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Bravo-Linares, C



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# Basin scale sources of siltation in a contaminated hydropower reservoir

Claudio Bravo-Linares[a]; Luis Ovando-Fuentealba[a,b]; Enrique Muñoz-Arcos[b];; Jessica L. Kitch[b]; Geoffrey E. Millward[b]; Ricardo López-Gajardo[a]; Marcela Cañoles-Zambrano[a]; Alfredo Del Valle[c]; Claire Kelly[b] and William H. Blake[b*].

[a] Universidad Austral de Chile, Facultad de Ciencias, Instituto de Ciencias Químicas. Independencia 631, Valdivia, Chile.

[b] School of Geography, Earth and Environmental Sciences, University of Plymouth, PL4 8AA, UK.

[c] Foundation for Participatory Innovation, Chile.

* Corresponding author: william.blake@plymouth.ac.uk

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## Abstract

Siltation and the loss of hydropower reservoir capacity is a global challenge with a predicted 26% loss of storage at the global scale by 2050. Like in many other Latin American contexts, soil erosion constitutes one of the most significant water pollution problems in Chile with serious siltation consequences downstream. Identifying the sources and drivers affecting hydropower siltation and water pollution is a critical need to inform adaptation and mitigation strategies especially in the context of changing climate regimes e.g. rainfall patterns. We investigated, at basin scale, the main sources of sediments delivered to one of the largest hydropower reservoirs in South America using a spatio-temporal geochemical fingerprinting approach. Mining activities contributed equivalent to 9% of total recent sediment deposited in the hydropower lake with notable concentrations of sediment-associated pollutants e.g. Cu and Mo in bed sediment between the mine tributary and the reservoir sediment column. Agricultural sources represented ca. 60% of sediment input wherein livestock production and agriculture promoted the input of phosphorus to the lake. Evaluation of the lake sediment column against the tributary network showed that the tributary associated with both dominant anthropogenic activities (mining and agriculture) contributed substantially more sediment, but sources varied through time: mining activities have reduced in proportional contribution since dam construction and proportional inputs from agriculture have increased in recent years, mainly promoted by recent conversion of steep lands from native vegetation to agriculture. Siltation of major hydropower basins presents a global challenge exemplified by the Rapel basin. The specific challenges faced here highlight the urgent need for co-design of evidence-led, context-specific solutions that address the interplay of drivers both within and without the basin and its communities, enhancing the social acceptability of sediment management strategies to support the sustainability of clean, hydropower energy production.

**Keywords:** Water-Energy-Food Security, Basin Scale, Sediments, Fingerprinting, Contaminants.

**Highlights**

- Geochemistry allows apportionment of sediment sources impacting hydropower dams
- Conversion of steep land from native vegetation to agriculture promotes soil erosion
- Mining is the key source of particulate Cu and Mo in this Andean river basin
- Sediment-associated Cu, Mo, As and P dominate reservoir contamination
- This study exemplifies global sediment management challenges in reservoirs

## CRediT:

**Claudio Bravo-Linares**: Conceptualisation; Data curation; Formal analysis; Funding acquisition; Investigation; Methodology; Project administration; Resources; Software; Supervision; Validation; Visualization; Writing-original draft; Writing-review & editing. **Ovando-Fuentealba, Luis**: Data curation, Formal analysis, Methodology, Software. **Muñoz-Arcos, Enrique**: Data curation, Formal analysis, Methodology, Software. **Kitch, Jessica L**: Data curation, Methodology, Software. **Millward, Geoffrey E.:** Formal analysis**,** Writing-review & editing. **López-Gajardo, Ricardo**: Data curation, Formal analysis, Methodology, Software. **Cañoles, Marcela**: Data curation, Formal analysis, Methodology, Software. **Del Valle, Alfredo**: Writing-review & editing. **Kelly, Claire**: Writing-review & editing. **Blake, H. William**: Conceptualization; Formal analysis; Funding acquisition; Investigation; Methodology; Project administration; Resources; Writing-review & editing.

# 1 Introduction

Siltation and the loss of hydropower reservoir capacity is a global challenge and recent analysis predicts a worldwide storage loss of 26% by 2050 (Perera et al., 2023). While there has been much analysis of storage capacity and remedial efforts to manage siltation at the reservoir scale e.g. flushing and dredging, there remain questions about the dynamics of sediment source and delivery at the basin scale to tackle the root cause of the problem. This study explores how investigation of sediment source dynamics can inform sustainable land management decisions using the evidence from an exemplar hydropower basin in Chile.

