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Impacts of intensive agricultural irrigation and livestock farming on a semi-arid Mediterranean catchment

Emi Martín-Queller · David Moreno-Mateos · César Pedrocchi · Juan Cervantes · Gonzalo Martínez

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Abstract Irrigation return flows (IRF) are a major contributor of non-point source pollution to surface and groundwater. We evaluated the effects of irrigation on stream hydrochemistry in a Mediterranean semi-arid catchment (Flumen River, NE Spain). The Flumen River was separated into two zones based on the intensity of irrigation activities in the watershed. General linear models were used to compare the two zones. Relevant covariables (urban sewage, pig farming, and gypsum deposits in the basin) were quantified with the help of geographic information system techniques, accompanied by ground-truthing. High variability of the water quality parameters and temporal dynamics caused by irrigation were used to distinguish the two river reaches. Urban activity and livestock farming had a significant effect on water chemistry. An increase in the concentration of salts (240–541 $\mu S \cdot cm^{-1}$ more in winter) and nitrate (average concentrations increased from 8.5 to 20.8 $mg \cdot l^{-1}$ during irrigation months) was associated with a higher level of IRF. Those river reaches more strongly influenced by urban areas tended to have higher phosphorus (0.19–0.42 $mg \cdot l^{-1}$ more in winter) concentrations. These results support earlier research about the significant consequences to water quality of both urban expansion and intensive agricultural production in arid and semi-arid regions. Data also indicate that salinization of soils, subsoils, surface water, and groundwater can be an unwelcome result of the application of pig manure for fertilization (increase in sodium concentration in 77.9 to 138.6 $mg \cdot l^{-1}$).

Keywords Irrigation return flows · Water quality · Pig farming · Salinity

E. Martín-Queller (✉)
Department of Agroforestry Engineering,
School of Agricultural and Forestry Engineering,
University of Lleida, Av. Alcalde Rovira Roure, 191,
25198 Lleida, Spain
e-mail: emi@eagrof.udl.cat

D. Moreno-Mateos
Integrative Biology Department,
University of California,
Berkeley, CA 94720, USA

C. Pedrocchi · J. Cervantes · G. Martínez
Pyrenean Institute of Ecology—CSIC,
Av. Regimiento Galicia s/n, 22700 Jaca, Spain

Introduction

Irrigation return flows (IRFs) are not only considered to be the major contributor of non-point source pollution to surface and groundwater (Law and Skogerboe 1977; National Research Council 1996) but also have the potential for disrupting the natural hydrologic water balance of river basins (Haddeland et al. 2006). The major impacts from IRF to receiving water bodies include eutrophication, due to excessive nitrogen (N) and phosphorus (P) loading, and salinization, as a consequence of an increase in dissolved salts or total dissolved solids (TDS) (Bellot et al. 1989; Ghassemi et al. 1995; Burkhalter and Gates 2005; Causapé et al. 2006; Mhlanga et al. 2006). Irrigation-induced salinity affects about 20% to 25% of the world's irrigated lands (Ghassemi et al. 1995; National Research Council 1996; Postel 1999; Tanji and Kielen 2002). Harker (1983) identified an inverse relationship between river discharge and salinity in IRF, that is, the negative impacts of elevated salinity are higher when river discharge is lower. Much of the salt in IRF originates from salt-affected soil over which it flows, creating salt levels that exceed the maximum allowable salt concentrations for human consumption by an order of magnitude (2.5 $dS \cdot m^{-1}$; European Union 1998; Causapé et al. 2006).

Several studies have reported that IRF causes an increase of NO_3^- concentration in surface water bodies (Aragüés and Tanji 2003; Causapé et al. 2004a, 2006). The three major sources of nitrate found in IRF are leaching from croplands, land disposal of urban sewage, and wastes from concentrated animal feeding operations (Rodvang et al. 2004). The potential for NO_3^- leaching is a function of soil type, weather conditions, and the crop management system. In general, the higher the N application rate, the greater the amount of N that can be lost, since recovery of fertilizer-applied N by harvested crops averages about 50% and tends to be even lower under high N application rates (Aragüés and Tanji 2003). The P concentration in subsurface drainage water is typically low because this element tends to sorb strongly to soil particles under arid conditions. However, the P concentrations measured in many agricultural IRFs may be orders of magnitude above the soluble (0.01 mg $P \cdot l^{-1}$) and total (0.02 mg $P \cdot l^{-1}$) critical levels assumed to accelerate eutrophication in freshwater aquatic ecosystems (Aragüés and Tanji 2003). The contribution of P by municipal and industrial effluent to the total load in a receiving body depends on the degree of effluent treatment, the amount of dilution in the receiving water, and the morphological characteristics of the water body (Riemersma et al. 2006).

In the Ebro River basin (NE Spain), irrigated agriculture is the primary water consumer, with nearly 0.7 million irrigated hectares and a total water demand of 6,310 $Hm^3 \cdot year^{-1}$ (CHE 1996). The IRFs from these irrigated areas have contributed to the salinization of water courses in the basin (Alberto et al. 1986; Quílez 1998; Causapé et al. 2006), a process driven by the presence of vast salt reservoirs characteristic of the regional geology (Tedeschi et al. 2001). Many of the environmental issues associated with irrigation are similar, regardless of location. However, the "off-site" irrigation-induced pollution associated with elevated salinity depends on several factors that can be highly variable, including the hydrological characteristics of the irrigated land and substrata, the agricultural production technologies used, and the water supply and drainage conveyance systems (Causapé et al. 2004a). The goal of this study was to evaluate the effects of intense agricultural activity (irrigation and livestock farming) on the water quality of streams in a semi-arid catchment. The use of geographical information system (GIS) tools allowed us to consider other catchment variables that could also explain the broad-scale trends in stream water chemistry.

