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Identifying physico-chemical indicators to assess the ecological quality of Mediterranean rivers in their dry-phase

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ABSTRACT

Temporary rivers, which support a dry phase, are abundant and expected to increase worldwide in the face of global change. The European Water Frame Directive still overlooks these ecosystems since information about a proper ecological assessment is scarce, especially during the complete absence of water. As a result, rivers that mostly run dry cannot be assessed yet. We aimed to examine the potential use of physico-chemical elements as suitable indicators during the dry phase of temporary rivers. We monitored 41 Mediterranean rivers (dry channel sediments and co-occurring riparian soils) previously categorized according to their level of anthropogenic impact by using both qualitative attributes at local scale (Mediterranean Reference Criteria) and by land use coverage at catchment scale. We examined common physico-chemical parameters used in monitoring programs (nutrients, electrical conductivity, pH), as well as organic matter and carbon, albeit measured in sediments and soils, to test whether they significantly changed across the monitored sites as impact level increased. Results from both approaches showed that leaching nitrate (NO_3^-) from dry-channel sediments and riparian zones increased significantly with the level of exposure of stressors in the study sites. High NO_3^- content, especially within sediments, seemed to respond to agriculture presence, which supports this parameter as a suitable indicator. However, natural variability linked to climate and geology of the study area hinders the reliability of the rest of parameters as robust indicators of the dry phase. We encourage more research across different regions to refine the physico-chemistry of dry phase to advance in properly assessing the ecological quality of temporary rivers.

1. Introduction

As rivers and streams are landscape elements that intercept nutrient and pollutants from watersheds, they are among the most impacted ecosystems by human pressure, thus, they experience strong degree of alteration of their natural structural and functional characteristics which ultimately shape their ecological quality (Vörösmarty et al., 2010). In fact, nutrient enrichment from diffuse and point sources is nowadays among the main causes of degradation in European rivers (Erba et al., 2022).

In 2000, the Water Frame Directive (WFD; 2000/60/EC; European Commission 2000) was established with the aim of achieving "good ecological status" in all water bodies in European countries for 2015 through the application of integrative methodologies that included the assessment of all water-related impacts (Lemm et al., 2021). Such evaluation integrated mainly the use of biological indicators, supported by factors informing about physico-chemistry and hydro-morphological components of the state of fluvial ecosystems and their catchments (Hering et al., 2010). To identify levels of degradation in the assessment of a determined waterbody, benchmarks are required. Consequently, the WFD adopted the Reference Condition Approach, which uses a previously defined range of reference locations (that is, waterbodies lacking of human perturbations) to determine the ecological state expressed as a deviation of such reference conditions (Sánchez-Montoya et al., 2012).

A critical handicap of the WFD is that it fails to consider fluvial ecosystems lacking surface water (Carvalho et al., 2019). Most of the paradigms in ecology of rivers have emerged from perennial ecosystems and thus management plans (included WFD) have also been developed for rivers presenting a perennial flow year-round (Skoulikidis et al., 2017). As a result, proper biomonitoring programs to determine ecological quality of temporary rivers are still lacking and challenging to develop. Indeed, bioindicators of dry-phase, i.e., lack of surface water, are barely starting to be developed (Steward et al., 2018; Stubbington et al., 2019). However, those indicators related to physico-chemical

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state are still lacking since their potential use to assess ecological status during the dry phase has been fully ignored until now.

Temporary rivers (those having a dry phase) comprise the challenging ecosystems in terms of ecological assessment since they combine flowing and drying conditions over space and time (Datry et al., 2023a). Aside from natural water fluctuations, drying can result from human-induced activities and affects rivers, lakes, and reservoirs (Leigh et al., 2016). Sixty percent of the worldwide fluvial network is dominated by non-perennial reaches (Messager et al., 2021), a proportion expected to increase with global warming and the increasing human pressure on rivers (Döll and Schmied, 2012; Datry et al., 2023b).

Dry phase (or dry riverbeds from a spatial perspective), considered as the key element differing from perennial waters, still represents a poorly investigated feature of temporary rivers (Bonada et al., 2024; Datry et al., 2014; Steward et al., 2012). Examples of an extreme non-perennial ecosystem type from an hydrological perspective are ephemeral streams in arid regions (also known as *dry rivers*), that only flow anecdotally for a few hours or days after heavy rainfalls but are mostly dry year round (Vidal-Abarca et al., 2020). In fact, from a conceptual view, dry riverbeds have been posed as mosaic habitats which host both terrestrial and aquatic features (Steward et al. 2012). Indeed, many dry riverbeds hardly support aquatic conditions and tend to undergo "terrestrialization" processes such as the colonization of terrestrial plants (Arce et al., 2019; Vidal-Abarca et al., 2020) and terrestrial invertebrates (Sánchez-Montoya et al., 2020; Steward et al., 2022).

The importance and the study of the dry phase in ecological research is relatively new. Empirical studies on dry riverbeds highlight distinctive features from aquatic habitats with important socio ecological values, including flood control and groundwater recharge (Datry et al., 2023a), as well as their contribution to human well-being (Nicolás-Ruiz et al., 2021). Besides, they have been pose as key elements for the biogeochemical functioning of the whole river network (Keller et al., 2020; Paranaíba et al., 2022) and for the distribution of terrestrial mammals (Sánchez-Montoya et al., 2022; 2023). Despite their evident scientific and social values, the main guidelines for biodiversity, conservation, and management of freshwaters resources ignore the existence of hydrological discontinuities throughout the fluvial networks. Consequently, temporary rivers still present a bad state of conservation since dry phases are not properly valued and recognized by water managers and society in general (Steward et al., 2012; Vidal-Abarca et al., 2020).

The Mediterranean region has a long history of anthropogenic disturbances of riverine ecosystems and soils at the basin and corridor scales, especially due to intensive agriculture (Aguiar and Ferreira, 2005). Furthermore, in arid zones, drought and low organic matter of soils associated to aridity make temporary rivers easily vulnerable to soil erosion and inputs of nutrient and other terrestrial elements (Ferreira et al., 2022). Such elements are subjected to minimum opportunities of dilution with considerable impacts on the ecosystem, which are probably exacerbated in dry sections of the rivers that hardly support a flow year-round.

After decades of research on river ecosystems monitoring, now it is evident that there is an urgent need to develop tools to assess the ecological status of Mediterranean rivers during their dry phase for their proper management and conservation (Prat et al., 2014; Skoulikidis et al., 2017). Biological indicators such as aquatic macroinvertebrates, hydro-morphological, and physico-chemical elements have been traditionally used in the legislative frameworks to asssess rivers and streams *sensu* WFD −2000. Among the general physico-chemical parameters included in the WFD, nutrient levels, conductivity, oxygen, acidification, and thermal status in surface waters are included. In a previous work, Sánchez-Montoya et al. (2012) provided reference conditions based on surface waters physico-chemical parameters. The integration of physico-chemical parameters in the ecosystem assessment of where surface water is lacking appears initially unfeasible. Indeed, early studies providing indicators to detect human pressures on dry riverbeds are based on biotic elements such as diatoms, plants (Stubbington et al., 2019), or terrestrial macroinvertebrates (Steward et al., 2011; 2018), albeit with no robust guidelines to date (Jupke et al., 2022). Compared with intermittent rivers, which tend to support flowing conditions in the wet season, searching indicators for dry phase in ephemeral streams is crucial since water appears punctually, thus it both makes it difficult to monitor and does not necessarily reflect a rivers ecosystem health. In fact, the directive 2008/105/EC, daughter of WFD and relative to environmental quality, suggests the use of sediments to monitor pollutants for a long-term, as their application have been useful to detect harmful elements during vulnerable low flow periods of intermittent rivers (Skoulikidis, 2008; Tzoraki et al., 2015).

We anticipated that during dry conditions certain physico-chemical characteristics measurable in the sediments of dry channels could hold reliable information on the specific ecological state of a temporary river (whether it is pristine or impacted). Given the position of fluvial elements within landscapes, the information stored in riverbed sediments is sensitive to environmental changes, thus reflecting the main use (natural or human-transformed) of the drainage catchment (Burton, 2002).

As many rivers are undergoing drying, a trend that is expected to increase with global change, developing assessment tools specific to drought periods is a priority. Furthermore, knowing the physico-chemical quality of dry riverbeds is also critical to assess the potential impacts to the ecological quality of downstream habitats when water transport resumes after heavy rainfall events, especially if dry riverbeds discharge any waterbody type (e.g., coastal areas, reservoirs or rivers).