Land degradation is a global problem affecting more than 2 billion people (UNCCD and World Bank, 2016). Soil erosion, promoted by multiple factors, has negative impacts on the future of soil fertility, water quality, food safety and power generation (Dercon et al., 2012; Li and Fang, 2016; Owens, 2020). Future land-management decisions and water quality assessments in reservoirs require quantification of the anthropogenic amplification of runoff and soil erosion processes, so that hotspots of soil erosion can be targeted and controlled by mitigation measures (Collins et al., 2020; Poesen, 2018; Walling, 2013). Erosion processes not only lead to loss of critical soil resources, but also the associated transport of contaminants that are bound to soil particles, such as pesticides, heavy metals, radionuclides, and phosphorus, that may have detrimental impacts on the healthy functioning of water bodies such as rivers, wetlands, lakes and estuarine ecosystems (Rodgers et al., 2020; Wharton et al., 2017; Wohl, 2015). Fine sediment (< 63 µm) has the potential to travel long distances from source to sink and impact local ecology by reducing water clarity and light penetration (Davies-Colley and Smith, 2001; Thrush et al., 2004). The identification of sediment sources is therefore essential to implement focused mitigation actions. Given the widespread impacts of soil erosion and siltation on the food-water-energy-security nexus (Blake et al., 2021) and the increasingly recognition of multiple co-benefits of natural capital, basin-wide solutions integrating upstream soil erosion and downstream sediment loading processes are required to develop sustainable management plans.

The impact of agricultural and forestry operations on sediment yields and consequent water pollution from different catchments, and the effectiveness of potential mitigation measures are traditionally assessed by measuring the sediment yield from the outlets of small experimental catchments that are subject to different levels of disturbance/management practices (Stenberg et al., 2015). Whilst this experimental approach can provide an effective basis for assessing the changes in sediment yields, it cannot provide specific information on the sources of eroded soils, especially when attempting to support collective and integrated decision making at the basin scale. Such information is essential to design and implement effective sediment control strategies and to target mitigation measures in the most cost-effective way. The sediment fingerprinting technique has emerged as a valuable tool in this regard, and it has been applied in a variety of landscapes and settings (Collins et al., 2020; Owens et al., 2016; Walling, 2013). This technique characterises potential sources according to their chemical and physical properties. These properties are also measured in sediment samples collected from areas of accumulation/receiving environments (e.g. dams, rivers, wetlands, lakes, estuaries). Then, potential sources and mixtures data are analysed, and subsequently mixing models are used to estimate the relative proportion of soils in a mixture from the various sources in a contemporary and historical perspective (Collins et al., 2017; Walling, 2013). The tracer properties used must be measurable in both sources and mixtures, be representative of their sources and must also have conservative signatures from their source, during transport and deposition (Collins et al., 2017; Owens et al., 2016; Walling, 2013). Several tracer properties have been used to this purpose: major and trace elements (Blake et al., 2018a; Kitch et al., 2019; Muñoz-Arcos et al., 2021), Fallout Radionuclides (FRNs) (Schuller et al., 2013), mineral magnetism (Blake et al., 2006; Gaspar et al., 2019), colour coefficients (Martinez-Carreras et al., 2010) and compound-specific stable isotopes (Blake et al., 2012; Bravo-Linares et al., 2018). These sediment tracing techniques have been used to determine both contemporary and historical sources of eroded soils and across a range of spatial scales (Bravo-Linares et al., 2020; Wynants et al., 2020). However, its application in large river basins, using a tributary tracing approach, has received little attention to date. This basin scale is critically important when tackling the food-water-energy-environment challenge, especially in the context of major silted hydropower basin projects in the global South (Zhang et al., 2018). The present study aims to close this critical gap.