Methods

Study area

The Flumen River belongs to the Ebro River basin in northeast Spain (Fig. 1a). The river originates in mountains that are characterized by a calcareous substrate. After leaving the mountainous regions and descending to flatter plains, the river crosses quaternary glacis and alluvial fans overlying a tertiary structure composed of conglomerates, sandstones, and clays. Saline mudstones and gypsum deposits found in the lower part of the basin influence river water quality at lower reaches. The main sources of contamination in the Flumen catchment are located in the agricultural areas that have little topographical relief, especially in Monegros County (Huesca, Spain), with

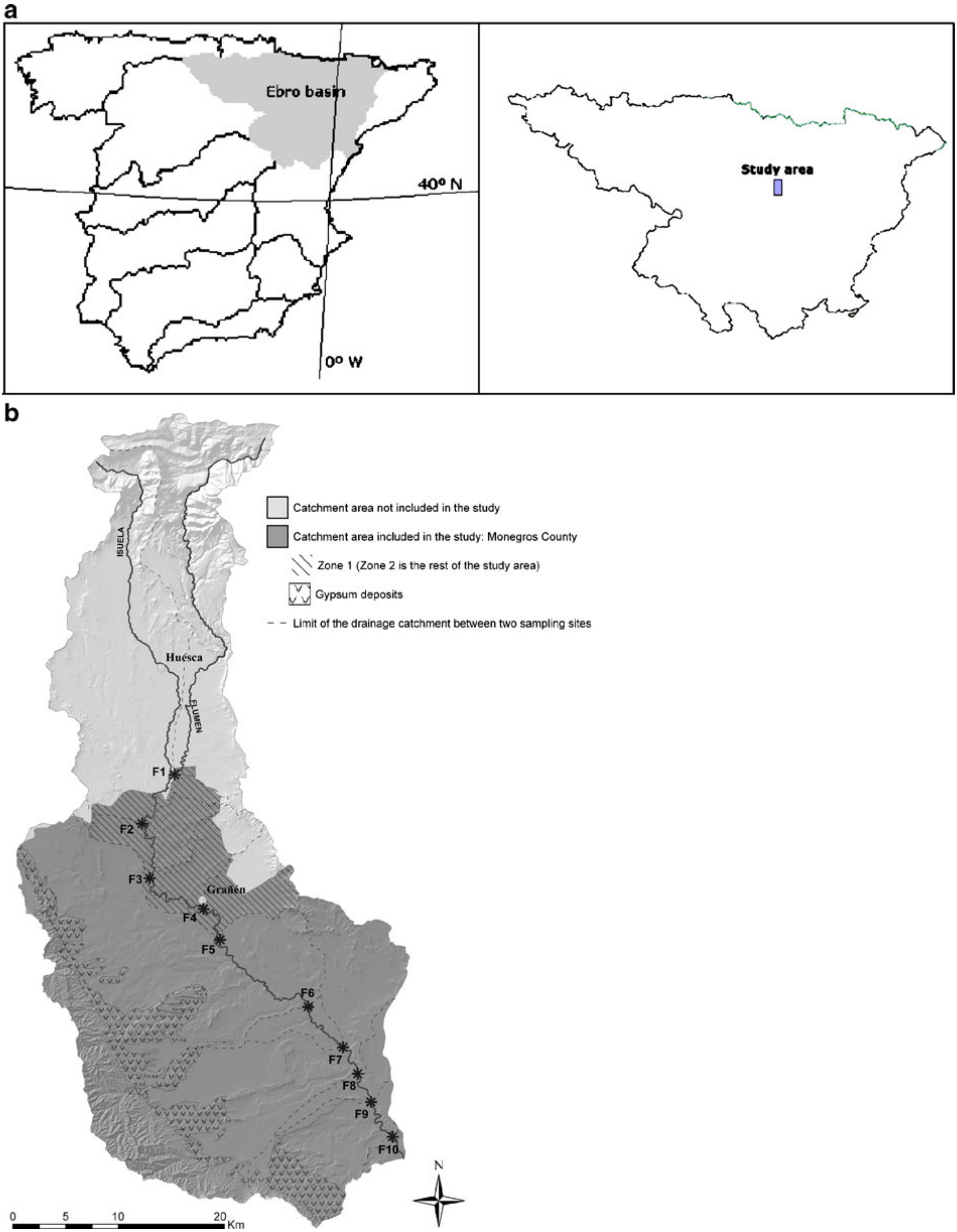

Fig. 1 The main catchments of the Spanish Iberian Peninsula. The location of the Flumen catchment in the Ebro basin (**a**) and the study area and water sampling points within the Flumen River catchment (**b**) are shown

a total surface area of about 1,000 km^2 and a maximum elevation of 830 m. The present study focuses on this catchment region, which is crossed by approximately 70 km of the middle to lower segment of the Flumen River (Fig. 1b).

