electrochemical	Impedance/current/capacitance change at the electrode–particle interface	Low-cost; miniaturisable; real-time; handheld-compatible	Electrode fouling by natural organic matter; limited intrinsic polymer specificity; batch-to-batch variability of biomimetic electrodes	~0.4–0.5 nM (biochar electrodes); sub-100 nM for functionalised electrodes	[13,64,68–72,101–103]
Raman/SERS	Surface-enhanced Raman scattering from particles adsorbed on nanostructured substrates	Molecular fingerprint; best nanoplastic-range sensitivity	Substrate reproducibility; matrix interference; costly substrates	Down to 20 nm (best case)	[73–75,78,82,104–106]

Table 2. Cont.

Platform	Detection Principle	Key Advantage(s)	Key Limitation(s)	Representative LOD/Range	Reference
Plasmonic/Optical (LSPR, SPR)	Shift in localised or propagating surface plasmon resonance upon particle binding	Label-free; fibre-optic, field-deployable formats possible	Sensitivity to non-specific binding; complex fabrication	Ultralow (ultrasmall aromatic/hydrophobic nanoplastics)	[79]
Fluorescence	Dye-based (e.g., Nile Red) or intrinsic polymer fluorescence	Simple readout; smartphone-camera compatible	Limited intrinsic polymer specificity; photobleaching	Not systematically reported	[107]
Colourimetric/Nanozyme	Catalytic or visual colour-change reaction	Naked-eye/field readout; dual-mode (visual + SERS) demonstrated	Narrower dynamic range; cross-reactivity	Not systematically reported	[108]
THz metamaterial	Terahertz resonance shift	Label-free; potential for rapid bulk screening	Early-stage; limited validation	Not systematically reported	[109]
O-PTIR	Combined optical photothermal heating with IR/Raman readout	Sub-micron chemical imaging	Instrumentation complexity and cost	Sub-micron	[110]
Imaging/AI-based analysis (hyperspectral + deep learning)	Spatially resolved spectral imaging processed with chemometrics or CNN/deep-learning classifiers	High throughput; increasingly interpretable (e.g., SHAP)	Requires large annotated datasets; no independent chemical confirmation; not yet field-deployable	Reported classification accuracy up to 96%	[58,83–85,111,112]
Biosensor (enzyme, antibody, aptamer, whole-cell)	Biological recognition element coupled to an optical, electrochemical or fluorescence transducer	High selectivity via molecular recognition; some field-portable formats	Reagent cost/stability; viability of whole-cell elements; fewer field validations	~1 ng/mL (whole-cell); ng/mL (aptamer/CRISPR)	[60,87,89–91,113,114]
Other/combined	Mixed/hybrid transduction	Combines strengths of more than one platform	Added system complexity	Not systematically reported	[115,116]

Note: The Electrochemical row ($n = 10$) follows the recognition-mechanism classification used in Figure 5, which places the one aptamer/CRISPR-Cas12a study with electrochemical transduction under Biosensor.

8. Matrix, Water Source, and Particle Size Trends in the Reviewed Literature

Beyond the specific detection principles, we analysed the choice of matrix, water source, and target particle size across the compiled literature to elucidate the current focus and corresponding gaps in method validation in the field. While this review centres on water contamination, non-water matrices (soil, sediment, air, and biological tissue) are included as key transport pathways and comparators. Evaluating these matrices quantifies the proportion of detection literature explicitly dedicated to water versus broader environmental compartments, contextualising the current analytical focus without compromising the review's primary aquatic lens.

Water dominates as the tested matrix by a wide margin, accounting for 30 of 53 studies (56.6%), more than three times the combined total of all solid environmental matrices, soil, sediment and food/packaging together (eight studies, 15.1%), as shown in Figure 6.

Seven studies (13.2%) reported only a generic or unspecified matrix, four (7.5%) used purely laboratory or model samples with no stated environmental origin, and biological tissue and air together account for just four studies (7.5%). Given that atmospheric deposition, groundwater recharge and stormwater pulses are documented contamination routes, the near absence of air, soil and groundwater from the detection literature indicates a mismatch between where contamination has been shown to occur and where detection methods are actually being tested. This gap is also highlighted by dedicated methodological work on soil microplastic quantification [117] and on airborne micro- and nanoplastic sampling

and analysis [118], neither of which is represented among the sensor platforms reviewed. Figure 7 shows that within the water-matrix subset specifically, the two largest categories are studies reporting only generic or unspecified water (11 studies, 36.7%) and studies conducted on laboratory-spiked or synthetic water (10 studies, 33.3%). Together these account for 70% of all water-based studies.

