

Descriptive coastal fish assemblage contrasts and a pilot sentinel monitoring framework for Al Hoceima National Park

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Abstract. Mediterranean marine protected areas require repeatable fish monitoring to evaluate conservation and fisheries outcomes, yet many management programmes begin with sparse, heterogeneous observations. This study evaluates a pilot underwater visual census undertaken on 19 August 2026 in Al Hoceima National Park, Morocco, places it within surveys conducted in 2002 and 2019, and uses the combined evidence to design an operational sentinel-monitoring framework. Two contrasting coastal units - one recorded as natural habitat and one as an artificial breakwater - were surveyed in 2026. Fish counts were consolidated by taxon and field size class, taxonomic names were quality-controlled, and assemblages were described using abundance, richness, Shannon diversity, Pielou evenness, Simpson dominance and Bray-Curtis dissimilarity. The 2002 MedMPA survey recorded 72 species across multiple sites and depth bands, whereas 22 visual surveys in 2019 recorded 3,577 individuals and 16 families; reported richness varies between 42 and 44 species within the source report. Those historical datasets identify recurrent sparid and labrid components and high proportions of small fish at some shallow locations, but their different coverage and effort preclude a direct numerical trend with 2026. Because the 2026 transect geometry, depth and spatial replication were not documented, analyses were intentionally restricted to raw-count descriptions and no habitat effect was tested. The pilot recorded 290 individuals representing nine taxonomic units. The natural unit contained 199 individuals and seven taxa, whereas the breakwater contained 91 individuals and five taxa. Four schooling or abundant groups - *Boops boops*, *Sarpa salpa*, *Oblada melanurus* and *Mugil* sp. - accounted for 90.0% of all records. Bray-Curtis dissimilarity was 0.745, indicating a marked descriptive contrast that cannot be attributed causally to habitat. The pilot supplies a traceable summer baseline, while the historical surveys guide station placement and ecological questions. Fixed georeferenced transects, replicated habitat units, standardized effort, environmental covariates, photographic validation and seasonally homologous sampling are required before interannual change can be estimated.

Keywords: Alboran Sea, fish assemblage, marine protected area, sentinel monitoring, underwater visual census.

Introduction. Marine protected areas (MPAs) are expected to conserve biodiversity, rebuild exploited populations and sustain ecosystem services that support coastal fisheries. These outcomes are not automatic: they depend on protection level, enforcement, habitat condition, time since establishment and the capacity of managers to detect ecological change and respond to it (Edgar et al., 2014; Di Franco et al., 2016; Zupan et al., 2018). Fish assemblages are therefore central monitoring targets because their abundance, size structure and composition respond to fishing pressure, habitat complexity and environmental variability, while also connecting conservation objectives with fisheries benefits.

Underwater visual census (UVC) is widely used for shallow coastal fish monitoring because it is non-destructive, relatively inexpensive and capable of recording taxon identity, abundance and size during a single survey. However, detectability varies with fish behaviour, body size, habitat structure, observer experience, visibility and transect dimensions. Inadequately documented effort can consequently produce apparent

differences that cannot be separated from observation bias (Edgar et al., 2004; Williams et al., 2006; Prato et al., 2017). Multiple or nested sampling areas, video validation and explicit recording of environmental covariates can improve the representation of different behavioural and size groups (Stobart et al., 2007; Tessier et al., 2013; Grane-Feliu et al., 2019).

Habitat contrasts are particularly relevant in coastal MPAs, where rocky or vegetated natural habitats coexist with ports, seawalls and breakwaters. Artificial structures may provide shelter and feeding surfaces, but their assemblages cannot be assumed to replicate the ecological functions of natural habitat. Previous studies have shown that protection status, habitat configuration and survey method all shape recorded fish abundance and diversity (Grimmel et al., 2020; Horta e Costa et al., 2020; Kiggins et al., 2020). Consequently, comparisons must control for habitat, depth, effort and time before they are interpreted as ecological effects.

The marine component of Al Hoceima National Park (PNAH), on Morocco's southern Alboran coast, contains a mosaic of rocky reefs, macroalgal assemblages, *Cymodocea nodosa* meadows, coralligenous formations, rhodolith beds and artificial coastal structures. These habitats support ecological functions relevant to fish recruitment and artisanal fisheries. The 2002 MedMPA and 2019 MedKeyHabitats surveys provide valuable regional references, including repeated evidence of small fishes in shallow locations. However, their sampling coverage and effort differ from the 2026 pilot, so the three campaigns do not constitute a continuous standardized time series (UNEP/MAP-SPA/RAC, 2020). The doctoral observatory framework developed for PNAH identifies standardized ecological observation, environmental context, quality control, indicators and feedback to management as a single decision chain.