The Flumen River is regulated by two reservoirs, Cienfuens and Santa María de Belsué (1 and 13 Hm^3, respectively), but irrigation is nevertheless the main source of fluvial perturbations. Approximately 44% of the study area is serviced by an irrigation system which is supplied by two channels that carry water from Pyrenean rivers. Flumen River water is not, therefore, diverted into the irrigation channel system in the district through which it flows. Irrigation effluent return occurs primarily in the lower 40 km of the river, and leads to changes in the hydrologic dynamics relative to the upper reaches (Fig. 2). The regional climate in Los Monegros is semi-arid and Mediterranean–continental. Average annual temperature is 14.5°C. Average annual precipitation is 400 mm (Pedrocchi 1998), although substantial temporal variability has been documented (Comín and Williams 1993).

The three most important urban areas in the Flumen basin have 50,000, 4,000, and 2,000 inhabitants (Huesca, Sariñena, and Grañén, respectively). Only Huesca has a wastewater treatment plant. Pig farming, which dominates all livestock husbandry in the region, accounted for 72% of animal production in 2003, 85% in 2005, and continues to grow (Anuario Estadístico Agrario de Aragón 2003–2005). In 2005, the density of pigs in Monegros County was 238 animals·km^{-2}. Pig production is important throughout the Flumen catchment in virtually all municipalities in the study area. The lower reaches of the basin have witnessed the most dramatic increase in intensive pig production operations, where densities have reached 5,900 animals·km^{-2} (Gobierno de Aragón, unpublished data).

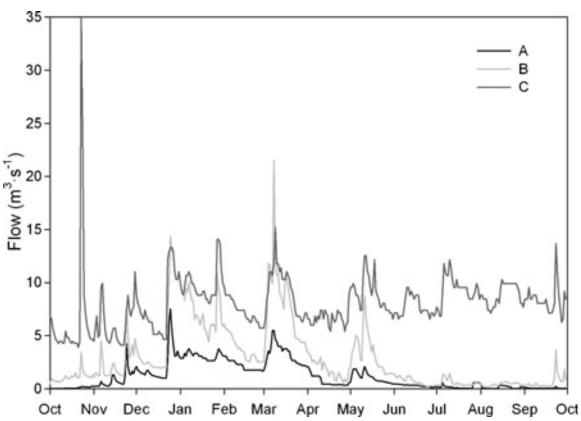

Fig. 2 Longitudinal changes in mean daily flow during the 2000–2001 hydrological year, measured at the entrance to the study area (**a**), the midpoint of the river (**b**), and the mouth of the river (**c**). Data facilitated by Confederación Hidrográfica del Ebro

Water sampling

Water chemistry samples were collected at ten sites located along the stream. Placement of the sampling sites (Fig. 1b) was based on the density of drainage channels discharging into the stream, which is an indicator of the influence of irrigation effluent. To evaluate the density of these drainage channels, the catchment drainage network within the Monegros region was drawn using the Hydrology tools in ArcGIS 9.1 (ESRI, Inc.) based on a digital elevation model at a resolution of 20 m and, then, refined using topographical maps (1:25,000) and aerial orthophotos (0.5 m resolution). Data were ground-truthed in the field along the entire stream length from January to June of 2005. The resulting cartographic representation of the irrigation infrastructures showed a higher density of drainage channels in the final river reaches, a reflection of the wider drainage area. Using this information, we divided the stream in the study area into two reaches that, while similar in length, were influenced to different degrees by irrigation. The first sampling site (F1) was placed outside of the intensive sampling zone, upstream from the main domestic and agricultural wastewater inputs, and thus was used as a reference site. The other nine stations were distributed along the 70-km river segment within the study area. The first four sampling sites (F2–F5) represent the first half of the stream, about 30 km long, called zone 1. It has a relatively small catchment area and very few incoming ditches. Downstream of the F5 sampling point, there is a substantial increase in IRF. This lower half of the river (F6–F10) was called zone 2.

Samples were collected from all sites on five different dates in 2005: 12 January, 16 March, 25 May, 4 July, and 13 October. One-liter samples collected from the river bank were preserved by adding 1 $cm^3 \cdot l^{-1}$ of trichloromethane, refrigerated (at 4°C), and analyzed in the laboratory within 1 week. Each sample was analyzed for SO_4^{2-} ($mg \cdot l^{-1}$), Na^+ ($mg \cdot l^{-1}$), mineral P ($mg \cdot l^{-1}$), NO_3^- ($mg \cdot l^{-1}$), NO_2^- ($mg \cdot l^{-1}$), and NH_4^+ ($mg \cdot l^{-1}$), using standard methods (APHA 1998). Electrical conductivity (EC) at 25°C ($\mu S \cdot cm$) was determined in the field using a conductivity meter.

Statistical analysis

Hierarchical cluster analysis was initially used to evaluate the laboratory results. The average linkage method and square Euclidean distance were used. Data was previously standardized. Given the seasonal nature of runoff from irrigated lands, the temporal component was taken into account by classifying each data point by site and sampling date. In order to assess the effect of IRF on water chemistry, annual mean values of every physical–chemical parameter were compared between the two study zones in the river. Significant statistical differences ($p \leq 0.05$) were evaluated through general linear models (GLM). Data from the two river zones were also compared separately during the irrigation season (May, July, and October) and the non-irrigation season (January and March). Some variables could not be transformed in order to meet the normality and homoskedasticity assumptions required by GLM; these variables were analyzed using the Mann–Whitney U non-parametric test.