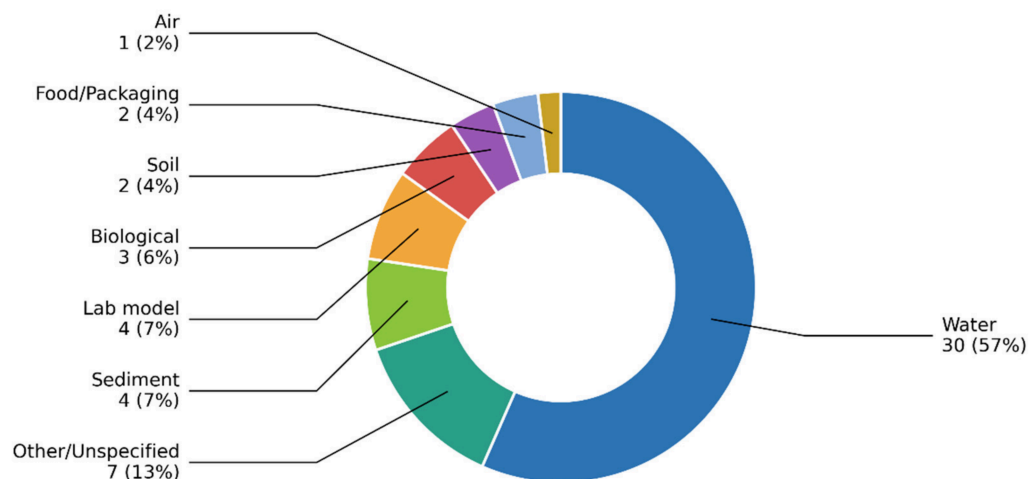


Figure 6. Distribution of the 53 reviewed studies by sample matrix. The whole-number percentages are rounded using the largest-remainder method and therefore sum exactly to 100%.

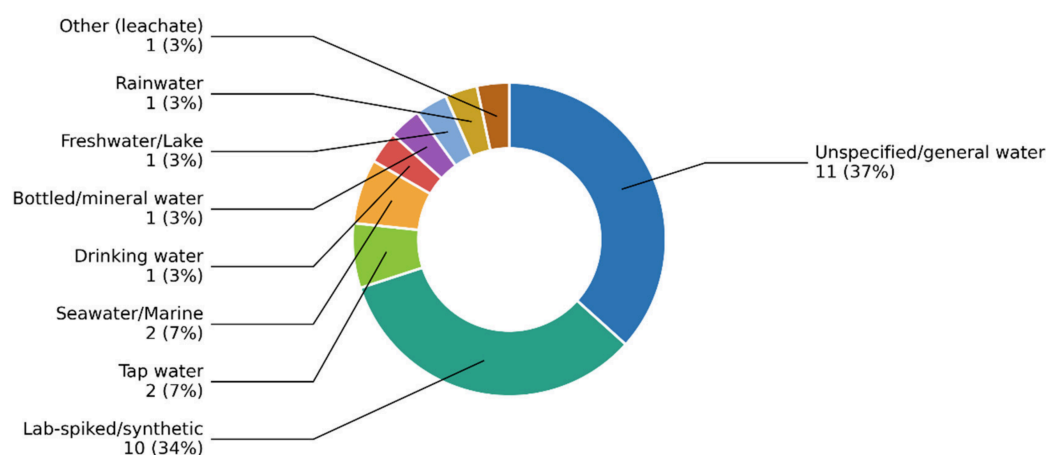


Figure 7. Water source among the 30 studies conducted in a water matrix. The whole-number percentages are rounded using the largest-remainder method and therefore sum exactly to 100%.

Named real-world water types are comparatively rare: tap water and seawater each appear in only two studies, and drinking water, bottled or mineral water, freshwater or lake water, and rainwater each in a single study. Reported sensitivity, selectivity and linear range figures therefore derive predominantly from conditions that do not reproduce the ionic strength, organic loading and particle heterogeneity of real environmental or drinking water. This reinforces the need for harmonised field protocols and reference materials that would allow laboratory and field performance to be compared directly [19,20]. This concern was raised by Koelmans et al. [119] in their critical assessment of data quality across fifty freshwater and drinking-water microplastic studies, where they similarly found methodological inconsistency to be a major barrier to reliable comparison. Moreover, a previous systematic review of drinking-water microplastic contamination reached similar conclusions about the state of the underlying evidence base [120].