This short communication has three objectives: (1) to provide an auditable descriptive analysis of a pilot census comparing a natural coastal unit and a breakwater on 19 August 2026; (2) to use the 2002 and 2019 datasets as historical ecological context while explicitly evaluating comparability; and (3) to translate the limitations revealed by all three sources into a replicable sentinel-monitoring design for PNAH and comparable data-limited Mediterranean MPAs. The study tests neither a causal habitat hypothesis nor an interannual trend.

Material and Method

Study context. Al Hoceima National Park is located on the southern margin of the Alboran Sea in northern Morocco. The protected area comprises approximately 48,460 ha, including about 19,600 ha of marine waters. Its steep rocky coastline, shallow infralittoral habitats, coves and artificial structures support both conservation values and small-scale fishing activity. The park's management context is typical of many southern Mediterranean MPAs: valuable baseline surveys exist, but sampling programmes have often been project-based and use different spatial units, depths and effort definitions. The area figures follow the current park management plan (Parc National d'Al Hoceima, 2023). The complete PNAH boundary, principal zoning elements and the location supplied for the 2026 pilot sampling habitat are shown in Figure 1A. The pilot sampling location is 35.235404° N, 3.986080° W.

The two units in the 2026 pilot were recorded in the field database as "natural habitat" and "breakwater". The source file did not include coordinates, depth or a formal substrate classification. The natural unit is therefore not assigned retrospectively to seagrass, reef, coralligenous or another specific habitat class. This preserves the distinction between documented observations and ecological interpretation. Operationally, "natural habitat" denotes the non-engineered coastal unit recorded by the observers, whereas "breakwater" denotes the artificial coastal structure sampled during the same campaign. A field photograph of the sampling environment is provided in Figure 1B; it documents the coastal setting and breakwater but is not used to assign a finer substrate class because depth and quantitative substrate composition were not recorded.