Similarly, in many other Latin American contexts, soil erosion constitutes one of the most significant water pollution problems in Chile. A recent study concluded that 36.8 million hectares of land is subject to some degree of soil degradation, an equivalent to 49% of the national territory and a further ~7 million hectares is subject to severe soil erosion. On the other hand, agriculture in Chile has intensified in the past decades causing changes in the physical properties of soils and consequently creating a potential risk for water erosion (Fleige et al., 2016). Increased erosion can thus augment sediment loads in waterbodies, especially in hydropower reservoirs, affecting water quality, energy production and inducing reservoir siltation (Bronstert et al., 2014). In Chile there are 60 dams, which are widely distributed throughout the national territory. This indeed, has the potential to affect the power generation capacity, as the dams are losing water storage capacity with increasing sedimentation rates (Kondolf et al., 2014). Particularly in central Chile, the area has eroded soils (classified into categories of light, moderate, severe and very severe erosion) of 861 thousand hectares, which represents 52.5% of the regional surface.

There is an evident lack of understanding when integrating, in both time and space, impacts of soil erosion processes from upstream sources to sediment delivery downstream in large scale hydropower systems. This, of course, brings the need of assessing the sources that are producing siltation in dams and thereby water pollution through a geographical lens. Here we hypothesised that a tributary tracing fingerprinting approach can be used to determine the dominant sediment sources for water reservoir siltation and their associated contaminants at the basin scale as well as the social and environmental drivers. In doing so, we also addressed the distribution, both spatial and temporal, and potential impacts of sediment associated contaminants in industrially impacted rivers of similar size across the world.