A second analysis was conducted where, for each model, other covariables not included in the original models that had a significant effect on water quality were also considered. The goal of this exercise was to determine, with more certainty, whether the observed differences in water quality between zones 1 and 2 might be caused by IRF. Previous studies have identified significant relationships between water quality and watershed factors such as pig farms, gypsum substrates, and urban centers (Aragüés and Tanji 2003; Rodvang et al. 2004). Since, in the current study, these factors are unevenly distributed in the two study zones, they may be considered confounding factors in the interpretation of the effects of IRF. Larger urban areas (>5,000 inhabitants) in zone 1 mean greater impacts on those upstream sites, while more livestock farming and a greater number of surface gypsum deposits in zone 2 make these two factors more important in the lower reaches of the river. The possible effect of these variables was considered, provided that statistical assumptions allowed it, through GLM analyses. Three covariables were defined, representing the influence of pig farms (PIG), gypsum deposits (GYP), and urban areas (URB). Standard covariance analysis assumes homogeneity of the regression slopes in each zone. Nevertheless, covariables characterized by a rather diffuse discharge into the river may be strongly influenced by the volume of flow from the catchment basin (Heathwaite et al. 1996). It is probably unrealistic, therefore, to assume homogeneity in the two zones for variables such as pig farming. For this reason, models were based on separate regression slope estimates in zones 1 and 2 for those covariables exhibiting a more diffuse discharge to the river (GYP and PIG). The final model included only those covariables that had a significant effect on the particular parameter, after separately analyzing the influence of each covariable.

Depending on the scale of the study, its extent, and the spatial configuration of sampling sites, some physical–chemical water parameters may present spatial autocorrelation patterns (Peterson et al. 2006; Chang 2008). This situation brings into question the assumption of independence of residuals in linear regression analysis (Legendre and Legendre 1998). Some studies of stream networks have addressed this problem by using geostatistical models based on Euclidean distance (Chang 2008) or hydrologic distance measures (Peterson et al. 2006). However, the small sampling size used here does not allow estimating such spatial models. We included the spatial structure in the calculation of covariables to correct this bias and reduce its impact in the dependent variable residuals obtained through regression analysis, which were used later to compare mean values between the two zones. Statistical analysis was conducted with SPSS 14.0 for Windows (SPSS Inc., Chicago, IL, USA).

Geographical analysis

Variables associated with diffuse types of pollution (PIG and GYP) were weighted by catchment area, as a surrogate of the discharge volume, assuming that runoff variability due to altitudinal changes between catchments is insignificant compared to the magnitude of the volume used on the irrigation crops. Their influence at every survey site was weighted by its proportional catchment area (inclusive of all upstream sites), related to the total site catchment, and by its hydrologic distance to the particular study site in question. We used a spatial-weights matrix that has been applied in other studies when replacing Euclidean distances by hydrologic distances in the framework of spatial covariance models (Peterson et al. 2006, 2007). Defining a stream segment as a portion of a stream located between two sampling sites, we calculated the upstream watershed area for the downstream node of each segment in the stream network using ArcGIS 9.1 (ESRI, Inc.) and, after, refining the result map with a digitized topographic map (10-m contour interval) (Fig. 1b). At each survey site, the total upstream watershed area was calculated by summing the watershed area for the incoming stream segments. Thus, the proportional influence for each incoming segment on the segment directly downstream was calculated by dividing its watershed area by the total upstream watershed area. Finally, the influence of one site on another was equal to the product of the proportional influence of each segment found on the path between the two segments being analyzed; the multiplication of weights (between 0 and 1) ensured that the influence of any particular segment decreased as we moved upstream. A spatial-weights matrix that included the influence for all pairs of sites was obtained in this manner (Peterson et al. 2006, 2007; Ver Hoef et al. 2006).

To provide a quantitative variable that measures the influence of pig farms on the river, we used the number of pigs in every Monegros municipality in the Flumen basin in 2005 (pig density; Gobierno de Aragón, unpublished data). For every municipal area within a segment of the watershed, the proportional quantity of pigs was calculated and summed. The resulting covariate, PIG, at every survey point was equal to the sum of products from that and all upstream sites, between their influence, as quantified through the spatial-weights matrix, and the number of pigs in the corresponding segment watershed. The influence of gypsum substrate (GYP) was measured using data obtained during the GIS survey and considering all upstream gypsum coverage for each site. GYP was weighted in the same manner as PIG. The covariable URB for each survey site was calculated as the sum of all inhabitants of upstream urban areas of the study region (including the city of Huesca) divided by its straight-line distance to the particular site. ArcGIS 9.1 (ESRI Inc., Redlands, CA, USA) was used to compute the covariables.

Results

Differences in the measured water quality parameters between zones 1 and 2 were highest during the non-irrigation season (Fig. 3). These differences faded once irrigation began. During the irrigation season, differences between zones 1 and 2 were apparent only in the July samples. Water chemistry in zone 2 was generally more homogeneous than in zone 1.

Water quality varied longitudinally in the Flumen River (Fig. 4). There were significant differences in the mean values of all water quality parameters in both study zones, except for NO_2^- concentration. NO_3^- concentrations in the winter were not significantly different between river zones. However, once irrigation activities began and return flows were entering the river, the concentrations of NO_3^- in zone 2, which is more affected by these return flows, began to increase, whereas zone 1 mean concentrations remained unchanged or decreased somewhat. During irrigation months, the NO_3^- concentration was significantly higher (Table 1) in the final segment of the Flumen River (mean concentration in zones 1 and 2 were 8.5 and 20.8 $mg \cdot l^{-1}$, respectively). When irrigation was occurring, zone 2 samples had an average NO_3^- concentration that was 5.5 times that of the reference site, F1, upstream of the main pollution sources. Average NO_2^- concentrations were never different between zones, although average NO_2^- in zone 2 was as much as 15.9 times higher than at F1.