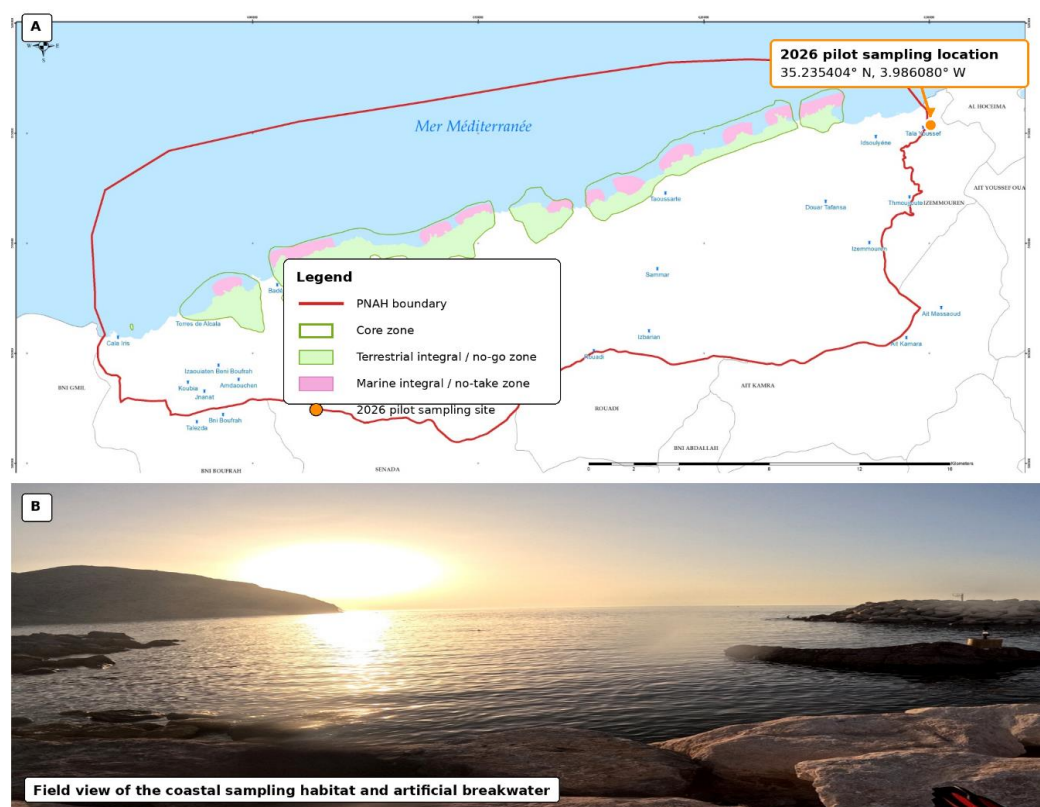


Figure 1. Location and field context of the 2026 pilot fish census in Al Hoceima National Park, Morocco. (A) PNAH cartographic zoning map showing the complete park boundary and the pilot sampling location (35.235404° N, 3.986080° W) at the eastern coastal limit. (B) Field view of the coastal sampling environment and artificial breakwater. The zoning map is adapted from the PNAH cartographic atlas supplied by the park administration.

Pilot survey and data provenance. A non-destructive visual fish census was conducted on 19 August 2026. The two habitat units were visited during the same campaign. For each entry, the field spreadsheet recorded taxon, number of individuals and one of three metric size classes (70-80 mm, 90-100 mm or > 100 mm). Environmental conditions were recorded contemporaneously and are reported in the Results section. The verified pilot design and analytical boundaries are summarized in Table 1.

Table 1

Verified pilot design and analytical boundary

Element	Pilot record	Analytical consequence
Date and season	19 August 2026; summer	Observed
Comparison units	Natural habitat; artificial breakwater	Observed; substrate details incomplete
Method	Underwater visual census	Documented; effort geometry missing
Fish variables	Taxon, count, field size class	Observed
Environmental variables	Air/water temperature, visibility, weather, wind, swell	Observed qualitatively except temperature
Coordinates, depth and time	Sampling location supplied subsequently: 35.235404° N, 3.986080° W; depth and survey time not available in the primary field record	Map the site context; do not reconstruct depth, time or survey effort
Transect area and replication	Not available	No density or inferential test

Note: "Not available" means undocumented, not zero.

Taxonomic and data-quality control. Thirteen size-class rows were consolidated into nine taxonomic units. Fish names and accepted nomenclature were checked against FishBase (Froese & Pauly, 2026). The field name *Oblada melanura* was retained in the provenance field but analysed under the accepted FishBase name *Oblada melanurus*. Two records remained above species level because the source material did not support a more precise identification: Blenniidae at family level and *Mugil* sp. at genus level. Two *Diplodus sargus* entries from the breakwater represented different size classes and were summed only for taxon-level composition.

Recorded schools of 100 *Boops boops* and 56 *O. melanurus* were retained exactly as entered. No record was removed solely because its count was high. Missing seasonal campaigns were coded as unavailable and were neither converted to zero nor interpolated.

Historical reference datasets and comparability. Historical fish observations were extracted from the PNAH MedKeyHabitats II technical report, which summarizes the 2002 MedMPA survey and reports the 2019 survey (UNEP/MAP-SPA/RAC, 2020). In 2002, visual counts covered four coastal sectors, 16 sites and four depth intervals (0-3, 4-7, 12-16 and 24-30 m). Each census followed a 15-minute path; abundance was assigned to eight numerical classes and fish to three relative size classes. The survey reported 72 species. In 2019, nine georeferenced transects were chosen partly to revisit the 2002 spatial framework. Twenty-two 15-minute visual surveys were completed across three depth bands (0-3, 4-7 and 12-16 m), recording 3,577 individuals and 16 families; reported richness varies between 42 and 44 species within the source report. The resulting comparability assessment and permitted uses of each dataset are summarized in Table 2.

Comparability was assessed before synthesis using five criteria: spatial overlap, depth coverage, survey duration or area, abundance measurement, and taxonomic/size resolution. The 2002 and 2019 datasets share a regional framework and 15-minute census duration, although effort and site coverage still differed. The 2026 record lacks location, depth, duration and area, and comprises only two unreplicated census units. Therefore, historical counts were not pooled with the 2026 data, and no percentage change, temporal model or stock trend was calculated. Historical information was used only to identify recurrent taxa, broad assemblage features, spatial hypotheses and candidate nursery/recruitment areas for future standardized testing.