Here, we aimed to identify potential physico-chemical elements as indicators of river quality during the dry phase. Focusing on a Mediterranean catchment, we monitored 41 temporary rivers during their dry phase and subjected to variable human impacts. Since riparian zones in the study catchment suffer from anthropogenic impacts and tightly interacted with the uplands (Bruno et al., 2014), we decided to include riparian soils in addition to the channel sediments. These habitats may provide key information to achieve a more thorough vision of the ecological status that might go undetected by only monitoring channel sediments. We mainly focused on easily measurable- target parameters previously used in the assessment of Mediterranean perennial rivers according to the European WFD (Sánchez-Montoya et al., 2012). We explored which parameters were sensitive to variable anthropogenic impact by following two approaches: i) a (qualitative) multiple-stressor approach and ii) a (quantitative) single-stressor approach. For the multiple-stressor approach, we used a set of 29 criteria, part of them already used for Mediterranean rivers (MRC, Sánchez-Montoya et al., 2009) but adapted to dry riverbeds, which classifies sites in categories of variable impact. For the single-stressor approach, we characterized rivers according to a gradient of variable land-use. We expected that the physico-chemical parameters measured in the study sites during dry conditions would be sensitive in response to the different impact levels, thus potentially acting as suitable ecological indicators.

We hope that our results provide new insights on using physico-chemical conditions during the dry phase for a complete ecological assessment of rivers so as to help decision makers in properly monitoring and conserving non-perennial rivers.

2. Material and methods

2.1. Study area and field sampling

The study was conducted in the Segura river catchment, in the Southeast Iberian Peninsula (Fig. 1). The climate is Mediterranean and characterized by very warm dry summers with annual temperature ranges between −2°C and 42 °C and precipitation that varies between 280 and 1000 mm (MIMAN, 2000). Besides, given it is a Mediterranean climate, fluvial systems in the studied region exhibit high seasonality in the hydrological regimes with high annual and inter-annual variability in the discharges and severe drying and flooding periods (Gómez et al., 2005).

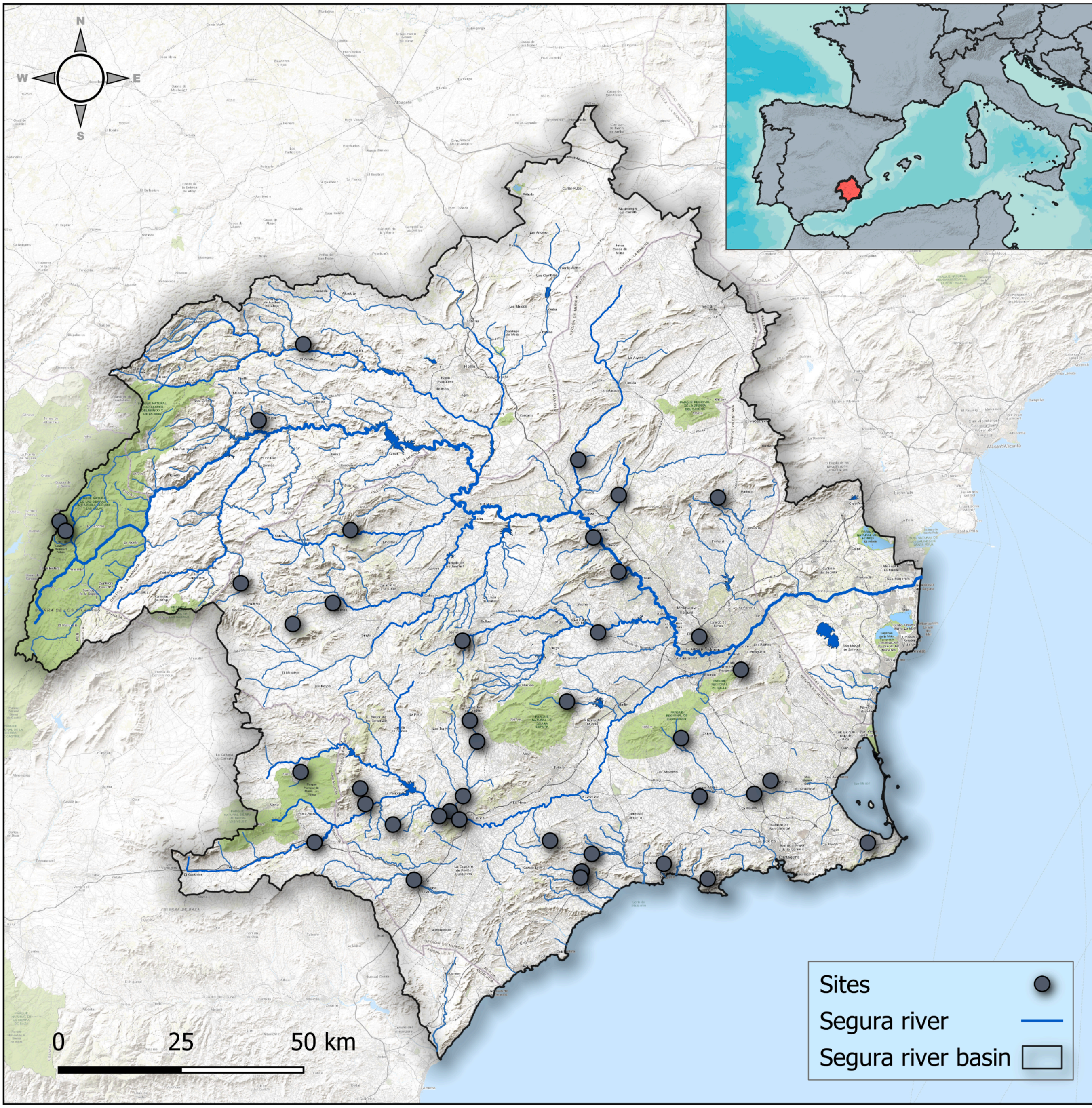

Fig. 1. Map of the Segura river catchment and study locations.

A total of 41 temporary streams presenting a dry phase were studied. In this study, we considered a dry phase to be the absence of visible surface water over the channel, thus we excluded isolated pools. As to the hydrological regime, 32 out 41 were considered *ephemeral* and *dry rivers* which support flowing periods shorter than dry periods, while only 9 sites were *intermittent* having flowing periods longer than dry periods. The dominant geology in the drainage area of the studied rivers was metamorphic (n = 10), evaporitic (n = 7), calcareous (n = 15), and marly (n = 8). The presence of evaporitic rocks of either Miocen or Triassic origin, rich in NaCl and SO_4Ca_2, provides high salinity to many soils and waterbodies (Gómez et al., 2005; Millán et al., 2011).

The variation in native riparian vegetation across the monitoring sites has strong dependence on water availability and geology (Salinas and Casas, 2007). Overall, native vegetation was dominated by species able to cope with water stress and natural soil salinity associated to geology with forestry species limited to more humid sites in the north-west. For instance, shrub species of *Lygeum spartum*, *Brachypodium retusum*, *Salsola genistoides*, *Helichrysum stoechas*, *Thymelaea hirsuta* and *Rosmarinus officinalis* are common. Halophytic species such as *Limonium* spp. and *Sarcocornia* spp. are also common in evaporitic geology. In some locations, higher sized species of *Tamaris canariensis* and *Nerium oleander* trees can be found. Among the forestry species, *Pinus halepensis* is the dominant type in more humid rivers.

The main anthropogenic impacts in the catchment studied include river channelization, compaction, presence of dams, urban and rubbish waste, and removal of native riparian vegetation for more dry and irrigated crops (Gómez et al., 2005; Vidal-Abarca et al., 2020).

In each monitoring site, a 30 m long- stream reach was selected to sample two habitats: the channel sediments and riparian soils. In each habitat, in the top 5 cm, three replicates were collected using a hand shovel, sieved (2 mm-mesh), and stored in plastic bags for their transport to the laboratory in dark conditions. Samples were processed for the set of parameters selected in this study potentially sensitive to anthropogenic pressures at local and catchment scale (Table 1).