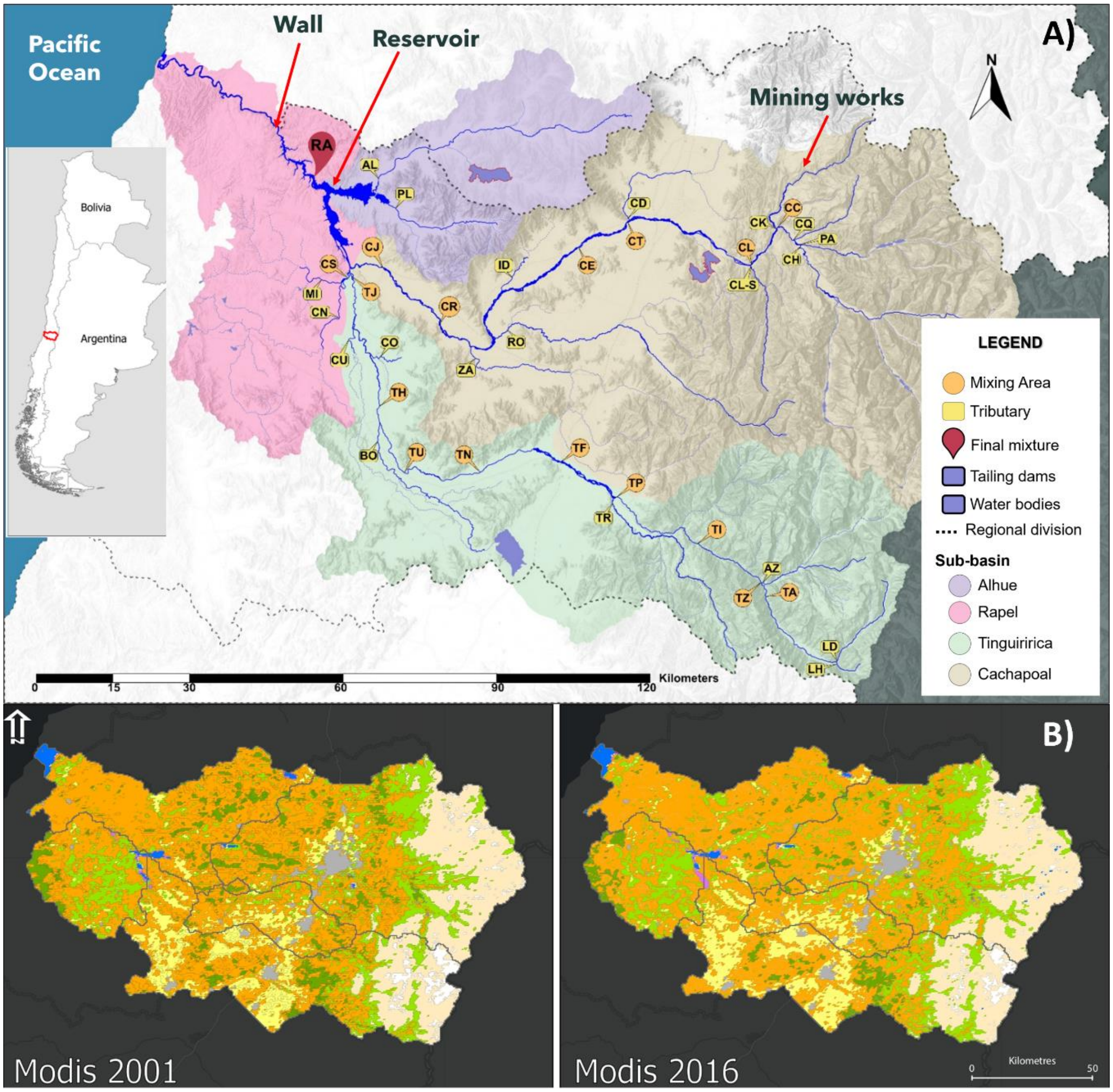
# 2 Materials and Methods

## *2.1 Study Site*

This study is based in the Rapel basin, located in Central Chile (VI Region), between the parallels 33°53'and 35°01' south latitude, and with a surface area of 13,695 $km^2$ (Fig. 1). The annual rain fall by the coast is 638 mm $yr^{-1}$, 406 mm $yr^{-1}$ in the central valley and around 686 mm $yr^{-1}$ in the Andes region. Of particular significance, during the winter of 2023 (July and August) normal rainfall patterns changed dramatically and two massive flooding events occurred during this period. Here, rainfall concentrated in the Andes produced dramatic changes in river flow leading to floods that left urban and agricultural areas with substantial amounts of deposited material. Average temperatures for the coast are 14°C and 9.6°C by the mountains. The catchment is fed by two main river systems (Cachapoal and Tinguiririca). Cachapoal River is characterised by a nivo-pluvial regime, with an average flow rate of 172 $m^3 s^{-1}$ during winter and 98 $m^3 s^{-1}$ during summer, while the Tinguiririca River has a primarily pluvial regime with average flow rates of 125 $m^3 s^{-1}$ during winter and 46 $m^3 s^{-1}$ during summer. Estimated sediment loads, for the Cachapoal and Tinguiririca rivers are approximately 1.11

and 0.71 million tons/year, respectively (Pepin et al., 2010). The confluence of these two rivers forms the Rapel River, which flows into an artificial lake system (Rapel Lake) impounded by a dam (built in 1968) with an initial water storage capacity of 695 million $m^3$ and power generation capacity of 350,000 kW.

In terms of socio-economic activities, the catchment has been used for hydropower generation since 1968. Upstream, intensive agricultural and forestry activities, mining industry, sand and gravel extraction, land use changes of hilly zones and recreational uses have severely affected the water storing capacity of the dam (36% loss by 2010, data provided by national energy company, ENEL). It has been reported that to 2010, 159 million $m^3$ of coarse material have reached the delta, and another 18 million $m^3$ of fine sediment have reached the dam's wall (Lecaros Sánchez, 2011); the latter being the input that provokes greater siltation in the reservoir and most severe consequences for hydropower generation. These processes have arguably been enhanced by natural erosion produced by glacial retreat, and the consequences can be seen in the lake environmental quality. Alongside the sediment load issues, water quality has also been affected by runoff from livestock activities resulted in a significant input of particulate phosphorous, generating eutrophication and associated issues (Vila et al., 2000). Moreover, urban development indicated an increase in organic matter and boron (B) (Lacassie and Ruiz-Del-Solar, 2021).




Rapel Catchment: Changes in land cover 2001 - 2016 **C)**

| Land use | Total area 2001 (ha) | Total area 2016 (ha) | Percentage Change |
|---|---|---|---|
| Water | 9360 | 8613 | -7.99 |
| Barren | 229993 | 237282 | 3.17 |
| Built up area | 30694 | 30919 | 0.73 |
| Cropland | 129014 | 137377 | 6.48 |
| Forest | 95570 | 61728 | -35.41 |
| Grassland | 220362 | 254011 | 15.27 |
| Savanna and scrub | 684165 | 675252 | -1.3 |
| Snow and ice | 26647 | 19293 | -27.6 |
| Wetland | 2828 | 4268 | 50.95 |