Table 2

Historical evidence and comparability. Different effort and spatial coverage prevent a direct 2002-2019-2026 abundance trend

<i>Dataset</i>	<i>Documented design</i>	<i>Main information</i>	<i>Permitted use in this study</i>
2002 MedMPA	Four sectors; 16 sites; 0-30 m; 15-min paths; abundance classes	72 species; sparids/labrids prominent; > 50% small fish at three reference sites	Regional baseline and station-selection evidence; not numerically combined with 2026
2019 MedKeyHabitats II	Nine transects; 22 surveys; 0-16 m; 15-min paths	3,577 individuals; 42-44 species reported; 16 families; > 80% small-size fraction at shallow Cala Iris sites	Context and recruitment hypothesis; not a direct trend with 2026
2026 pilot	Two unreplicated units; depth, location, duration and area undocumented	290 individuals; nine taxonomic units; natural-breakwater descriptive contrast	Traceable pilot baseline and protocol stress test; no MPA-wide extrapolation

Descriptive analysis. For each unit, total abundance (N), taxonomic richness (S), relative abundance and maximum taxon dominance were calculated from raw counts. Shannon diversity was calculated as $H' = -\sum p_i \ln(p_i)$, Pielou evenness as $J' = H'/\ln(S)$, and Simpson

dominance as $\lambda = \sum p_i^2$, where p_i is the proportion of individuals assigned to taxon i . Bray–Curtis dissimilarity between the two census units was calculated as $BC = \sum |x_i - y_i| / \sum (x_i + y_i)$. Here, S is the number of taxonomic units recorded in a census unit, while x_i and y_i are the counts of taxon i in the natural and breakwater units, respectively. Diversity-index calculations followed Magurran (2004).

All metrics are exploratory. With one unit per habitat, no within-habitat variance can be estimated; confidence intervals, hypothesis tests and habitat-effect estimates would be pseudoreplicated. Size-class results are reported in the original metric bands and are not translated into juvenile/adult status because maturation thresholds are species-specific. No extrapolation is made from census counts to fish stocks across the MPA.

Derivation of the sentinel framework. The monitoring framework was derived by matching each limitation of the pilot dataset to a design requirement and a management use. This follows adaptive-monitoring principles: monitoring questions are linked to decisions; variables and effort are standardized; uncertainty is explicit; and the design can be revised as knowledge accumulates (Holling, 1978; Walters, 1986; Lindenmayer & Likens, 2009). The PNAH thesis framework supplied the operational sequence linking ecological, oceanographic and use-pressure observations to quality control, indicators, diagnosis and management feedback. No thesis-derived value was treated as an additional fish observation.

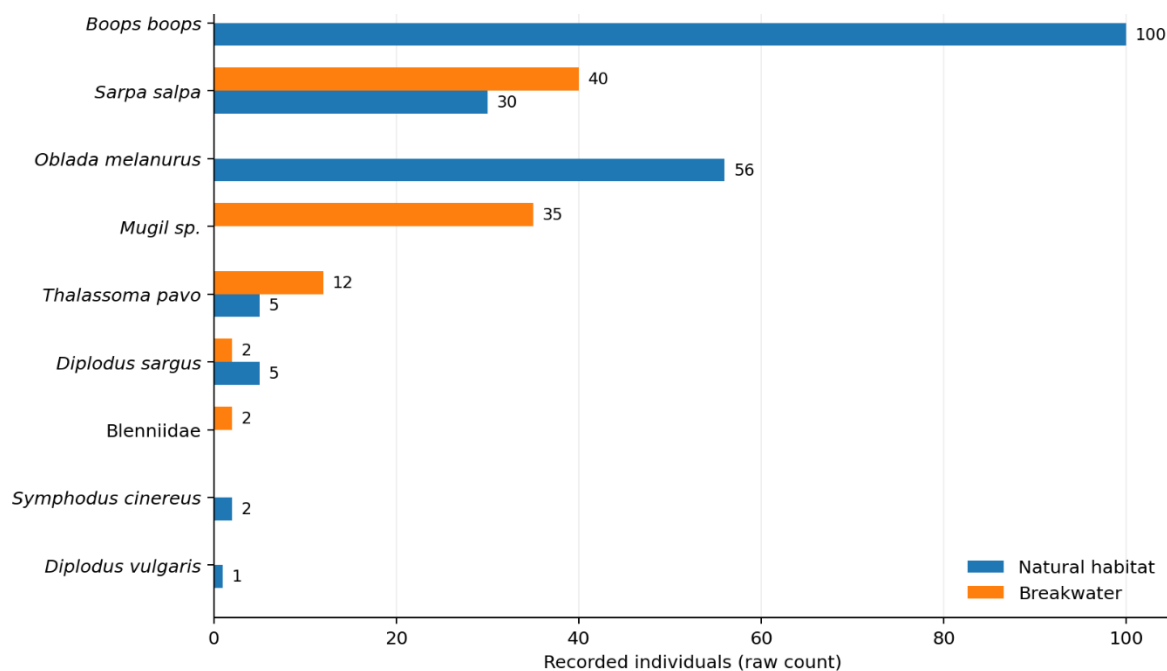
Results

Field conditions. During the 19 August 2026 survey, air temperature was 27°C and water temperature was 23°C; visibility was poor, skies were overcast, no rainfall was recorded, wind was weak from the north-east, and swell was weak from the south-west.