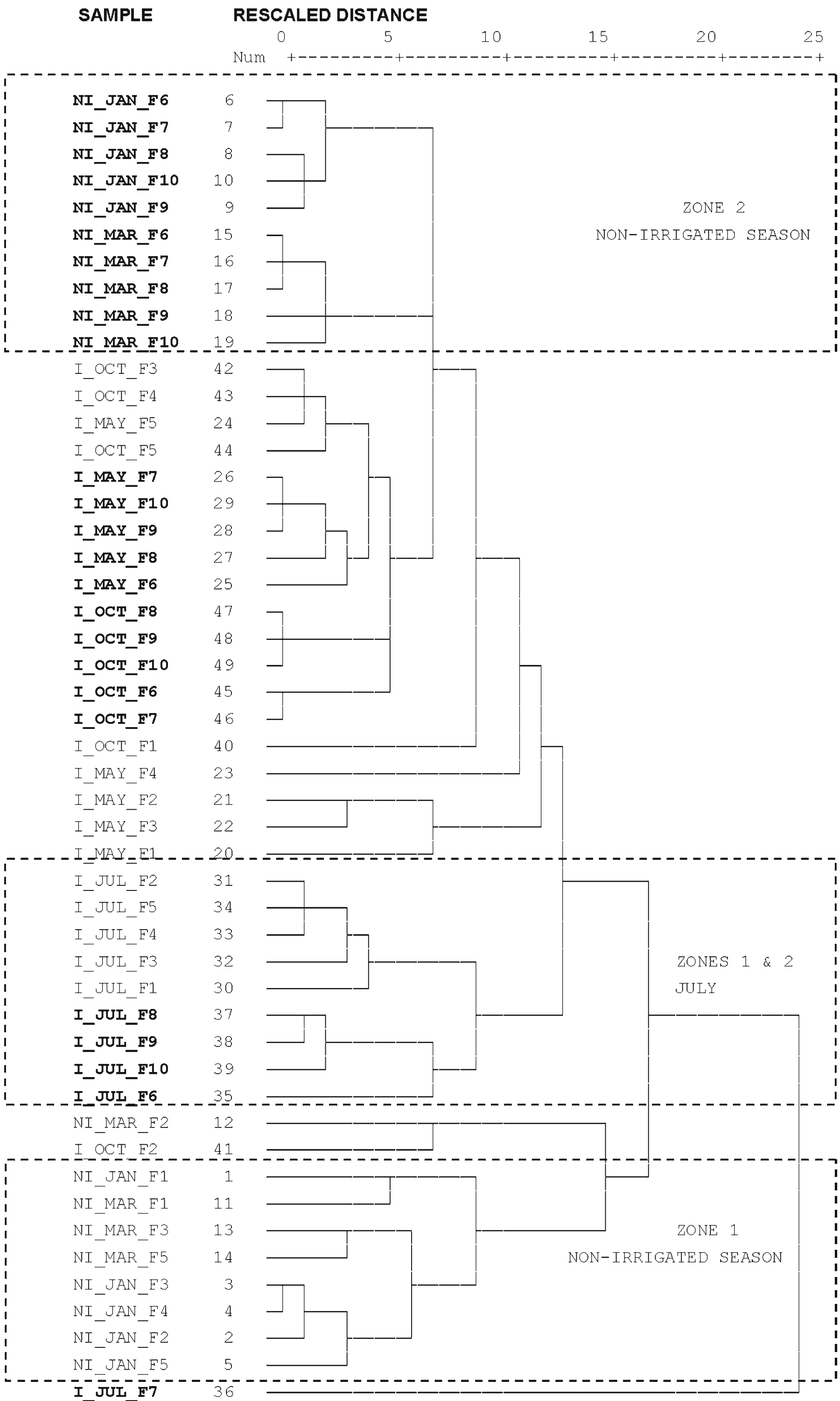

Fig. 3 Dendrogram classifying all samples according to the physical–chemical parameters assessed. Samples collected in zone 2 are highlighted in *bold*. Samples are characterized by *I* (irrigation season) and *NI* (non-irrigation season), the month when the sample was collected, and the name of the sampling site