**Fig. 1**.A) Map of the Rapel Basin location in central Chile indicating their sub-basins and sediment sampling sites given by their acronyms. Cachapoal sub-basin: PA: Pangal river, CQ: Clonqui stream, CC: Cachapoal river at Clonqui stream confluence, CH: Cachapoal river in the Andes, CK: Coya river, CL-S: Claro river of Cauquenes, CL: Cachapoal river at Claro river confluence, CD: Cadena stream (in Cachapoal river catchment), CT: Cachapoal river at Punta Cortés, CE: Cachapoal river at Doñihue, ID: Idahue stream, RO: Claro river of Rengo, ZA: Zamorano stream, CR: Cachapoal river at La Ratonera and CJ: Cachapoal river at La Junta. Tinguiririca sub-basin: LH: Lo Herrera stream, LD: Las Damas river, TA: Tinguiririca river in the Andes, AZ: Azufre river, TZ: Tinguiririca river at Azufre river confluence, TI: Tinguiririca river at La Iglesia, TR: Negro river, TP: Tinguiririca river at Puente Negro, TF: Tinguiririca river at San Fernando, TN: Tinguiririca river at Nancahua, TU: Tinguiririca river at Cunaco, BO: Chimbarongo stream, TH: Tinguiririca river at Huique. CO: La Condenada stream, CU: Cunaco stream and TJ: River Tinguiririca at La Junta. Rapel sub-catchment: CN: Las Cadenas stream, MI: San Miguel stream and CS: Las Cadenas stream at the confluence between San Miguel and Las Cadenas streams. Alhué sub-basin: AL: Alhué river and PL: Las Palmas stream, and RA: Rapel lake. B) Land use maps and C) Land use changes (%).

The Rapel Basin is dominated by a Mediterranean climate (Aitken et al., 2016). The catchment is characterised by a tectonic topography, which includes the Andes Mountain range to the east and the lower elevation Coastal Range to the west. The topography and millennial-scale denudation and deposition processes have resulted in the creation of two lowland plains in between the ranges, with fertile soils that have become increasingly dominated by intensive agricultural production since the late 20$^{th}$ Century. Land cover varies along the stream profile in the catchment (See Fig. 1). Upstream, barren land with natural vegetation is most common. Along the coastal range and throughout the lowland plain cropland, evergreen forest and savannah are dominant. The coastal range is the area where agricultural activity is concentrated. Cultivation of this land has been increasing within the basin (~7%), as more olive trees, avocado orchards, citrus and vineyards have been planted on hilly zones in response to market demand and commercial opportunities for expansion. Changing land use practices is hypothesised to have increased erosion risk, where permanent native flora cover is lost through clearance and tillage for cultivation. In addition to the long-term trend for extensively managed montane pasture to be converted into fruit and grain production, cultivation patterns within the lowlands also are subject to temporal variation (Schulz et al., 2010).

Copper (Cu) mining covers a small area within the catchment. It is an important economic activity in the basin. The state mining company operating here is one of the biggest underground mines in the world and produced near 460 thousand metric tons during the year 2021. It is located in the Andes and mining products (Cu) have the potential to enter the river network through the Coya River which feeds into the Cachapoal River (Pizarro et al., 2003), contributing other metals such as Molybdenum (Mo) and Antimony (Sb) to the sedimentary signature of this river (Lacassie and Ruiz-Del-Solar, 2021).

### *2.2 Sampling strategy to evaluate sediment source dynamics: the tributary tracing approach.*

The sediment fingerprinting technique can be implemented by using several tracer properties that are representative of characteristics of the main sources within the catchment (Haddadchi et al., 2013). Nevertheless, not all properties may account to all different characteristics (e.g. land uses) when basins are inherently complex. Due to the wide variety of natural and anthropogenic activities carried out in the catchment, including mining, agriculture (several crops), livestock, industrial, and urbanisation, we used major and minor element geochemical fingerprints as a comprehensive approach likely to ascribe sediment sources and subsequent water pollution in this context.