Composition and abundance. The pilot campaign recorded 290 individuals distributed among nine taxonomic units: seven identified to species, one to genus and one to family. The natural census unit contained 199 individuals and seven taxa; the breakwater contained 91 individuals and five taxa. The raw abundance ratio was therefore 2.19:1 and richness differed by two taxa. These values describe the sampled units on one date and are not standardized habitat estimates. Taxon-level counts are summarized in Table 3 and illustrated in Figure 2.

Table 3
Raw taxon counts recorded on 19 August 2026 (effort area was not documented)

<i>Taxonomic unit</i>	<i>Natural</i>	<i>Breakwater</i>	<i>Total</i>	<i>Share</i>
<i>Boops boops</i>	100	0	100	34.5%
<i>Sarpa salpa</i>	30	40	70	24.1%
<i>Oblada melanurus</i>	56	0	56	19.3%
<i>Mugil</i> sp.	0	35	35	12.1%
<i>Thalassoma pavo</i>	5	12	17	5.9%
<i>Diplodus sargus</i>	5	2	7	2.4%
<i>Symphodus cinereus</i>	2	0	2	0.7%
Blenniidae	0	2	2	0.7%
<i>Diplodus vulgaris</i>	1	0	1	0.3%



One survey date (19 August 2026); effort geometry and replication were not documented.

Figure 2. Recorded taxon abundance in the natural and breakwater census units. Values are raw counts from one campaign without spatial replication.

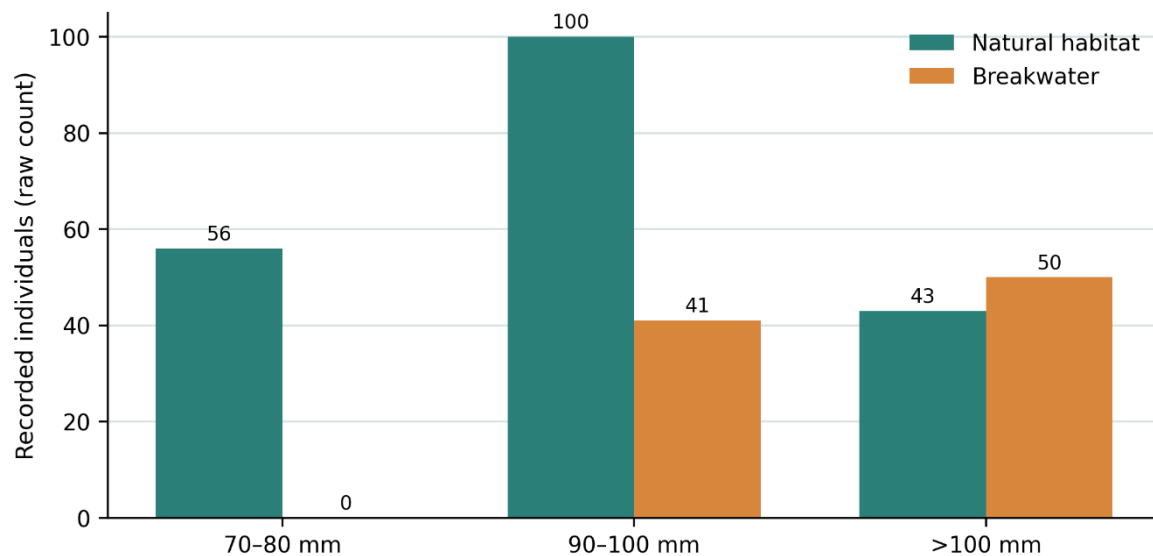
Dominance and assemblage contrast. Four units accounted for 90.0% of all recorded individuals: *Boops boops* (100; 34.5%), *Sarpa salpa* (70; 24.1%), *Oblada melanurus* (56; 19.3%) and *Mugil sp.* (35; 12.1%). *B. boops* represented 50.3% of individuals in the natural unit. At the breakwater, *S. salpa* represented 44.0% and *Mugil sp.* 38.5%. The dominance of schooling taxa strongly influenced abundance-based contrasts.