2.2. Multiple-stressor (qualitative) classification: Mediterranean reference criteria

To examine which parameters were sensitive to the different degrees of exposure to anthropogenic impacts on rivers, we applied a multi-stressor approach by using the Mediterranean Reference Criteria (MRC) adapted from Sánchez-Montoya et al. (2009) and Prat et al. (2014). These have been previously used in the assessment of Mediterranean rivers (Soria et al., 2020), which included non-perennial sites through water-column physico-chemical parameters (Sánchez-Montoya et al., 2012). The set of 29 criteria used in this study reflects the characteristics of Mediterranean rivers and their most frequent disturbances that are applicable to a river with no water flow (Table 2). Unlike the traditional MRC, criteria reflecting conditions concerning the water-column impacts were removed; however, criteria reflecting the main land uses in circular buffers of 250 m and 500 m diameters were included (Table 2).

MRC method distinguished three levels of exposure to stressors according to the number of criteria that each river fulfilled. Thus, we classified sites according to the level of impact as: *low impacted; moderately impacted, and highly impacted* (Fig. 2). Only sites that fulfilled $\geq$ 22 criteria were considered of low impact. When sites fulfilled between 22–16 criteria and $\leq$ 15 criteria, these were classified as moderately- and highly- impacted sites, respectively.

2.3. Single-stressor (quantitative) classification: Land-use coverage data

Sites were also characterized by their percentage of different land-uses using the ArcGis vs. 10.3 software (ESRI, 2011). In this study we used the land-cover cartography from the Spanish Land Cover Information System (SIOSE; vs.2014). In each sampling site, three land-covers were calculated: the drainage surface area at each sampling point and two circular buffers of 250 m- and 500 m- diameters similar to the widely used approach in terrestrial ecosystems for impact characterization (Maiorano et al., 2008; Buffa et al., 2018). The sub-catchment land-cover was obtained from the IDE (Infrastructuras de Datos Espaciales) available from the Spanish Ministry of Ecological Transition data set (https://www.miteco.gob.es/es/cartografia-y-sig/ide.html). Three main land uses: irrigated agriculture, dryland agriculture, and urban use were calculated for each land-cover.

Table 1
Physico-chemical parameters used in this study and the main information they provide with regards to human impacts. Common units and measurement type in both perennial river and temporary river are shown. DM: dry mass. N/A: not applicable.

Parameter	Perennial river	Temporary river (channel sediments and riparian soils)	Potential information related to anthropogenic impacts
Nitrate (NO_3^-) concentration	mg L^{-1} in water column	mg g^{-1} DM in leachates	Presence of irrigated agriculture and fertilizer influence and livestock
Ammonium (NH_4^+) concentration	mg L^{-1} in water column	mg g^{-1} DM in leachates	Urban waste and livestock Impacts
Orthophosphate (PO_4^{3-}) concentration	mg L^{-1} in water column	mg g^{-1} DM in leachates	Agriculture, urban waste, and livestock impacts
Dissolved organic carbon (DOC) concentration	mg L^{-1} in water column	mg g^{-1} DM in leachates	Urban waste and organic fertilization impacts
pH	No units in water column	No units in leachates	Mining and agricultural impacts
Electrical conductivity	μS cm^{-1} in water column	μS cm^{-1} in leachates	Presence of irrigated agriculture; urban water inputs
Particulate organic matter content	N/A	% ash free dry mass in solid samples	Development status of riparian vegetation (removal of native vegetation and impervious surfaces, land use change)

Table 2
Mediterranean Reference Criteria (MRC) used to qualitatively classify the study sites into three impact levels at different spatial scales.

Category	Attribute
Riparian vegetation zone (Reach scale)	1. Cover and composition appropriate for the type and geographical location of the river2. Lateral connectivity between river and riparian corridor is maintained (no cultivation and significant impervious area in riparian zone)
Exotic species (Reach scale)	3. No significant influence of exotic species of plants and animals on autochthonous organisms
River morphology and habitat conditions (Reach scale)	4. Representative diversity of substrate materials appropriate for the type5. No canalization (stream bottoms and stream margins must not be fixed)6. No transversal structures “dams” (no retention of sediments) 7. No sand or gravel extraction8. No use as roads or tracks (human or vehicles)9. No urbanization (parking places, parks and building construction) 10. No presence of rubbish
Hydrological conditions (Drained basin)	11. No alterations to the natural hydrological regime and discharge 12. No deviation of water to irrigation and other activities 13. No effect of water transfer between catchments 14. No effect on groundwater
Point-source pollution (Reach scale)	15. No urban waste inputs 16. No industrial waste inputs 17. No agricultural waste inputs
Land uses (Drained Basin)	18. Dry land farming < 20 % of drainage area (cereal, vineyard and tree crops as olive) and not connected to riparian vegetation zone 19. Intensive irrigated farming < 3 % in drainage area (rice field, irrigated vineyard and other irrigated fruit tress) and not connected to riparian vegetation zone 20. Urban and industrial uses < 1 % (impervious area) 21. Natural land uses > 80 % in drainage area
Land uses (500 m buffer)	22. Dry land farming < 10 % of drainage area (cereal, vineyard and tree crops as olive) and not connected to riparian vegetation zone 23. Intensive irrigated farming < 0.2 % in drainage area (rice field, irrigated vineyard and other irrigated fruit tress) and not connected to riparian vegetation zone 24. Urban and industrial uses < 0.5 % (impervious area) 25. Natural land uses > 90 % in drainage area
Land uses (250 m buffer)	26. No dryland farming of drainage area (cereal, vineyard and tree crops as olive) and not connected to riparian vegetation zone 27. No intensive irrigated farming < 0.2 % in drainage area (rice field, irrigated vineyard and other irrigated fruit tress) and not connected to riparian vegetation zone 28. No urban and industrial uses 29. Natural land uses < 100 % in drainage area

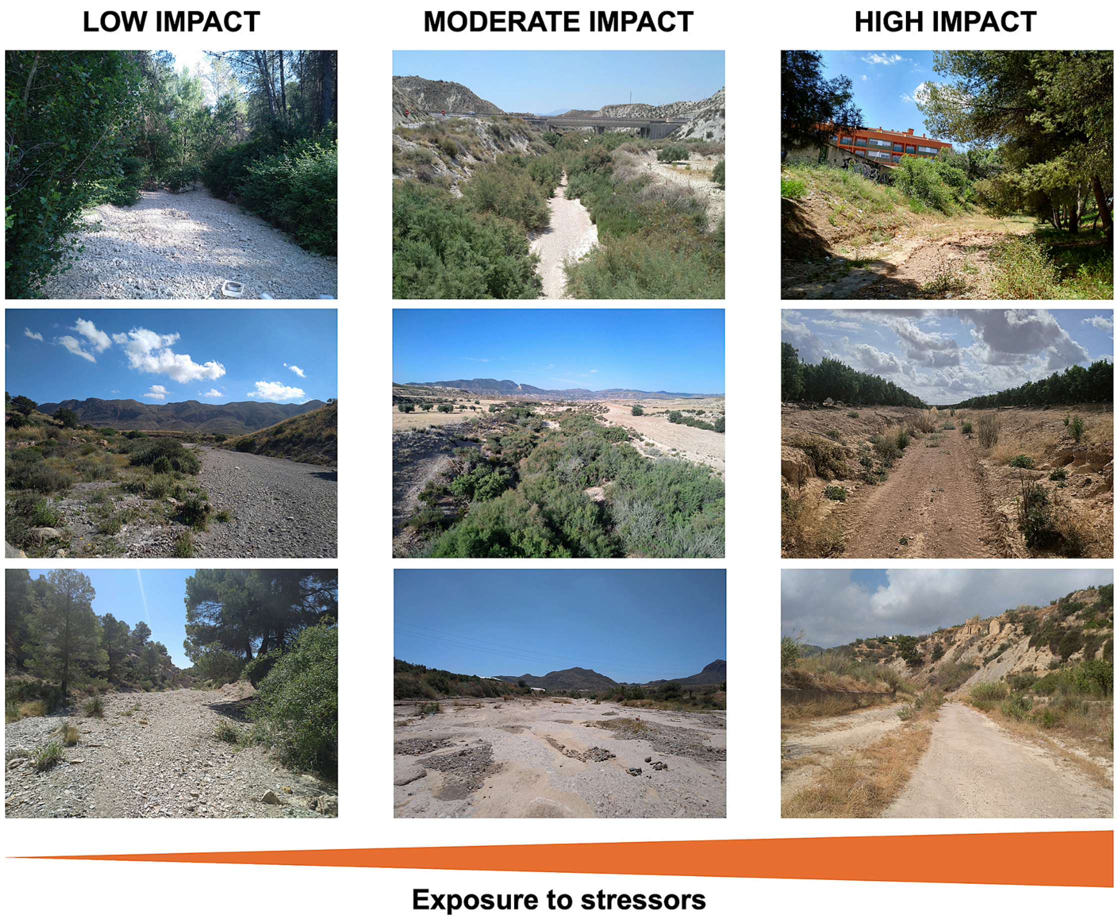

Fig. 2. Examples of different Mediterranean temporary rivers in their dry phase according to their level of anthropogenic impact.