Biofouling poses a major challenge to the sensitivity and reliability of biosensors used for micro- and nanoplastic detection in environmental matrices. The process begins with

the rapid formation of a bio-organic film over the biosensor surface, composed of different proteins, lipids and natural organic matter, and progresses to microfouling as bacteria and algae colonise the surface and to macrofouling as a compact biofilm develops. This reduces access to target binding sites, directly lowering analytical sensitivity and elevating the lower limit of detection [121,122]. Biofilm accumulation creates a dense physical barrier over transducer surfaces, severely restricting mass transport and slowing the diffusion of micro- and nanoplastic particles towards the active sensing interface, leading to delayed response times and attenuated signal magnitudes [31].

Finally, the target particle size class shows a comparable pattern of technical progress outpacing validation. Microplastics remain the single largest target category (28 studies, 52.8%), but nanoplastics accounts for a substantial share of the literature (16 studies, 30.2%), and nine further studies (17.0%) addressed both size classes within the same platform (Figure 8).

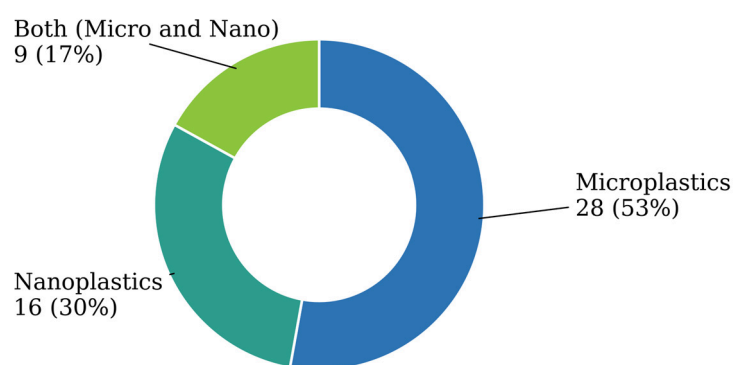


Figure 8. Distribution of studies by target particle size class (Microplastics, Nanoplastics, Both).

Combined, nanoplastic-capable methods now account for just under half of the dataset (25 of 53 studies, 47.2%), a high share given how recently reliable sub-micron detection became technically feasible. This shift towards the nanoscale is driven disproportionately by electrochemical and Raman/SERS platforms, several of which report detection limits below 100 nm, rather than by traditional mass spectrometric methods, which remain concentrated at the microplastic end of the size range. These findings have direct regulatory implications. Nanoplastics represent the size fraction least understood by regulatory bodies, yet they are also where conventional spectroscopy and mass spectrometry encounter their most severe detection limits. Consequently, advancing sensor development within this sub-micron regime represents the area of greatest environmental and analytical significance.

Two structural gaps connect these matrix and water-source patterns back to contamination pathways. None of the 53 studies reviewed were designed around the high-frequency, near-continuous sampling that would be needed to resolve a storm-driven contamination pulse as documented in a prior study [42]. However, several of the electrochemical and optical platforms reviewed here are, in principle, fast enough to do so. This capability highlights the primary advantage of portable sensors over benchtop reference methods: beyond reducing per-sample costs, they enable high-frequency temporal monitoring capable of capturing transient events that discrete, scheduled grab sampling inherently misses. Groundwater, which supplies roughly two-thirds of EU drinking water, is essentially absent from the matrix results in Figure 6, appearing at best within the small “Other/Unspecified” or “Lab model” categories. None of the studies reviewed here explicitly targeted groundwater or well-water sampling. Because groundwater contamination can persist undetected for decades once it occurs, this is a second, largely independent monitoring blind spot, and closing it would likely require sensor formats deployable directly at wellheads and

abstraction points rather than the single-sample, benchtop formats that dominate the current literature.