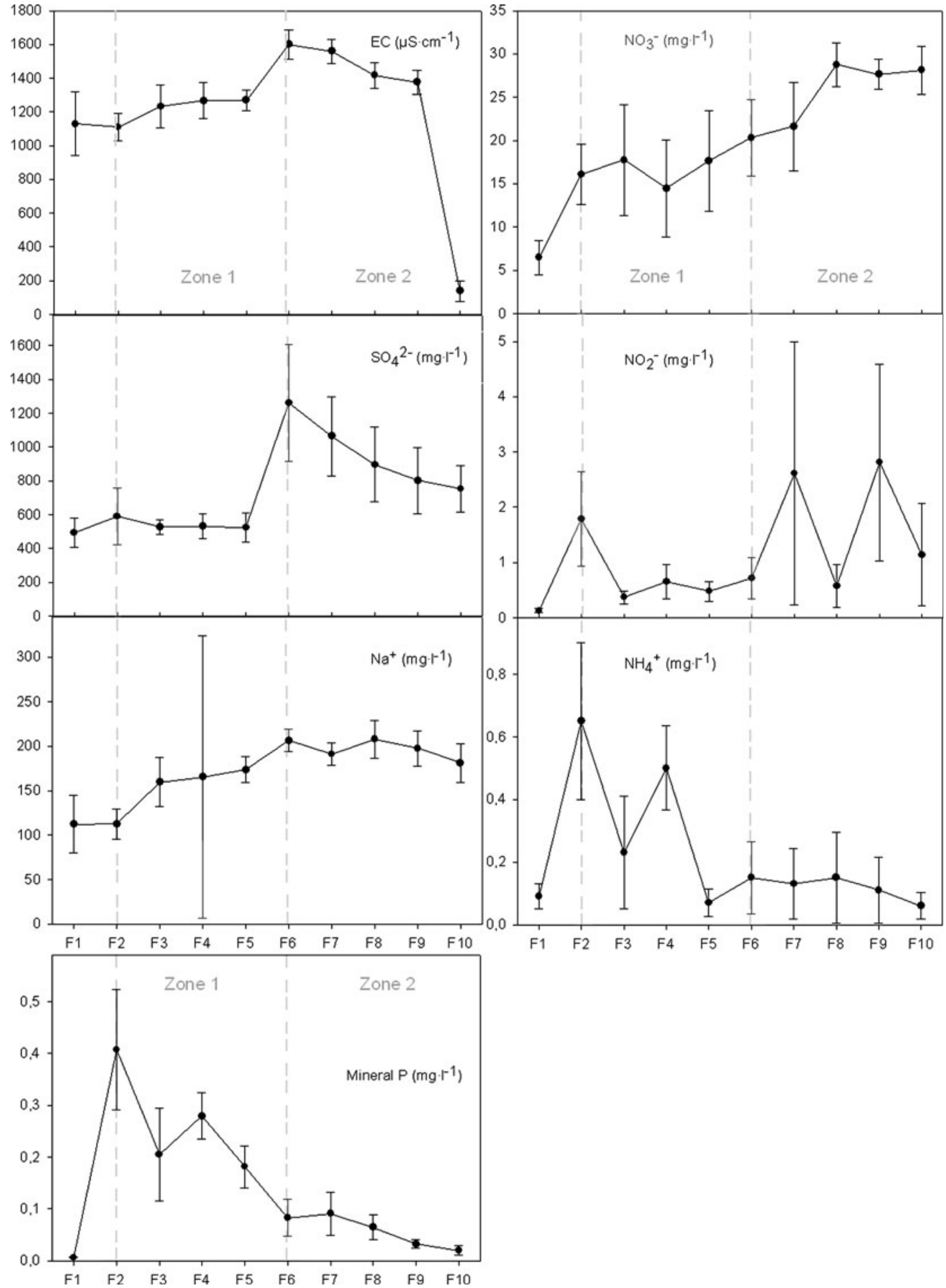

Fig. 4 Mean annual values of the seven water quality parameters measured seasonally at ten sampling points (F1–F10) in the Flumen River in 2005. *Bars* represent standard errors. Zones 1 and 2 are differentiated in the plots. F1 is a reference site outside of the intensive sampling area where the main pollution sources are found

Table 1 Results of the GLM and Mann–Whitney test comparing differences in the concentrations of several water quality parameters between zones 1 and 2 during the whole year (annual), the irrigation season, and the non-irrigation season in the Flumen River

	GLM				Mann–Whitney *U*		
	EC	SO_4^{2-}	Na^+	Min P	NO_2^-	NO_3^-	NH_4^+
Annual	18.19***	16.31***	10.92**	30.08***	n.s.	139.0*	128.0**
Non-irrigation	30.14***	68.64***	57.71***	30.54***	n.s.	n.s.	7.5**
Irrigation	n.s.	n.s.	n.s.	25.20***	n.s.	15.0***	n.s.

n.s. not significant
$^*p < 0.05$, $^{**}p < 0.01$, $^{***}p < 0.001$

The NH_4^+ concentration in Zone 1 was significantly higher in the non-irrigation months (Mann–Whitney U, $p < 0.01$). During the irrigation season, an increase in NH_4^+ concentrations in zone 2 was accompanied by NH_4^+ reductions in zone 1, resulting in overall homogenization of longitudinal concentrations. The average NH_4^+ concentration at sites F2 to F10 was higher than at the reference point (F1); indeed, when irrigation was occurring, the NH_4^+ concentration in zones 1 and 2 (0.193 and 0.196 mg·l^{-1}, respectively) was up to seven times the level of site F1 (0.026 mg·l^{-1}). The concentration of P was significantly higher in zone 1 throughout the whole year (between 0.19 and 0.42 mg·l^{-1} higher in winter; Table 1). Its relationship with the urban runoff covariable, though not on the annual basis, was significant on both the irrigation and non-irrigation seasons (Table 2). The elevated concentrations in zone 1 are illustrated in Fig. 5, where the outlier is from F2. During the winter, the mean mineral P concentration in zone 1 (0.402 mg·l^{-1}) was nearly two orders of magnitude higher than the concentration at the upstream reference site, F1.