2.4. Laboratory analysis

Immediately after collection, samples were characterized in the laboratory according to organic matter (OM). First, ~10 g of subsample were dried at 100 °C for 24 h. The OM % was estimated by combusting the dried subsample at 550 °C for 4 h. Then, all samples, sediment channels and riparian soils, were kept air dried under laboratory conditions safe from humidity and light until all analysis were run within a month after collection.

Inorganic nutrients and dissolved organic C (DOC) were analyzed by generating leachates from sediments and soils in a proportion 1:5 (solid: liquid). Ammonium (NH_4^+) was extracted by using KCl 2 M (Mulvaney, 1996), while nitrate (NO_3^-), orthophosphate (PO_4^{3-}), and DOC were extracted by using ultrapure water. Approximately 10 g of sediment were agitated during 1 h in 50 mL Falcon® inert tubes by using a rotary shaker. Afterwards, tubes were centrifuged at 3400 rpm for 10 min and the supernatant was filtered through pre-combusted Whatman GF/F filter (England, UK). NH_4^+ was immediately determined by using a phenol hypochlorite method (Solorzano, 1969) in an autoanalyzer EasyChem Plus, Systea Analytical Technologies, Italy). Subsamples for the rest of parameters were frozen and further analyzed for NO_3^- and PO_4^{3-} by ionic chromatography (HPLC, model 861, Metrohm AG, Herisau, Switzerland) and for DOC with a TOC-analyzer (Analytik jena multi N/C 3100, Germany). They were expressed as μg per g of dry mass (DM). Similarly, additional leachates were produced using pure water to measure pH and electrical conductivity (EC) using probes (Intellical HQD, Hach Lange, CO, USA).

2.5. Statistical analyses

To examine which physico-chemical parameters changed as a function of the ecological status of the study sites, we conducted two-way ANOVAs to test differences among the three categories identified by the MRC method (*low- moderately- and highly- impacted*) according to the habitat type (*channel and riparian*). When ANOVAs were significant, Tukeýs post-hoc tests were done for pair comparisons between the three categories by fixing the habitat type.

We also performed a Principal Component Analyses (PCA) (Primer vs.6, UK) to examine the distribution of study sites according to their different land-use coverage. We excluded from PCA those variables that appeared to be strongly correlated. The weight of a variable on a PCA component was considered significant when its loading was > 0.6. We then created a single-stressor gradient of land-use coverage to examine the sensitiveness of the physico-chemical parameters to catchment land use. To do that, the first-axis score of the PCA was considered as a stressor gradient and the projected site values were standardized along this axis (from 0 to 1). This first component was used as an independent variable to run simple linear regressions with physico-chemical variables.

Variables were log-transformed whenever necessary to meet the assumption of normality. Significant results were considered significant if $P < 0.05$ and marginally significant if $0.05 > P < 0.1$. ANOVA and regression analysis were performed using R software (R Development Core Team, 2008).

3. Results

3.1. Classification of the study sites according to MRC and main environmental features

After applying the MRC, 18 out 41 sites were classified into low impacted, 14 into moderately impacted, and 9 into highly impacted. Predominant pressures affecting study sites were related to the criteria 2, 29, 25, 26, 22, and 21 (attributes included in Table 2), respectively: lateral connectivity between river and riparian corridor (fulfilled by 46 % sites), natural land uses equal to 100 % in the 250 m buffer, and natural land uses > 90 % in the 500 m buffer (fulfilled by 17 % sites), no dry land farming in the 250 m buffer (fulfilled by 32 % sites), dry land farming < 10 % in the 500 m buffer (fulfilled by 39 %sites), natural land uses > 80 % in drainage area (fulfilled by 41 %sites).

Based on the number of sites having low impact, 5 out 18 presented intermittent flow and the rest supported ephemeral flow (i.e., dry rivers). In terms of the dominant geology, most of them (11 out 18) were calcareous, 4 metamorphic, and 3 marly. As to the sites having moderate impact, most of them (12 out 14) were ephemeral and had all geologies represented. Among the 14 sites, 4 were calcareous, 4 metamorphic, 4 marly, and 2 evaporitic. Considering the rivers highly disturbed, 7 out 9 supported ephemeral hydrological regimes and 2 were intermittent. Among them, 5 out 9 had evaporitic geology, while in the rest of rivers, 2 were metamorphic, 1 calcareous, and 1 marly.

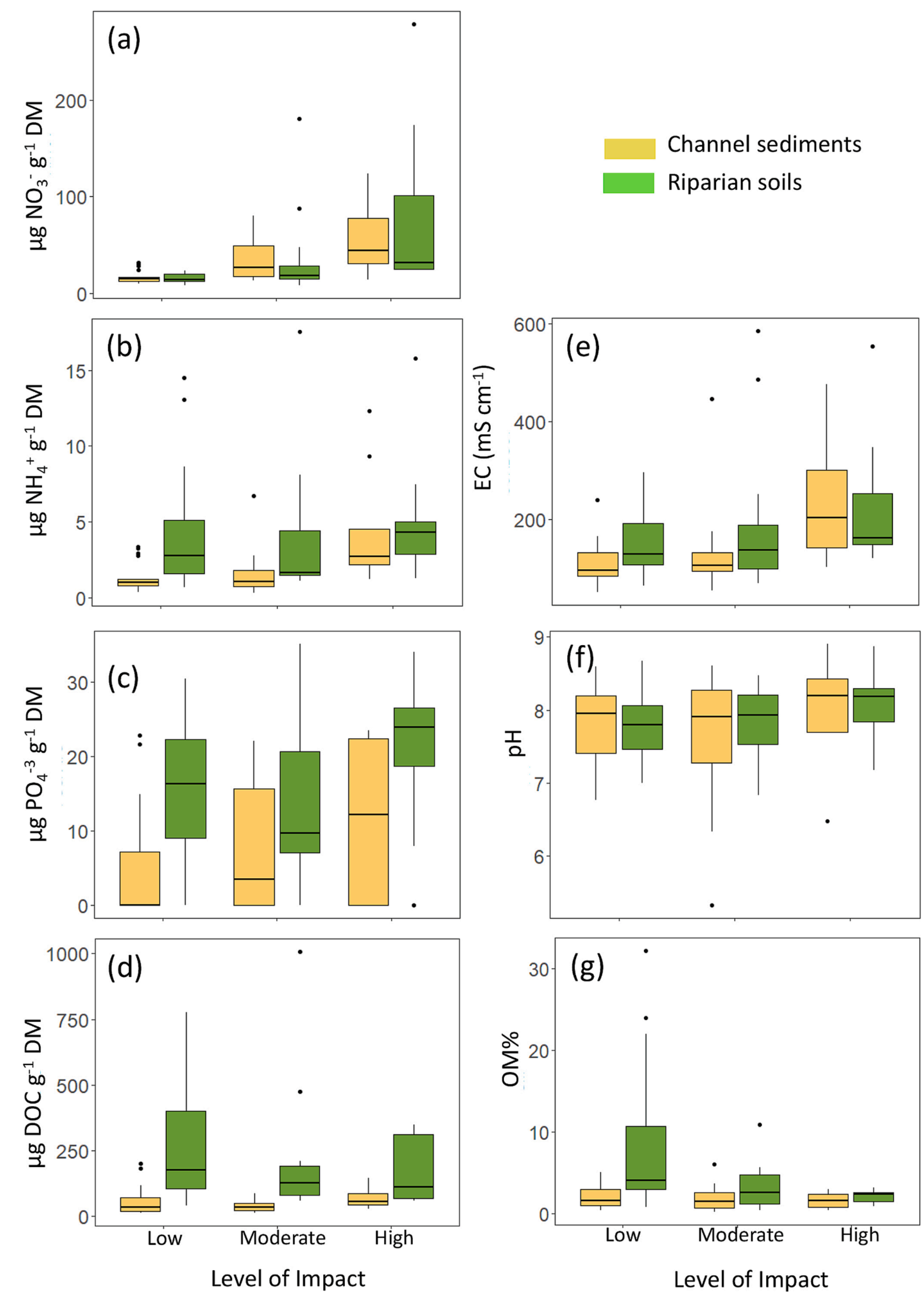

Fig. 3. Boxplots of the physico-chemical parameters studied in study sites (sediments and co-occurring riparian soils) classified by their level of anthropogenic impact according to qualitative Mediterranean Reference Criteria. The median values (central line), 25th and 75th percentile values (box) and the maximum and minimum values are shown. DOC: dissolved organic carbon, OM: organic matter, EC: electrical conductivity, DM: dry mass.