9. Water Regulatory Frameworks

Current regulatory frameworks treat microplastics cautiously. The EU Drinking Water Directive (EU) 2020/2184 and the WHO Guidelines for Drinking-water Quality set parametric values for established contaminants, and WHO has flagged PFAS and disinfection by-products as priorities for its next revision [123–125]. Within the EU’s broader water-protection architecture, Directive (EU) 2026/805 recently expanded the binding list of priority substances that Member States must monitor in surface water and groundwater but placed microplastics and antimicrobial-resistance indicators on a parallel watch list rather than the binding list itself, explicitly pending harmonised, EU-wide testing methods [126,127]. This is the key point for the present review: the absence of a validated, standardised detection method, the same gap that the classical and sensor-based platforms discussed above are trying to close, is the formal bottleneck preventing microplastics from reaching binding regulatory status, not a lack of political will.

A parallel pattern holds outside the EU. The U.S. EPA added microplastics to its draft Sixth Contaminant Candidate List in 2026 but excluded them from the accompanying monitoring rule, again citing the lack of a validated, consensus test method [128]. Non-binding EU water-policy instruments, such as the 2025 Water Resilience Strategy, likewise leave microplastics without harmonised drinking-water limits, and none of the frameworks reviewed here impose monitoring obligations for atmospheric or event-driven contamination pulses [129]. This measurement gap is compounded by the difficulty of monitoring groundwater, which supplies roughly two-thirds of EU drinking water yet remains structurally harder to sample than surface water, with about a quarter of EU groundwater bodies already in poor chemical status [130]. Table 3 summarizes the key instruments discussed here and their current treatment of micro- and nanoplastics.

Table 3. Key regulatory instruments and their treatment of micro- and nanoplastics.

Instrument	Level	Status (As of Mid-2026)	Relevance to Micro-/Nanoplastics
Drinking Water Directive (EU) 2020/2184	EU, binding	In force since 12 January 2021	Sets parametric values for drinking water; does not yet set a binding microplastics limit
WHO Guidelines for Drinking-water Quality	Global, non-binding	4th edition + 3rd addendum, June 2026; 5th edition in preparation	Reference values for national regulators; PFAS and disinfection by-products (DBPs) flagged for the next edition
EU Zero Pollution Action Plan	EU, policy targets	Adopted 12 May 2021	2030 target of 30% cut in microplastics released to the environment
Directive (EU) 2026/805 (amending WFD, GWD, EQSD)	EU, binding	In force since 11 May 2026	Adds PFAS, pharmaceuticals and bisphenols to the binding priority-substance list; adds microplastics and AMR indicators to a watch list pending harmonised test methods
EU Water Resilience Strategy	EU, non-binding	Adopted 4 June 2025	Sets a 10% water-efficiency target by 2030; frames water scarcity alongside pollution

10. Conclusions

This review compiled and classified 53 original research articles published between 2021 and mid-2026, identified through a PRISMA-guided search of Google Scholar and ScienceDirect, to compare classical analytical methods with advanced electrochemical, opti-

cal and biosensor platforms for the detection of micro- and nanoplastics in water. Optical techniques remain the dominant detection method overall; mass spectrometry continues to serve as the reference approach for polymer-specific quantification. Electrochemical and biosensor platforms represent the fastest-growing segments of the literature. Water remains the overwhelmingly dominant test matrix, but a majority of water-based studies rely on laboratory-spiked or otherwise unspecified samples rather than named real environmental or drinking water, and groundwater and atmospheric or event-driven contamination are essentially absent from the detection literature despite documented evidence that both are meaningful contamination pathways. Unlike thermal mass spectrometry, such as Py-GC/MS, standard visual/imaging-based optical and deep-learning approaches focus heavily on particle counts, shapes, or surface interactions without directly yielding reliable, mass-based quantitative composition. Despite their popularity, most optical platforms, including these based on SERS and SPR, have lower sensitivity and selectivity complex real-world environmental matrices due to interference from natural organic matter, high turbidity, or ionic background. Fluorescence background from organic co-contaminants or biofouling can obscure optical signals. Furthermore, environmental weathering causes the optical and spectral signatures of polymers to drift away from pristine reference libraries over time, increasing misclassification risks.

Nanoplastic-capable methods now account for nearly half of the dataset, a rapid technical advance that has not yet been matched by regulatory uptake: current EU legislation places microplastics and antimicrobial resistance indicators on a watch list rather than a binding substances list, explicitly pending the harmonised testing methods that the literature has not yet converged on.