The tributary approach for sediment fingerprinting involved sampling in the different upstream tributaries and using them as potential sediment sources (Laceby et al., 2017). Since most of the significant particle size effects occur typically during the initial stages of mobilisation and transportation from slope to channel, leading to potential particle size enrichment or depletion on fingerprinting properties, the tributary tracing approach offers a means to account for these variations (Laceby et al., 2017). In this study, an assumption that tributary sediments comprise a composite mixture of eroded sediment from the tributary catchment dominant sources has been made (Wynants et al., 2020).

Approximately 300 sediment samples were collected at different locations in the Rapel lake and its main tributaries during 2019 (Fig. 1): Alhué, Las Palmas, Cachapoal, Tinguiririca and Las Cadenas. At every tributary, 7 to 10 spatially integrated channel bed deposited sediment samples were taken upstream of the confluence where recent sediment deposits were clearly evidenced. With regards to Tinguiririca and Cachapoal sub-catchments, low order tributaries to their higher order mainstem were also sampled before the confluence to capture characteristics of lowland feeder streams (Fig. 1). Surficial sediments were sampled using plastic spatulas to avoid cross contamination, placed in double plastic bags and the central point of each sampling area was recorded using a GPS. Hence, the surficial sediment samples were used in the following context: First: Recently deposited sediments collected at the main rivers and tributaries were employed to characterise the signature of dominant sources within the tributary catchment. Second: Sediment samples collected in the lake (sediment mixing/accumulation zone) were used to estimate the contribution of the different upstream tributary sediments. Of note, Additional sampling points presented in the map were only considered when evaluating the contaminant distribution along the river channel and not for sediment source apportionment.

Sediment cores (n = 6) were taken in the upper zone of the reservoir during 2020 (around 12 km from the wall and around 22 km from the confluence of the two main tributaries). The sampling was done in a meander area that was selected as a natural and representative sediment accumulation zone (Fig.1. Lat. -34.113278°; Long. -71.516222°). The average water depth was 30 m. This location represented the maximum feasible depth for coring with depths at the centre of the dam and places near to the wall reaching greater than 50 m. The samples were collected from an anchored zodiac using a Ubitech gravity core equipped with acrylic tubes of 2.5 m length and 6 cm of internal diameter. Sediment core depths ranged from 88 to 122 cm. In parallel, surficial sediment samples (3 samples with 3 to 5 sub-samples for analysis) were taken using a Van Veen grab sampler and short-cores (about 10 – 15 cm of sediment depth) to assess contemporary sediment source contributions and spatial variability therein.

### *2.3 Sample preparation and analysis*

Riverine sediments were oven-dried at 60°C for 3 to 4 days, sieved through a 2 mm sieve and then through a < 63 µm mesh with the fine fraction packed into plastic bags for further analysis. Core samples were sliced every 2 cm immediately after sampling, placed in double Ziploc plastic bags and subsequently dried using a drying cabinet at 60°C. Samples were gently disaggregated with a pestle and mortar, then packed for further analysis.

A subsample of approximately 1 g of dried and sieved particles was used for particle size analysis using a Malvern 2000 series laser granulometer. Herein, 3 ml of 6% Hydrogen Peroxide ($H_2O_2$) was added to every sample to remove organic matter over a 12-hour period. Subsequently, another aliquot of 6% of $H_2O_2$ was added to each sample to assess if effervescence still took place. Samples were then placed into a water bath at 80°C for 2 to 4 hours and cooled at room temperature. Before laser diffraction analysis, sodium hexametaphosphate was added to every sample as a dispersant. Particle size distribution analyses were performed in triplicate and results were assessed statistically to check for large dispersion between particle size distributions percentiles in which a coefficient of variation < 10 % between percentiles was accepted as a valid result.

Samples were analysed for major and minor elements by Wavelength Dispersive X-Ray Fluorescence (WD-XRF, PANalytical Axios Max) as pressed pellets. The sieved < 63 µm fraction was homogenised by milling for 20 min at 300 RPM, mixed with Ceridust 6050M S1000 polypropylene wax binder, packed into aluminium cups and pressed into pellets. The instrument was operated at 4 kW using a Rh target X-ray tube. During sequential elemental analysis, tube settings ranged from 25 kV, 160 mA for low atomic weight elements, up to 60 kV, 66 mA for higher atomic weight elements. All analyses were carried out using the "Pro Trace" analysis application. For Quality Control, analyses of random samples were carried out in triplicate to evaluate method precision. Results showed a relative standard deviation ranging between 0.5 to 31% for all elements analysed. Instrument performance is validated by routine analysis of certified stream sediment reference material (NCS DC 73309) where recovery is typically within ±10% from certified values. Instrumental Limits of Detection (LoD) were obtained, and instrumental drift (using a multielement glass sample) were also checked during sample runs.