3.2. Variation of the physico-chemical parameters among categories according to MRC

The two-way ANOVAs detected significant differences between the three levels of impact for NO_3^-, EC, and OM (Fig. 3a, e, g, respectively). In the case of NO_3^- and EC, such differences were found regardless of the habitat type (*category* factor df = 2, F = 21.54, P < 0.001; F = 4.83, P = 0.01, respectively). The lowest concentrations of NO_3^- were observed in sites of low impact (mean ± SD in channel and riparian habitats, respectively of 16 ± 7 and 15 ± 5 µg NO_3^- g DM^{-1}), while the greatest values were found in the moderately- (34 ± 21 and 36 ± 46 µg NO_3^- g DM^{-1}) and highly- disturbed ones (56 ± 36 and 81 ± 89 µg NO_3^- g DM^{-1}) (Fig. 3a).

In the case of the EC, values in the moderately impacted sites (in channel and riparian habitats, respectively = 1,203 ± 1,834 and 2,068 ± 6,029 µS cm^{-1}) were significantly lower than those observed in sites of low (166 ± 189 and 148 ± 65 µS cm^{-1}) and high impact (515 ± 644 and 386 ± 336 µS cm^{-1}). EC values detected in the low- and highly-impacted sites were not significantly different (Fig. 3e).

The ANOVA results for OM detected marginally significant differences between categories in dependent of the habitat (*category x habitat* interaction factor: F = 2.87, P = 0.06) (Fig. 3g). While OM% in channel sediments did not appear different among categories (mean values between categories = 1.9 % and 2.9 %), mean values in riparian soils varied from 2.3 % to 8.6 %, where the low-impact sites showed the highest % if compared with moderately- and highly- disturbed ones.

The rest of nutrients; NH_4^+, PO_4^{3-}, and DOC (Fig. 3b, c, d, respectively) tended to show higher values in channel sediments as the level of impact augmented. However, concentrations did not appear statistically different among the three categories, and analysis only detected significant differences between habitats (*habitat* factor df = 1; F_{NH4+}=8.62, P = 0.004; F_{PO4}^{3-} = 14.23, P = 0.0003, F_{DOC} = 60.6, P < 0.001). For the three parameters, riparian soils generally exhibited larger concentrations than those observed in channel sediments. Thus, while the NH_4^+ (µg g DM^{-1}) concentrations in channel sediments ranged from 0.36 to 15.7 across sites, it varied from 0.8 to 22.4 µg g DM^{-1} in riparian soils. In the case of PO_4^{3-}, it varied in a wider range, from 0 to 23 µg g DM^{-1} in channel sediments, and from 0 to 35 µg g DM^{-1} in riparian soils. DOC concentrations showed values raining from 12 to 202 µg g DM^{-1} in channel sediments and from 42 to 1,006 µg g DM^{-1} in riparian soils across the studied sites.

Finally, no significant differences were found in pH, which hardly changed among categories and habitats, with values ranging from 7.6 to 8.0 in channel sediments and from 7.7 to 8.1 in riparian soils (Fig. 3f).

3.3. Response of the physico-chemical parameters to variable land-use gradient

The first axis of the PCA (Fig. 4) explained 38 % of the variance in the 41 study sites with a positive loading of anthropogenic impact gradient as indicated by the land-uses coverage in buffer scale of 250 m of irrigated agriculture; (R = 66 %) and urban (R = 60 %) uses, as well as the irrigated land-use at draining catchment scale (R = 72 %). The variables that correlated negatively with component 1 were the natural land-uses in a buffer scale of 250 m (R = -77 %) and 500 m (R = -83 %), and the natural land-use at draining catchment scale (R = -61 %). The second axis of the PCA accounted 23 % of the total variance and correlated negatively with dry agriculture, both in a buffer scale of 250 m (R = -88 %) and drainage catchment scale (R = -87 %).

The PCA showed that half of the highly impacted study sites (n = 5 out 9) presented larger percentages of anthropogenic land uses, while in the rest (n = 4), dry agriculture and natural coverage were the main land uses. On the contrary, most of the low disturbed sites clearly had natural coverages as the dominant land-use. The moderately disturbed category had representative sites of both natural and dry-agriculture coverage as main land-uses (Fig. 4).

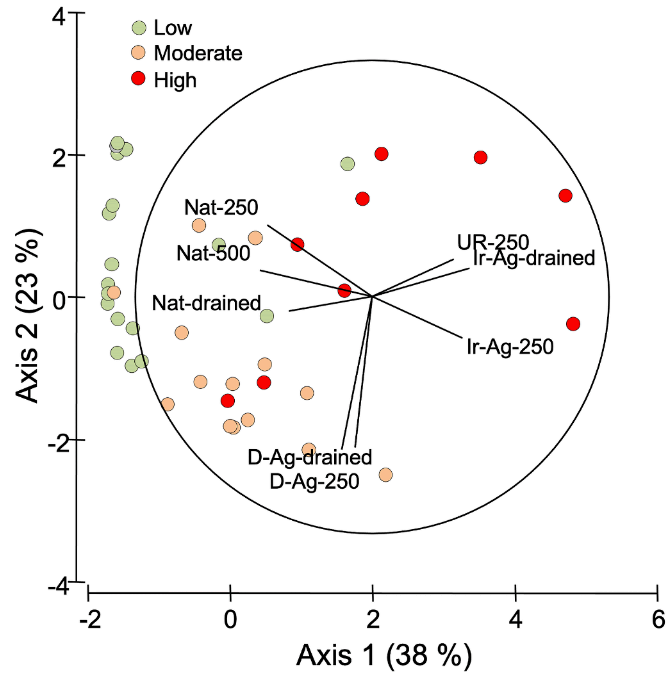

Fig. 4. Principal component analysis of the study sites according to land uses in their catchments. Different colours of the study sites represent their variable level of stressor exposure according to the MRC classification used in the qualitative approach.

The simple regression analyses made using the first PCA component as an independent variable, indicative of impact gradient, showed significant relationships between some physico- chemical parameters with anthropogenic land uses (Fig. 5). Only in channel sediments, NO_3^- and NH_4^+ increased significantly with the increasing impact gradient (respectively R^2 = 0.50, P < 0.001 and R^2 = 0.32, P < 0.001, Fig. 5a, b). In the case of EC, this variation was significant and positive in both habitats (in channel R^2 = 0.18, P < 0.01; in riparian zone R^2 = 0.12, P < 0.05, (Fig. 5e). Only in riparian soils, pH increased in response to anthropogenic land uses, which condensed the impact gradient (R^2 = 0.16, P < 0.05) (Fig. 5f) while OM% decreased (R^2 = -0.11, P < 0.05) (Fig. 5g). In any habitat, the content in PO_4^{3-} and DOC did not show any significant response (Fig. 5c, d, respectively).

4. Discussion

4.1. Sensitiveness of physico-chemical parameters to anthropogenic pressures during the dry-phase

By focusing on a Mediterranean fluvial network, our study provides insight into how physico-chemical parameters, examined in dry sediments and co-occurring riparian soils, respond to different levels of exposure to stressors affecting temporary rivers at local and catchment scale. Through two complementary methodological approaches, our results indicate that NO_3^-, EC, OM, NH_4^+, and pH measured during the dry phase were at some level sensitive elements to anthropogenic impacts, thereby potentially informing us about the ecological state of a river's ecosystem when water is no present.

By using the qualitative approach that followed 29 criteria (MRC; Sánchez-Montoya et al., 2009), we found that the content of NO_3^- leached from channel sediment and riparian soils increased in the moderately- and highly- impacted sites, respectively, if compared with low impacted locations, which makes it a suitable indicator. Likewise, OM% in riparian soils was significantly lower in sites with moderate and high impact, suggesting potential degradation of riparian coverage as number of local human stressors increase.