Several limitations of the review itself should be acknowledged. The search was restricted to two databases, Google Scholar and ScienceDirect, and to English-language original research articles, which may have excluded relevant work indexed elsewhere or published in other languages. The classification scheme used throughout, particularly the collapsing of thirteen sensor categories into four broader detection-method groups and the assignment of hybrid platforms such as AI-assisted biosensors to a single primary category, involved judgement calls that a different reviewer might have made differently, though the underlying study-level data are provided in full in Table S1 (Supplementary Materials) so alternative groupings can be reconstructed. Reported limits of detection also use an inconsistent mix of units, mass concentration, particle concentration, molar concentration, across the source literature, so a direct, unit-harmonised comparison of sensitivity across techniques was not attempted here and remains an open task once the field converges on more standardised reporting practices. Closing the gap between technical capability and both regulatory relevance and field-ready deployment, particularly towards matrix-realistic validation, higher-frequency monitoring and wellhead-deployable formats, represents the clearest direction for future work in this field.

Supplementary Materials: The following supporting information can be downloaded at: <https://www.mdpi.com/article/10.3390/chemosensors14090198/s1>, Table S1: Full study-level dataset for all 53 studies included in this review, including all extracted fields (Year, Authors, Detection technique/Sensor, Detection principle, Target polymer(s), Matrix, LOD, Advantages, Limitations, Reference DOI) and all derived classification fields (Sensor Category, Detection Method, Matrix Category, Water Source, Micro- vs Nanoplastics); Table S2: Completed PRISMA 2020 checklist for this review, reporting the manuscript section or figure in which each checklist item is addressed. Items relating to statistical synthesis and formal risk-of-bias assessment are marked as not applicable as this review performed a descriptive and narrative classification of the literature rather than a meta-analysis.

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Conflicts of Interest: The authors declare no conflicts of interest.

Abbreviations

MP/NP	Microplastic(s)/Nanoplastic(s)
LOD	Limit of Detection
PAH	Polycyclic Aromatic Hydrocarbon
PAE	Phthalate Acid Ester
BTEX	Benzene, Toluene, Ethylbenzene, Xylene
PFAS	Per- and Polyfluoroalkyl Substances
ARG/AMR	Antibiotic Resistance Gene/Antimicrobial Resistance
WFD/GWD/EQSD	Water Framework Directive/Groundwater Directive/Environmental Quality Standards Directive
DWD	Drinking Water Directive
WHO	World Health Organization
EU/EEA	European Union/European Environment Agency
PRISMA	Preferred Reporting Items for Systematic Reviews and Meta-Analyses
SERS	Surface-Enhanced Raman Scattering
LSPR/SPR	Localised Surface Plasmon Resonance/Surface Plasmon Resonance
EIS	Electrochemical Impedance Spectroscopy
FTIR	Fourier-Transform Infrared Spectroscopy
Py-GC/MS	Pyrolysis Gas Chromatography–Mass Spectrometry
TED-GC/MS	Thermal Extraction Desorption Gas Chromatography–Mass Spectrometry
ICP-MS	Inductively Coupled Plasma Mass Spectrometry
GC-MS/LC-MS/MS	Gas Chromatography–Mass Spectrometry/Liquid Chromatography–Tandem Mass Spectrometry
AI	Artificial Intelligence
CRISPR	Clustered Regularly Interspaced Short Palindromic Repeats
PMT/vPvM	Persistent, Mobile and Toxic/very Persistent and very Mobile
PE, PP, PS, PVC, PET, PA, PMMA, and PC	Polyethylene, Polypropylene, Polystyrene, Polyvinyl Chloride, Polyethylene Terephthalate, Polyamide, Poly(methyl methacrylate), Polycarbonate, respectively

BPA	Bisphenol A
UV	Ultraviolet
EPS	Expanded Polystyrene
TED	Thermal Extraction Desorption
TMAH	Tetramethylammonium Hydroxide
MALDI	Matrix-Assisted Laser Desorption/Ionisation
O-PTIR	Optical Photothermal Infrared
THz	Terahertz
PEG	Poly(ethylene glycol)
PVA	Poly(vinyl alcohol)
SELEX	Systematic Evolution of Ligands by EXponential Enrichment
ELISA	Enzyme-Linked Immunosorbent Assay
DBP	Disinfection By-Product
DNA	Deoxyribonucleic Acid
RNA	Ribonucleic Acid

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