Annual average Na^+ concentrations and Na^+ concentrations during the non-irrigation season were significantly higher (Table 1) in zone 2 relative to zone 1 (77.9 to 138.6 mg·l^{-1} higher). During the irrigation season, however, Na^+ was not significantly different between the zones. The GLM with covariables indicated that pig farming explained part of the Na^+ variability in the Flumen River (Table 2). The relationship between the PIG variable and Na^+ concentration was significant on both an annual basis and during the irrigation period. Nevertheless, it did not explain all of the observed variability, as statistically significant differences in annual Na^+ concentrations

Table 2 Results of the GLM (F statistic value and significance level) with covariables for those parameters found to be significant ($p \leq 0.05$)

		GLM		
		URB	PIG	ZONE
Mineral P	Annual	n.s.	n.s.	n.s.
	Non-irrigation	7.42*	n.s.	6.60*
	Irrigation	7.15*	n.s.	n.s
Na^+	Annual	n.s.	5.8*	254.69***
	Non-irrigation	n.s.	n.s.	n.s.
	Irrigation	n.s.	6.47*	159.43***

The statistical significance of the mean values in zones 1 and 2 (ZONE), after removing the effects associated with urban areas and pig farming (URB and PIG), is shown
$^*p < 0.05$, $^{**}p < 0.01$, $^{***}p < 0.001$

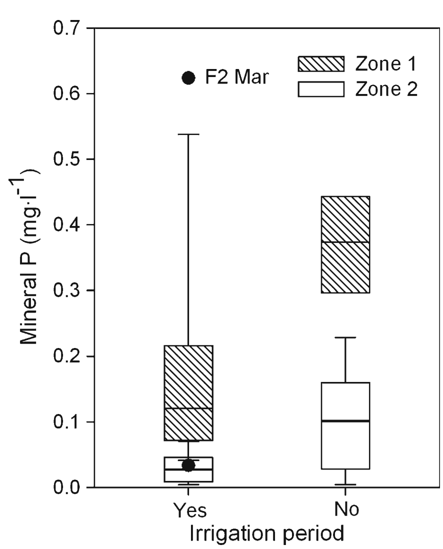

Fig. 5 Comparison of mineral P concentration between the two study zones during the irrigation and non-irrigation seasons

between the two river zones remained, even after the pig farming effect was discarded. In fact, during the irrigation season, pig farming appeared to promote homogeneous Na^+ concentrations along the stream. In the winter, EC in the lower river reaches was 240 to 541 $\mu S \cdot cm^{-1}$ higher than in the upper ones (Table 1). During the rest of the year, EC increased in zone 1, while EC in zone 2 did not demonstrably change, thus promoting longitudinal homogenization of the salt concentration (Table 1). The SO_4^{2-} concentration in the Flumen River during the winter was significantly higher in zone 2 relative to zone 1 (between 318 and 537 $mg \cdot l^{-1}$ more). Once irrigation effluents began to flow into the river, SO_4^{2-} concentrations declined in the lower reaches, resulting in no statistically significant differences between the study zones (Table 1).

Discussion

The Flumen River can be separated into two zones with different spatial and temporal hydrochemical characteristics. In the river reach that was included in zone 2, water chemistry was found to be relatively homogeneous. The differences in water chemistry between zones was very apparent during the winter months when IRF, and overall river discharge, was low. During the irrigation season (e.g., in July), when flows were elevated, there was greater homogeneity in water chemistry. The main sources of contamination to the Flumen River were the city of Huesca and the region of Monegros. Data collected upstream and downstream of these sources indicated very different water quality. Study results revealed that irrigation has two main effects on the agricultural runoff to the Flumen River. First, there is an increase in NO_3^- concentration, as shown in similar studies (Aragüés and Tanji 2003; Causapé et al. 2004a, 2006). Second, the irrigation water contributes to significant salinization of the river water, which also concurs with previous studies in agricultural areas with soils that are affected by natural salinity (Bellot et al. 1989; Ghassemi et al. 1995; Burkhalter and Gates 2005; Causapé et al. 2006; Mhlanga et al. 2006).

The NO_3^- concentration showed a longitudinal gradient in the river (Fig. 4), with lowest values in the upper reaches (F1 = 6.5 $mg \cdot l^{-1}$) and maximum values in the lower reaches (F10 = 27.7 $mg \cdot l^{-1}$). Causapé et al. (2004b) reported similar results, with NO_3^- concentrations of 4 $mg \cdot l^{-1}$ and 32 $mg \cdot l^{-1}$ in water collected upgradient and downgradient, respectively, of an irrigation-affected area in the Ebro basin. Despite the dilution that occurs from irrigation effluents, NO_3^- concentrations rose in the final reach of the river in the spring and summer. This is probably due to the leaching of nitrogen fertilizers, which are applied in April for sown crops, and in June for maize (Bellot et al. 1989; Isidoro 1999; Causapé 2002). The use of non-parametric statistical analyses did not allow for the inclusion of covariables in the models. Nevertheless, an increase in NO_3^- concentration in rivers receiving irrigation runoff in semi-arid conditions has been reported by several researchers (Aragüés and Tanji 2003; Causapé et al. 2004a). Isidoro and Aragüés (2007) found the highest levels of NO_3^- (>20 $mg \cdot l^{-1}$) in the Ebro basin were in those rivers receiving irrigation effluents, including the Flumen River in its lower reaches. Rodvang et al. (2004) related an increase of NO_3^- concentrations in surface and ground water to the land application of excessive quantities of manure as fertilizer. They estimated that NO_3^- concentrations in a Canadian stream increased by at least a factor of 4.3 in areas with high densities of livestock farming operations.