From the six cores taken, the deepest (122 cm) was selected to perform sediment geochronology and thereby estimating sedimentation rates via fallout $^{210}Pb$ (Appleby, 2001). This deepest core was selected for profiling to minimise the risk of disturbance due to seasonal emptying and filling of the lake for recreational/hydropower generation purposes. $^{210}Pb$, $^{214}Pb$, $^{137}Cs$ and $^{241}Am$ activity concentrations were measured using a well type HPGe gamma spectrometer. Total $^{210}Pb$ was determined via its gamma emissions at 46.5 keV, $^{226}Ra$ (to determine the supported $^{210}Pb$ component) by the 295 keV and 352 keV gamma rays emitted by its daughter isotope $^{214}Pb$, and $^{137}Cs$ was measured by its emissions at 662 keV (Appleby, 2001). The CRS (Constant Rate of Supply) model (Appleby and Oldfield, 1978) was applied to construct core chronologies and sedimentation rates from activity concentrations. Since the reservoir was constructed after the $^{137}Cs$ peak fallout in the 1960s, there was a lack of an independent chronological marker. The CRS-model dates were benchmarked to the basal core base date by reservoir construction in 1968 and the subsequent chronological estimate was independently evaluated by comparing sedimentation rates patterns through time to floods records from the main tributaries in the upper part of the core (Fig. S1). All samples were analysed at the ISO9001 certified University of Plymouth Consolidated Radioisotope Facility (CoRIF), UK.

### *2.4 Statistical analysis and mixing modelling*

#### *2.4.1 Tracer selection*

An important aspect before performing any calculation for geochemistry-based sediment apportionment is the determination of relevant tracers. From 44 elements analysed, tracers to be used in the mixing model were selected as follows: (a) Firstly, we removed all the elements that were below the instrumental limit of detection (see Table S1); (b) we removed the elements that have the potential to be soluble and may present issues of transformation of the signatures from the released sediment to the final sink. This criterion was done according to the 90% ratio between natural dissolved and total elemental river transport (Collins et al., 2020). The elements discarded were Cl, Br, S, Na, Sr, Ca, Sb, Mg, Mo, As, Ba and K. (c) a further selection procedure based on boxplots was applied (Ovando et al., accepted for publication). In brief, those tracers that fulfilled the following criteria were selected: 1) The median of the mixture lies inside the maximum third quartile and minimum first quartile among the sources. 2) The mixture distribution lies within the maximum and minimum median among the sources. 3) Considering just the sources, there was a positive difference between the maximum first quartile and minimum third quartile. 4) At least, one of the sources distributions has to be different from the mixture distribution. Finally, d) source and mixture particle size distributions were assessed and relationships with element concentrations were evaluated through correlation analysis where required (Laceby et al., 2017). After the tracer selection procedure, Principal Component Analysis (PCA) was conducted to visualise source and mixture groupings in the ordination plot along the first two main principal components retained based on their percentage of explained variance. Tracer selection and PCA were performed using R statistical package.

#### *2.4.2 Mixing model structure*

The mixing model employed was MixSIAR (Stock et al., 2018). The model was run with uninformative prior, error structure as ‘residual error' and chain length of ‘very long’. Convergence of the model was evaluated using the Gelman-Rubin diagnostic, in which case all variables were below 1.05. This in accord with accepted limits in this field of application (Blake et al., 2018a; Smith et al., 2018). All results are provided as the mean and standard deviations as well as credible intervals from the posterior distributions.