Bray-Curtis dissimilarity between the two units was 0.745 (similarity = 0.255). The contrast was driven largely by *B. boops* and *O. melanurus*, recorded only in the natural unit, and *Mugil sp.*, recorded only at the breakwater. Because location, depth, effort and replication were uncontrolled, this dissimilarity is a description of the two censuses rather than a general habitat effect. Exploratory diversity descriptors are summarized in Table 4.

Table 4
Exploratory diversity descriptors calculated from raw counts; no inferential comparison is supported

Unit	N	S	H'	J'	λ	$1-\lambda$	Max. dominance
Natural habitat	199	7	1.246	0.640	0.356	0.644	0.503
Breakwater	91	5	1.164	0.723	0.359	0.641	0.440
Combined	290	9	1.627	0.741	0.233	0.767	0.345

Recorded size structure. In the natural unit, 56 individuals were assigned to 70-80 mm, 100 to 90-100 mm and 43 to > 100 mm. At the breakwater, no fish were entered in the 70-80 mm band, 41 were in 90-100 mm and 50 were > 100 mm. These correspond to 28.1%, 50.3% and 21.6% in the natural unit, and 45.1% and 54.9% in the breakwater, respectively. The pattern cannot be interpreted as recruitment without species-specific size thresholds and replication. The recorded size-class distribution is shown in Figure 3.



Classes are field categories and were not converted to juvenile/adult stages.

Figure 3. Recorded size-class distribution by census unit. Field classes were preserved and were not converted to life stages.

Historical context from 2002 and 2019. The historical surveys show that the 2026 taxa belong to a broader, repeatedly observed coastal assemblage. The 2002 survey reported 72 species and identified *Atherina* sp., *Pagellus acarne* and *Boops boops* as highly abundant schooling taxa; when schooling taxa were set aside, *Sarpa salpa*, *Mugil* sp. and *Thalassoma pavo* remained well represented. The 2019 survey recorded 3,577 individuals and 16 families; the source report gives internally inconsistent richness totals of 42 and 44 species, with Sparidae, Labridae and Serranidae contributing most of the recorded species richness. Five of the nine 2026 taxonomic units - *B. boops*, *S. salpa*, *O. melanurus*, *T. pavo* and *D. sargus* - were also explicitly recorded in both historical datasets.

The 2019 report recorded 6-381 individuals per survey, with a mean of 162.59 ± 110.84 . Small and medium size classes co-dominated overall, and the small-size fraction exceeded 80% at shallow Cala Iris transects. Because the report's east-west interpretation is not fully consistent across its narrative and site descriptions, no directional spatial gradient is inferred here. Cala Iris and the shallow sites highlighted in 2002 are treated as candidate strata for future recruitment monitoring. They do not establish a temporal increase or demonstrate that the 2026 stations sampled the same nursery function.

Sentinel framework generated from the pilot. The pilot identified five linked requirements for a defensible sentinel system: standardized and replicated observation; quality control; indicator production with uncertainty; change detection using homologous seasons and reference units; and an explicit management response followed by evaluation (Figure 4). This chain turns field data into evidence for decisions without treating every fluctuation as ecological change. The minimum design standards for the permanent sentinel programme are summarized in Table 5.