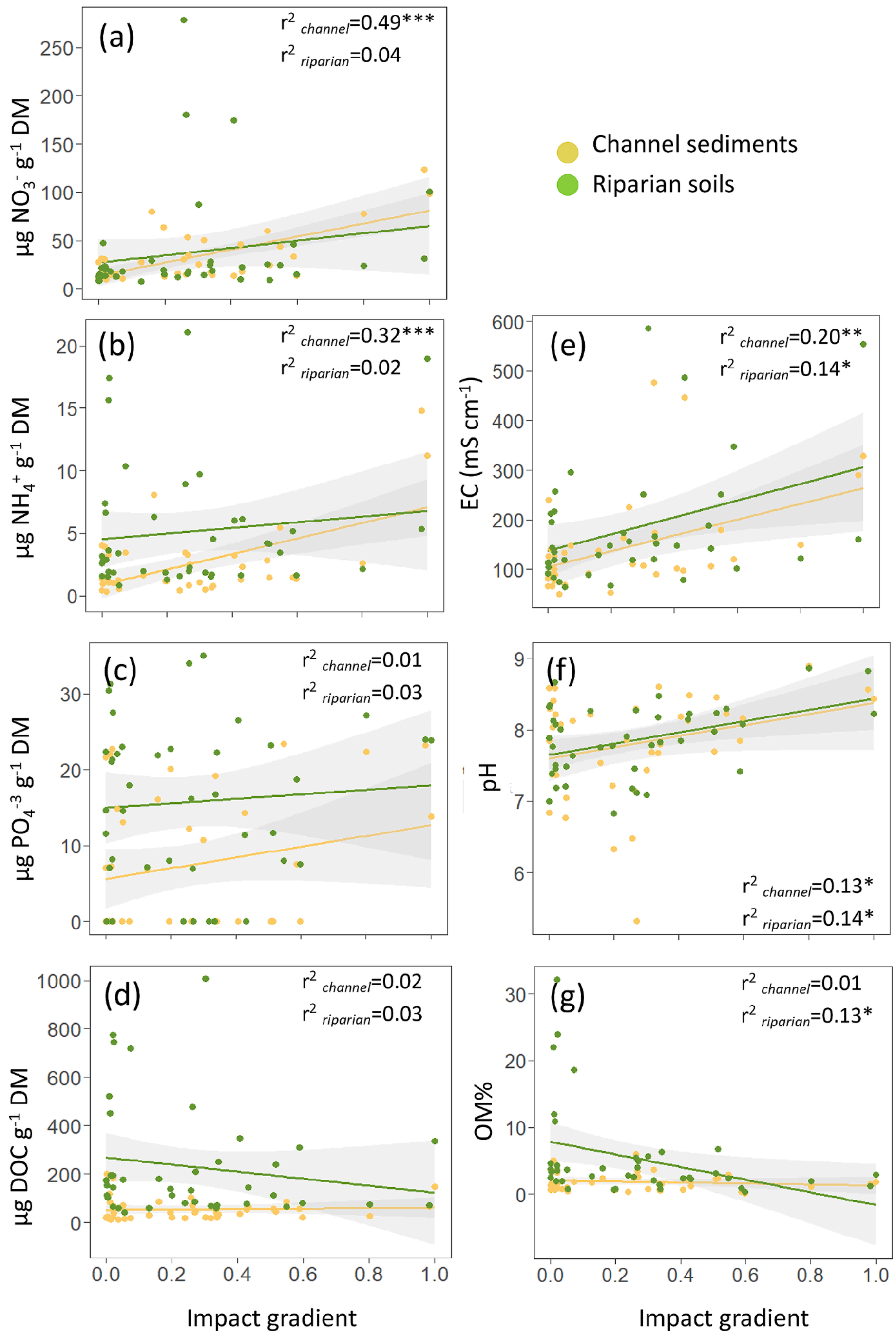

Fig. 5. Regression analysis between the physico-chemical parameters both in channel sediments and riparian soils, with the gradient increasing anthropogenic land use across study sites. Y axis is derived from the PCA – component 1. Values close to 0 represent natural land use while values close to 1 indicate anthropogenic land use. DOC: dissolved organic carbon, OM: organic matter, EC: electrical conductivity, DM: dry mass. The degree of significance of regression analysis is shown: $***p < 0.001$, $**p < 0.01$, $*p < 0.05$.

According to 29 criteria, the moderately- and highly- impacted sites (number of criteria < 21) included those sites affected by point-sources related to human activities, especially sites largely occupied by dry land farming and irrigated agriculture in uplands, which explain the excess of NO_3^- in sediments and soils. As widely described, agriculture is the main anthropogenic driver of NO_3^- levels due to fertilizer use which end up entering aquatic ecosystems (Vitousek et al., 2009). These outcomes agree with early works in some study sites where large amounts of NO_3^- in sediment leachates as a consequence of irrigated farming were observed (Arce et al., 2023). Additionally, evidence of livestocks around some sampling sites might also contribute to the high NO_3^- levels detected in sediments and soils.

Degraded riparian vegetation can be directly attributable to alterations in riparian zone and in river morphology, but the loss of natural land uses (Fernández et al., 2014) may explain the low %OM found in the highly impacted sites. Riparian zones represent essential compartments that trap elements from valley slopes and help control nutrient pollution in rivers (Naiman et al., 2005). In fact, poorly developed riparian vegetation coverage and interrupted connectivity with riverbed due to changes in morphology can also contribute to high NO_3^- levels, both within channel sediments and riparian soils.

The single-stressor approach, despite seeming oversimplified if compared with the multiple criteria methodology, allowed us to confirm the sensitivity of NO_3^- and OM content in response to the land use gradient that was condensed in the first PCA-axis. However, as the regression analysis indicated, the sensitivity of NO_3^- to quantitative changes in land use was only significant in channel sediments, while in the riparian zone this response was not found. Several reasons must be considered to explain this finding. One aspect relies in the fact that riparian zones embedded in catchments supporting high levels of farming may experience certain degradation levels which affect the amount of OM in soils, but it does not reflect the effects of runoff from fertilizer lands as effectively as channel sediments do. For instance, the prevalence of longitudinal transport over lateral inputs from agricultural areas may result in channel sediments storing larger amounts of NO_3^- than co-occurring riparian habitats. Another aspect to consider is the degree of vegetation coverage able to cope with the NO_3^- inputs. Unlike riparian zones, dry channel in some Mediterranean rivers are completely devoid from vegetation due to aridity and recurrent floods (Gómez et al., 2005). Indeed, this poor vegetation coverage under natural conditions within the riverbeds would explain the lack of significant variation in the case of OM in channel sediments both across the impact gradient of land use and among categories. Thus, these empty river channels, that have a low probability of retention from plant uptake, are sensitive to storing continued nutrient runoff from the uplands after rainfalls. Ultimately, NO_3^- levels in channel sediments with no factors that modulate their concentrations, such as vegetal elements, may reflect better the anthropogenic pressures linked to irrigated farming in the catchment if compared with heterogeneous riparian zones more susceptible to changes in the OM standing stocks.

In the case of EC, both methodological approaches confirmed the significant increase of this variable in the study sites with the exposition to human impacts. Salinization (i.e., high EC due to anthropogenic reasons) is recognized as one of the main problems in landscapes under intensive agriculture, especially in arid zones (Cañedo-Argüelles et al., 2013). In the Mediterranean area, fertilizer use and livestock waste in irrigated lands may contribute to increase EC in soils and receiving freshwaters (Skoulikidis, 2008; Acosta et al., 2011). Indeed, some of the study sites classified as highly impacted were surrounded by agricultural and urban areas showcasing potential to discharge into dry channels. However, dominant geology based on evaporitic rocks and marls rich in salts is known to provide high EC to draining fluvial ecosystems. In Southeast Spain, the presence of material rich in $CaCO_3$, NaCl, and SO_4Ca_2, associated to marine sediments from Triassic, results in an elevated natural conductivity of soils and water bodies (Gómez et al., 2005; Millán et al., 2011). In our study, 6 out 9 highly-impacted sites and 6 out 14 moderately −impacted sites presented marly and evaporitic geology. Thus, in practice, the interaction of dominant evaporitic geology with abundant irrigated agriculture in impacted sites hinders the potential of EC as an ecological indicator in our study region.