Although the increase of P compounds in surface waters as a consequence of IRF has been commonly reported (Riemersma et al. 2006), in the present study, the concentration of P was significantly higher during the whole year in stream reaches (i.e., zone 1) that were less affected by IRF. The results of the covariable analyses suggest that urban pollution, rather than IRF, is the primary factor influencing P dynamics. Urban discharge occurs throughout the year, during both irrigation and non-irrigation seasons, and thus can account for the punctual P pollution. This also could explain why the P concentration in F2, situated immediately downstream the sewage treatment plant of the largest city (Huesca), was an outlier in Fig. 5.

Study results indicate that irrigation did not affect NO_2^- concentration, and its influence on mineral P levels was secondary to the impact of urban sewage. Isidoro and Aragüés (2007) also suggested that, despite the high P concentration in streams directly affected by irrigation, the main source of P and NH_4^+ pollution is domestic urban effluent. Because of the non-parametric tests used in the study analyses, we could not determine if IRF or urban runoff directly affected NH_4^+ variability. However, the influence of IRF is likely to be significantly overshadowed by the impact of urban discharge and livestock farming.

The longitudinal enrichment of salts (EC) observed in winter along the catchment occurs naturally in all streams and originates in the discharge from the drainage network. However, soils in the study area were affected by salinization (Pedrocchi 1998) which, consequently, caused higher salt concentrations in the stream, reaching EC values as high as 1,784 $\mu S \cdot cm^{-1}$ (Fig. 4). While the highest in-stream salinity was observed during the non-irrigation season, it was during the irrigation season when salt export was highest. This phenomenon has been described previously under similar environmental and soil conditions (Causapé et al. 2004b). The lowest EC values were measured at the entrance of the irrigation district during the irrigation season (seasonal average = 450 $\mu S \cdot cm^{-1}$); maximum EC values were measured at the river outlet during the non-irrigation season (seasonal average = 1,550 $\mu S \cdot cm^{-1}$). During the irrigation season, therefore, the mass of salt exported from the irrigated watershed into the river system is at its peak, but dilution keeps the actual concentration lower. During the winter, as water levels drop, the high amounts of salt received during the irrigation season continue to influence water chemistry and result in the highest salt concentrations, as evidenced by maximum EC values.

Variations in the level of Na^+ can be partially explained by the same irrigation practices that impact EC and SO_4^{2-}. The increase in Na^+ concentration in the second reach of the river, indicated by the analysis of covariance, was also due to the use of manure from hog production operations in Monegros as natural fertilizer in croplands. Yao et al. (2007) found that manure generated in pig farms contained high levels of soluble salts. Major salt components were SO_4^{2-}, NaCl, and KCl, which are commonly added to the animal diet. They found that the Na^+ concentration increased linearly with the rate at which chicken and pigeon manure was applied to croplands (Yao et al. 2007). Díez et al. (2001) also found that higher application rates of pig slurry significantly increased Na^+ concentrations in the soil, which suggests that Na^+ from the manure leaches into the soil and, possibly, into groundwater. These data indicate that long-term application of manure to cropland can have significant adverse impacts on soil quality and subsequent effects on both surface and subsurface water as a result of salt leaching. In the Monegros region, with the recent proliferation of intensive hog production operations, large quantities of manure are generated every year. It should be noted that legislation regulating pig manure management in Spain (RD 324/2000) only considers NO_3^- concentration when establishing restrictions on animal waste application to croplands. Finally, the pattern of SO_4^{2-} concentration in the Flumen River was similar to EC; there did not appear to be any significant effects from the gypsum deposits in the catchment.

Conclusions

The spread of irrigation systems in the Monegros region is a major cause of the increase in salinity and NO_3^- concentrations in the Mediterranean semi-arid catchment. Though basin geology contributes to higher salinity in the middle reaches of the Flumen River, excessive salinity and nutrients are a more acute problem in the lower reaches where agricultural uses exacerbate the problem. As a result, the river in the Monegros region can be divided into two zones based on physical–chemical characteristics and variations in temporal dynamics. The considerable increase in discharge that occurs during the irrigation season helps to ameliorate irrigation-related water chemistry effects through a higher amount of dilution. During low flow, when irrigation drainage is minimal (winter), the impact of salts from IRF becomes more significant, and EC, SO_4^{2-}, and Na^+ concentrations increase downstream.

In this study, the effect of intensive agriculture, including the application of chemical fertilizers, was apparent in higher NO_3^- concentrations. The increasing number of hog production operations in the region has aggravated the problem because of the high levels of salt in pig manure which, at least in the current study, contributed to a higher in-stream Na^+ concentration.

Urban sewage, which is untreated in most cases, is also responsible for degradation of water quality in the river. The variable most affected by urban sewage was mineral P.

The effects of irrigation on the water quality of a Mediterranean river described in this paper reveals a critical need to adopt appropriate measures to promote the recovery of the lotic system. Such measures must consider the applicable restrictions for domestic, industrial, and agricultural use of downstream water. There is also an urgent need to conduct an in-depth study of the effects of intensive hog farming on soil, surface, and subsurface salinity in semi-arid areas. The acquisition of this information is made even more important by the high growth rate of high-volume hog operations in many Mediterranean areas. We propose the application of some specific measures, such as the construction of sewage treatment plants for small urban areas (<500 inhabitants), or the use of wetlands to reduce the concentration of nutrients (N and P) before effluent is discharged to a receiving water. Finally, it is also crucial that irrigation efficiency be improved in order to reduce the quantity of salts and NO_3^- exported to streams and rivers (Causapé et al. 2006).

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