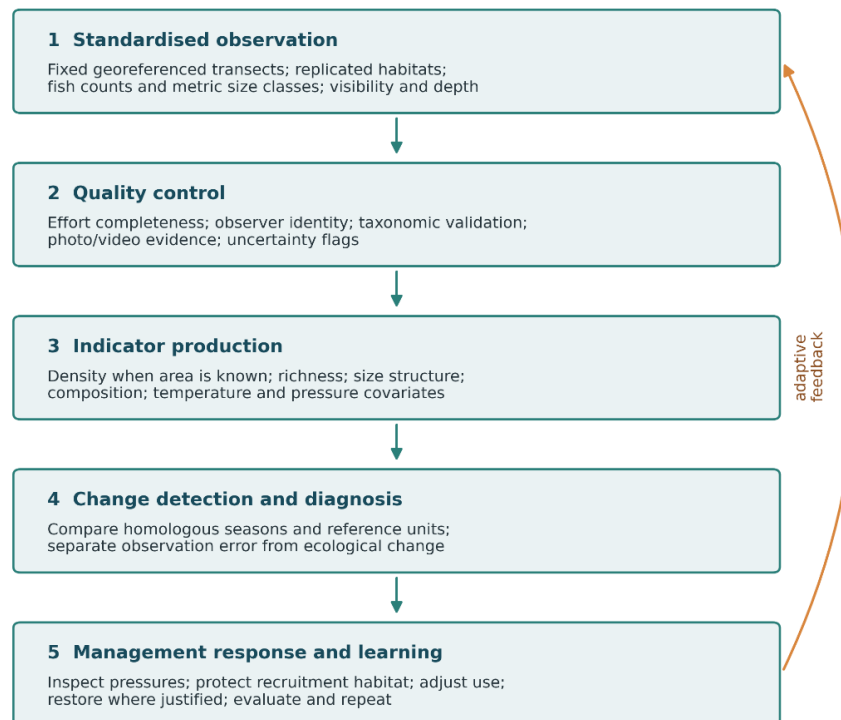


Figure 4. Decision-linked sentinel monitoring cycle adapted to the PNAH marine-observatory architecture developed in the associated doctoral research.

Table 5
Minimum design for the permanent sentinel programme. Numerical alert thresholds require calibration from replicated observations

<i>Component</i>	<i>Minimum standard proposed</i>	<i>Purpose</i>
Spatial design	Fixed georeferenced natural and artificial stations; ≥ 3 independent transects per habitat stratum	Separates habitat variation from single-site effects
Effort	Constant length, width, duration and swimming speed; calculate surveyed area	Enables density and effort-standardized comparison
Timing	Four fixed seasonal windows; repeat at comparable time of day and tidal/sea conditions	Supports seasonal and interannual contrasts
Environment	Depth, horizontal visibility in metres, temperature, swell, wind, cloud cover	Quantifies detectability and physical context
Fish observations	Taxon, count, metric size class; species-specific size thresholds for priority taxa	Tracks composition and demography without retroactive assumptions
Validation	Observer calibration; image/video evidence for uncertain or unusual records	Reduces identification and school-count error
Analysis	Descriptive annual summaries; model-based inference only after adequate replication	Prevents premature causal claims
Decision link	Predefine questions, diagnostic checks and management options before thresholds are activated	Makes monitoring operational and auditable

Discussion

What the pilot shows - and what it does not. The 2026 census documents a strong descriptive contrast: the natural unit held more recorded individuals and taxa, whereas the breakwater assemblage was narrower and dominated by *S. salpa* and *Mugil* sp. This pattern

is ecologically plausible because natural and artificial habitats can differ in structural complexity, refuge, food resources and exposure. Comparable studies have found assemblage differences across protected seagrass, shallow-water and artificial habitat settings (Grimmel et al., 2020; Kiggins et al., 2020). Nevertheless, plausibility is not evidence of causality. One site per category confounds habitat with location, and poor visibility, unrecorded depth and schooling behaviour can all alter observed counts.

The recorded abundance was heavily influenced by two schools entered as 100 *B. boops* and 56 *O. melanurus*. Schooling species create high temporal variance in visual censuses; their presence can dominate total abundance without representing persistent habitat capacity. Future analyses should retain school size as ecologically meaningful while documenting count method, observer confidence and image/video support. Biomass should not be estimated until individual size is recorded with sufficient resolution and survey area is known.

Value and limits of the historical datasets. The 2002 and 2019 datasets are important because they prevent the 2026 pilot from being interpreted in isolation. They show that several taxa prominent in the pilot are recurrent components of the PNAH fish assemblage, and they identify shallow locations where small size classes have previously been concentrated. These findings support a testable monitoring hypothesis: shallow nursery or feeding habitats should be represented explicitly in the permanent station network and evaluated with species-specific size thresholds.

Their value is primarily contextual, not trend-based. The 2002 survey covered more sites and a deeper band and used numerical abundance classes, the 2019 campaign completed 22 standardized 15-minute surveys, and the 2026 pilot lacks documented duration, area, depth and coordinates. Differences in reported richness or abundance may therefore reflect effort, coverage, detectability and classification as much as ecological change. Even the report attributes lower 2019 richness relative to 2002 mainly to sampling effort. A valid long-term comparison should revisit georeferenced 2002/2019 stations with a harmonized protocol, retain a calibration overlap with legacy methods, and compare matched depth–habitat–season strata.