By applying the quantitative land-use information condensed in the first PCA-axis, we also observed significant responses of other physico-chemical parameters not reflected by the qualitative analyses which suggests that their variation may react better to impacts concerning the variable land use gradient alone. For instance, a positive response of NH_4^+ content in channel sediments across land use gradient was found. High NH_4^+ levels in sediments may be indicative of high loads of agricultural runoff to rivers where uptake from plants, normally scarce, is not enough to remove this nutrient. Urban effluents and presence of punctual shepherding may also explain high NH_4^+ stocks within the channels. Yet, the tight linkage between NO_3^- and NH_4^+ within the different biogeochemical routes of N cycle does not confirm the independence of NH_4^+ as useful indicator.

pH in riparian soils tended to slightly increase along the gradient of increasing anthropogenic land-use, indicating high values coupled to the urban and irrigated −agricultural uses. Considering human impacts, pH variability in soils and sediments is prone to be strongly modulated by activities linked to mining (Cuevas et al., 2023) as well agrochemicals in farming (Mandal et al., 2020). Using fertilizers based on NH amendment in farming is known to reduce soil pH (Clay et al., 1993). Despite having irrigated agriculture, in many study sites, our pH results showed an increase of this parameter along the anthropogenic land use gradient in riparian soils. However, depending on the agronomic management practices, pH can be punctually alkaline (e.g., when using organic compounds) (Sánchez-Navarro et al., 2022), which could support certain linkage between pH and irrigated agriculture presence. Nevertheless, given the varying geology in the catchment studied, pH in soils may vary strongly coupled to the nature of original materials, as previously described (Sánchez-Montoya et al., 2012). For instance, rocks rich in carbonates, reactive materials with buffer capacity and abundant in calcareous and marly catchments (Gómez et al., 2005), dominated in the highly impacted sites. On the contrary, most of the natural-use sites were metaphoric, thus supporting slightly lower pH. Since our monitoring region did not include soils clearly impacted by mining activities, we suspect that the relationship observed with pH was a result of the varying geology rather than an indication of consistent human pressure. Therefore, pH does not seem to act as accurate indicator of the ecological quality in our catchment study since we have to consider that geology is the primary factor determining either the acidity or alkalinity of soils.

In our study, PO_4^{3-} did not show a clear response to anthropogenic impact in any methodological approach, and concentrations only varied between habitats, with riparian soils showing generally higher values., The qualitative analysis which compared sites among categories, although not significant, showed that PO_4^{3-} concentrations tended to increase in channel sediments with moderate and high impact. Anthropogenic sources of P may include use of commercial fertilizers, manure from livestock production, and untreated or incompletely treated human sewage (Withers and Jarvie, 2008) which either enters directly to watercourses with rainfall driving runoff or in association with the eroding of upland soils (Mallin and Cahoon, 2020). Depending on a sediment/soil's characteristics and the biotic consortia present, dissolved P will be available and detectable. We consider that our leachates mainly represented the inorganic P, which can be easily released into the water rather than the P bound to soil particles like calcium compounds, clays, or the ones bound to organic matter (McDowell and Sharpley, 2001). Thus, incomplete leaching of the total pool of P from samples might explain a lack of robust linkage between this parameter and its impact in the study sites. Furthermore, larger levels of PO_4^{3-} in riparian soils, if compared with channel sediments, may be proof of the strong buffering capacity of potential pollution from upland areas before it reaches the riverbed. Alternative tests informing better on the presence of P as a pollutant (e.g., Soil P Sorption Saturation in Kleinman, 2017) could be included in monitoring programs, yet the high cost and running times of these analysis must also be considered.

Like PO_4^{3-}, the DOC leached from sediments and riparian soils does not appear to act as a suitable indicator. Previous findings in some rivers in the study region (Arce et al., 2023) showed that high DOC availability in river sediments may be the result of either natural and native vegetation, as well as of the drainage of crops (Royer and David, 2005; Lugato et al., 2014). DOC concentrations in channel sediments leachates for most streams studied were very low and hardly varied among categories. In riparian zones, despite not observing significant variations, the regression analysis showed a decreasing DOC concentrations trend as land-use impact gradient increased. Yet, this could be linked more to a degraded riparian vegetation, as indicated OM parameter, rather than as an influence of direct agricultural leachates or organic fertilization.

4.2. *The potential of physico-chemical parameters as ecological indicators for the dry phase: strength, caveats, and recommendations*

In the last years, freshwater research has encouraged efforts in the development of tools to evaluate the health of non-perennial rivers during the dry phase (Steward et al., 2011, 2018; Wilding et al., 2018). While intermittent rivers can be at some point assessed during the flow phase, ephemeral streams (and *dry rivers*), lacking water most of the time, are still devoid of ecosystem health indicators. Developing dry phase indicators in temporary rivers encompasses one of the current challenges for freshwater scientists and managers (Stubbington et al., 2019; Bonada et al., 2024). Trying to reduce existing gaps we have pointed out that channel sediments and the associated physico-chemical elements can be a suitable starting point to inform on the impacts present in temporary rivers.

The wet phase in temporary rivers is extremely variable in space and time and so is the physico-chemistry (Gómez et al., 2017), especially in ephemeral streams, water stagnates to occasionally form isolated pools with very particular physico-chemical characteristics (Gómez et al., 2017; Magand et al., 2020). Collecting any type of sample from a water compartment, which is only present after rainfall, is hard to program and it is unlikely to reflect the ecological state of the ecosystem. On the contrary, sediments represent long-term monitoring of specific elements that tend to accumulate, which makes their examination without need of existing surface water an asset.

Other advantages when applying physico-chemical elements arises if compared with other indicators. Some studies in Australian rivers have focused on examining and validating the potential use of biological communities, semi-aquatic, hyporheic, and terrestrial biological communities when flow vanishes (Leigh et al., 2013; Steward et al., 2018; Stubbington et al., 2019). Despite growing research, uncertainties about a clear response of terrestrial metrics to anthropogenic pressure, together with the high natural variability and monitoring difficulties, make using terrestrial organisms as bio indicators not a robust tool to date (Jupke et al., 2022). In this respect, using indicators based on biological assemblages that are pre-conditioned (with adaptation strategies) to frequency of flow regime to assess any type of fluvial dry section might give biased results. Comparably, the physico-chemistry conditions of riverbed sediments and soils appear to be less dependent on frequency of regime in general terms. In other words, a physico-chemical element might similarly respond to pressures regardless of the hydrological regime. In fact, we did not find any effect on the hydrology type (*intermittent* vs. *ephemeral*) to the parameters response to impact, yet this aspect still needs to be carefully examined in the future.

Besides, collecting sediment and soil samples is easier in terms of time, material, and further processing to achieve relatively fast results. In some instances, getting refined taxonomic resolution of biological assemblages and traits to establish reliable relationships between ecosystem health and the impacts is not an easy task. Furthermore, in highly impacted rivers with canalized channels and impervious substrate where placing invertebrate traps is impractical, a few grams of sediments may potentially capture the content of pollutants and other elements entering from the upland valleys.

According to the soil-health assessment, a parameter should satisfy several criteria in order to be used as an indicator. Such criteria mainly include being: i) relevant for the ecosystem's functions and services, ii) sensitive where change is noticeable and fast, not only reflecting merely short-term fluctuations, iii) practical, cheaply measurable within a short turnaround time and iv) informative for management and conservation (Lehmann et al., 2020). Similar arguments have been proposed when choosing an indicator to guarantee the successful monitoring, diagnosis, and management of non-perennial rivers (Boulton, 1999; Boulton et al., 2010). All the selected parameters in our study *a priori* meet these criteria. Yet, only NO_3^- appeared to accurately reflect the anthropogenic impact variable for the monitoring sites in our catchment.