Implications for fisheries and habitat management. Several recorded taxa are relevant to coastal fisheries or trophic functioning, including sparids (*B. boops*, *S. salpa*, *O. melanurus*, *D. sargus* and *D. vulgaris*) and mugilids. Their presence makes the stations potentially useful for monitoring the ecological support that protected habitats provide to small-scale fisheries. However, census counts cannot substitute for catch, effort or fishery-dependent data. A mature observatory should connect UVC indicators with fishing effort, landings or catch-per-unit-effort, compliance records and habitat condition rather than claiming a direct fishery benefit from ecological counts alone (Lloret et al., 2018; Trégarot et al., 2020).

The PNAH thesis identifies *C. nodosa*, macroalgal forests, coralligenous formations and rhodolith beds as habitats that may support refuge, feeding and recruitment functions. The 2026 field label “natural habitat” is too broad to assign these functions to the sampled unit. Future station records must therefore use a controlled habitat classification and quantify structural attributes such as substrate type, vegetation cover, rugosity and exposure. This will allow fish change to be interpreted alongside habitat condition and support better targeting of anchoring control, restoration or nursery protection.

From baseline to adaptive sentinel monitoring. A sentinel station is useful when it offers repeatable sensitivity to change and a clear pathway from observation to action. The first survey establishes a dated baseline, but trends require repeated, comparable observations. For PNAH, four seasonal windows are a practical starting point, with at least three independent transects in each habitat stratum and consistent effort. This design can later be expanded using a power analysis based on observed variance. Homologous-season comparisons are essential because coastal fish assemblages vary through recruitment, migration, reproduction and weather-driven detectability.

Alert thresholds should not be imported as arbitrary percentage declines. They should be calibrated after sufficient data are available and should include diagnostic checks for visibility, observer, effort, depth, school events and habitat change. A possible response sequence is: validate the observation; repeat the survey; inspect local pressures; compare reference stations; then decide whether management action is justified. This approach is consistent with adaptive monitoring, in which data collection is revised as management questions and uncertainty evolve (Lindenmayer & Likens, 2009).

Limitations. The present study has a certain amount of limitations, as follows:

- only one summer campaign and one census unit per habitat were available;
- transect geometry, sampled area, depth, coordinates, time and repetitions were undocumented;
- poor visibility may have reduced detectability and size-estimation accuracy;
- two identifications remained above species level, and unusual school counts lacked attached photographic evidence;
- size classes were coarse and could not be converted into life stages for all taxa;
- the habitats were not matched for depth, exposure or structural complexity;
- historical surveys differed in site coverage, depth bands and abundance measurement; they cannot be combined with 2026 as a standardized temporal series;
- diversity indices have no estimated variance and must remain descriptive;
- the census cannot estimate total fish stock, reserve effect, recruitment rate or climate impact.

Conclusions. The 2002 and 2019 surveys are important historical references: they document a diverse coastal assemblage, recurrent sparid and labrid components, and shallow areas with high proportions of small fish. They should guide station placement and monitoring hypotheses, but they cannot be joined directly to the 2026 counts as a standardized temporal trend. The 19 August 2026 pilot provides a traceable fish-assemblage baseline for the proposed PNAH sentinel station: 290 recorded individuals, nine taxonomic units and a marked descriptive contrast between a natural habitat unit and an artificial breakwater. Its principal scientific value lies not in demonstrating a habitat effect, but in revealing the design and metadata required for one to be tested. Fixed georeferenced transects, replicated habitat units, standardized effort, measured visibility and depth, taxonomic evidence and seasonally homologous sampling would transform the available evidence into a defensible long-term programme. Embedded within the PNAH marine observatory, the resulting indicators can support adaptive fisheries and habitat management by linking ecological change to environmental context, local pressures, diagnostic verification and evaluated management responses.

Author Contributions. SK conceived the study, conducted the field investigation, curated and analysed the data, prepared the figures, and wrote the original manuscript; HB contributed to the methodology, validation and interpretation of the results and critically revised the manuscript; HCD supervised the study, contributed to the methodology and interpretation, and critically revised the manuscript. All authors reviewed and approved the manuscript.

Conflict of Interest. The authors declare no known competing financial interests or personal relationships that could have appeared to influence the work.

Data Availability. The consolidated data supporting the descriptive results are available from the corresponding author on reasonable request, subject to the data-sharing conditions of the Parc National d'Al Hoceima and the Agence Nationale des Eaux et Forêts. Sensitive or incomplete station metadata are not publicly reported.

Ethics Statement. The survey was non-destructive and involved visual observation of free-ranging fish only; no fish were captured, handled or removed, and no experimental procedures were performed.

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