The strong natural variability combined with methodological issues affected the robustness in the rest of parameter to act as suitable indicators and they represent caveats worth considering. Discriminating between natural and human variability contributes to the difficulties in the ecological assessment of temporary rivers, not only for the wet phase of but also for dry periods (Bonada et al., 2024). For instance, in our study, geology and aridity imposed natural features in EC and to the particulate organic matter content of soils. While high EC values are associated to high agricultural or urban effluents in many fluvial networks (Skoulikidis, 2008), the trajectory of this parameter along an increasing human impact gradient may be different in evaporitic catchments. Paradoxically, a "dilution effect" of high conductivity in surface waters due to freshwater infiltrations from diversion channel for irrigation lands has been described as one of the main anthropogenic impacts on rivers where elevated salinity is a natural feature (Gutiérrez-Cánovas et al., 2009). Yet, a similar effect with respect to the EC measured in sediments might not be so evident unless there is a strong, chronic impact. Nevertheless, changes in EC could be difficult to tie to specific environmental impacts in catchments where salinity is a primary source of natural variation. In our study, the geologic typology of the monitored rivers did not control the variation of the examined physico-chemical parameters (after checking through analysis of variance), but we cannot isolate its interactive effects with anthropogenic disturbances. Unfortunately, our data set does not allow us to check this influence since we could not account for monitoring sites representing a wide range of exposure to stressors within the same geologic typology. Indeed, it is common to find the highest impacted sites by agriculture under the same geology since certain soils can allow for better farming than others. Thus, a thoroughly evaluating the response of geology-dependent parameters, such as pH and EC, to human impact variables within the same typology should be considered in future studies.

Likewise, changes in OM stocks must be interpreted with caution in our study. Unlike humid and temperate catchments, where changes in OM content might suggest steady alterations in riparian vegetation (due to land use transformation or river morphology changes), low particulate organic matter content is associated with natural soils in Mediterranean arid regions and by extension to pristine streams without implying ecosystem degradation. On the contrary, it is not surprising to find dry channels completely invaded by *Phragmites australis* due to agricultural effluents rich in inorganic nutrients.

In the case of nutrients, shifts at small scale and in the biogeochemical processing linked to hydrological fluctuations should not be neglected. For instance, within the same temporary river, transitions from wet to drying periods have been suggested to punctually enrich sediments with high inorganic N that does not necessarily reflect pollution (Gómez et al., 2017; Arce et al., 2018). However, extrapolating this punctual increase in time and space to a global ecological state of a river ecosystem during its dry phase is unclear and requires more attention.

Nevertheless, *when* and *where* to collect samples are also methodological aspects to keep into consideration. While representative ecological states could be captured better during dry conditions, point-source impacts, such as those from disturbing activities located upstream of a dry section, may require specific monitoring (e.g., linked to activity periods). Also, if a riverbed is heterogeneous, dried riffles are not *a priori* locations where the physico-chemistry print is representative due to the rapid passage of upland elements during rewetting, which is a similar reason as to avoid sloped soils in the co-occurring riparian habitats. Thus, focusing on accumulation areas, such as residual dried pools, is beneficial to gather better information about the upland impacts.

Another detected caveat relies on the methodological resolution of the techniques used. For instance, in the case of extractable PO_4^{3-}, accurate information on its availability and anthropogenic source is known to depend on the type of protocol selected (Ortiz et al., 2023). To initially follow a low cost – single extraction procedure, we measured PO_4^{3-} leached to pure water (i.e., capable of being released from soils to water). Yet, despite accounting for sampling locations potentially being affected by eutrophication, this variable did not show a strong variation along the impact gradient probably due to strong P- bound to mineral particles as calcium carbonates surfaces of many study samples. Thus, based on different purposes and available lab facilities, deciding and adapting the extraction protocols is essential, especially when working in rivers embedded in agro-environmental networks where P harm is also expected (Ceulemans et al., 2014).

On the other hand, additional parameters measured in sediments, including polycyclic, heavy metals, pharmaceutical compounds, and other industrial pollutants, should also be included in the monitoring programs when dry riverbeds are used for dumping wastes (Skoulikidis, 2008; Tzoraki et al., 2015).

A crucial purpose for seeking potential indicators of ecosystem health is to establish reference values. Sediment reference conditions that represent un-impacted dry phases are also a priority to run complete management of temporary rivers (Stubbington et al., 2018; 2019). According to WFD, the ecological quality of a given river site is determined in relation to its reference condition (Bailey et al., 2014). In our study, NO_3^- in channel sediments varied among ecological status categories based on MRC, thus allowing us to propose a starting point for a threshold value (Sánchez-Montoya et al., 2012). As done in surface water, if we consider low impact sites, that is, those having natural state or near-natural conditions (MRC criteria > 22), a benchmark in NO_3^- levels could be in the 75th percentile (USEPA, 2000). The 75th percentile approach in this *low impact* category takes into account that this value is minimally associated to any type of impact (Sánchez-Montoya et al., 2012). Yet, the considerable source of natural variation of physico-chemical elements linked to climate, geology, etc., is a drawback for establishing reference values that can be easily transferable among regions where these environmental features vary. Properly selecting indicators and establishing reference conditions should be done according to the typology of catchments based on their main natural characteristics. For instance, certain indicators and their reference conditions could become comparable across regions sharing the same geologic typology. Focusing on this aspect in the future will help refine the use and suitability of physico-chemical elements across impact gradients.

Based on our experience, we recommend monitoring pilot programs that include both disturbed and undisturbed locations, as well as easy laboratory protocols to test potential suitable parameters as indicators. When possible, samples must be collected from accumulation areas and avoiding strong hills in riparian zones. Of course, monitoring programs and laboratory protocols should be adapted according to both the nature of the catchment being studied and its principal impacts. Furthermore, given the potential variability of some anthropogenic impacts (e.g., point-source pollution, Table 2), several campaigns in the pilot program are recommended to ensure success, yet technical capacity and funding levels must be weighed. To establish relationships with impacts, both the qualitative and quantitative approaches used in our study appear to be acceptable and complementary for Mediterranean catchments. Besides, while the land-use gradient approach seems appropriate to infer about the impacts at the ecosystem level (Truchy et al., 2022), the MRC method is suitable for establishing reference conditions (Sánchez-Montoya et al., 2012).

Finally, diversifying indicators will guarantee the success of a complete ecological assessment in any kind of ecosystem. Physico-chemical elements combined with biotic indicators could ideally be used since organisms may respond differently across the same impact gradient. For instance, a level of degradation for a dry section linked to a specific NO_3^- level could be classified differently if biotic elements are used. For instance, the impact of fertilizers is likely better reflected by sediment nutrients stocks than by changes in the invertebrate community. Developing biotic indicators to assess the health of the dry phase (e.g., terrestrial invertebrates) and their reference conditions are promising research arenas to advance in an integral management of non-perennial rivers (Bonada et al., 2024; Steward et al., 2018; Stubbington et al., 2019).

5. Conclusions

Increasing effort for a complete assessment of temporary rivers still relies on the lack of indicators on the dry phase (Bonada et al., 2024). Importantly, dry phase indicators are the only elements available to assess the health of ephemeral streams where the presence of a flow is merely anecdotic and does not likely represent their general state. Management policies biased toward considering only flowing phases will fail to preserve non-perennial streams from a holistic perspective when dry phase is neglected.

Our results from 41 monitoring sites provide for the first time insights of how physico-chemical elements (measured in leachates) may well reflect impact at local and catchment scale, as well as support their potential use to assess ecologically the dry phase. Although it is limited to Mediterranean rivers, the two methodological approaches used (qualitative criteria and land-use gradient) are complementary when one seeks the sensitiveness of physico-chemical elements to stressors of diverse nature and scale of influence.

Furthermore, our results highlight the importance of including riparian zones in assessment studies since they can also suffer from degradation and lose their buffering capacity to intercept pollutants from uplands (Pinay et al., 2024).

Despite all the selected parameters potentiality as indicators, NO_3^- content reliability responded to anthropogenic changes, mainly to those related to the presence of irrigated agriculture in the drainage catchment. Natural variability associated to geology and climatic conditions might hinder the use of EC, pH, and OM as trustworthy indicators in the study catchment.

The target parameters, protocols, sites, and sampling frequency must be adapted not only according to natural features potentially interfering (e.g., geology) but they should also consider the management priorities; for example, whether the focus is on 1) general state of conservation, 2) establishing reference conditions, or 3) monitoring the activity of a particular impact.

Providing dry phase indicators to properly evaluate temporary rivers, specifically ephemeral streams, will help catchment managers to explicitly include these ecosystems not only in the Iberian Peninsula (Magand et al., 2020, Bonada et al., 2024) but also in other regions when the increasing patters of climate and land use alteration that will drive longer extension and duration of dry conditions on riverine ecosystems are considered (Döll and Schmied, 2012).

CRediT authorship contribution statement

María Isabel Arce: Writing – review & editing, Writing – original draft, Methodology, Investigation, Conceptualization. **María Mar Sánchez-Montoya:** Writing – review & editing, Investigation, Funding acquisition, Conceptualization.

Declaration of competing interest

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

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Data availability

Data will be made available on request.

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