# 3 Results and discussion

## *3.1 Tracer selection*

The potential influence of particle sorting effects on sediment fingerprinting properties (e.g. element concentrations) have been widely debated in the literature (Gaspar et al., 2022; Laceby et al., 2017; Smith and Blake, 2014). Differences in grain size between soil and sediment particles can bias mixing model results (Gaspar et al., 2022). To address the potential effects, several approaches have been applied such as sample fractionation and particle size correction factors. However, it has been reported that the use of particle size correction factors can lead to overcorrection (Smith and Blake, 2014). Here, a similar approach to that reported in Smith et al., (2018) and Gaspar et al., (2022) where evaluation of potential grain size effects was carried out. Differences in the median particle size distribution (D50) between tributary and lake sediments were found (see Fig. S2). Therefore, a correlation analysis between the Specific Surface Area (SSA) and elemental concentrations of all tributary samples was performed (Table S2). Only one element (out of 27) showed strong correlation with SSA (i.e. Cr, R = -0.55, see Table S2) while four presented moderate association (i.e. V, Mn, Rb and Cs. R between -0.3 and -0.5 or 0.3 and 0.5, see Table S2) and six elements had weak although significant correlations (i.e. Fe, Ni, Zn, Zr, Pb and U. R between -0.2 and -0.3 or 0.2 and 0.3, $p < 0.05$, $n = 60$). Consequently, due to low association between these elements and SSA, instead of removing them (or going further in particle size effects assessment) we tested them within the tracer selection procedure (see section 2.4.1). Here, the inclusion of some weak-to-moderate correlated tracers was not considered likely to increase, or significantly influence the source apportionment results.

It is noteworthy that when performing historical sediment source apportionment, it may not be appropriate to use elements coming from pollution point-sources (such as Cu from mining in this study) due to changes in concentrations along the core profile in relation to pollutant dynamics compared to contemporary source samples used to characterise the input. This implies that those elements may be discarded during the tracer selection procedure for some mixtures and included in others. For instance, range test evaluation of Cu concentrations at the top of the core suggest inclusion of this tracer in the modelling, while the opposite occurs when evaluating the fit of sediment from the bottom (Fig. S3). The influence on the inclusion of Cu in the tracer set for modelling can be seen in sections 6 and 7 of the core, where results did not differ significantly when Cu is included (section 7) and not included (section 6) (see Table S5), possibly due to the number of tracers employed in a single calculation (between 12 to 18). Here, it is expected that one rogue tracer would not significantly drive or influence mixing model outputs.

The effect of tracer selection procedure was assessed using principal component analyses. It was found that for each mixing area (Fig. S4a – e): 1) mixture data points always fell within the source factor space, 2) in general, source (tributary) groupings presented low dispersion, and 3) discrimination among source groups improved. The explained variance ranged from 56.8 to 81.8% which can be considered acceptable for all PCAs performed.

### *3.2 Sediment source apportionment*

#### *3.2.1 Contemporary source apportionment*

To visualise the sediment apportionment at different scales within the catchment, calculations were performed at different mixing areas in the two main rivers (Cachapoal and Tinguiririca) and the final mixture (Rapel Lake). The results obtained are detailed in Table 1.

**Table 1.** Spatial contemporary source apportionment summary statistics obtained from MixSIAR for the Rapel catchment (SD: Standard Deviation and CI: 95 % Credible Interval).

| | Mixing Zone | Sources | Mean contribution (%) | SD | CI |
|---|---|---|---|---|---|
| Cachapoal | CL | CH | 2.7 | 1.8 | 0.1 - 6.9 |
| | | CK | 81.4 | 4.7 | 71.7 - 90.1 |
| | | CQ | 1.7 | 1.7 | 0.04 - 5.9 |
| | | PA | 14.2 | 5.0 | 4.6 - 24.1 |
| | CR | CD | 3.5 | 3.2 | 0.1 - 11.9 |
| | | CL | 17.3 | 4.0 | 8.9 - 24.9 |
| | | CL-S | 9.3 | 4.0 | 1.7 - 17.5 |
| | | ID | 5.7 | 5.0 | 0.2 - 18.9 |
| | | RO | 60.5 | 8.9 | 40.1 - 76.0 |
| | | ZA | 3.7 | 3.2 | 0.1 - 12 |
| Tinguiririca | TF | AZ | 32.8 | 6.3 | 19.6 - 44.5 |
| | | LD | 7.1 | 5.6 | 0.2 - 20.9 |
| | | LH | 14.9 | 6.2 | 3.3 - 27.4 |

| | | | | | |
|---|---|---|---|---|---|
| | | TR | 45.2 | 7.7 | 30.0 - 60.3 |
| | TJ | BO | 6.3 | 3.9 | 0.5 - 15.1 |
| | | CO | 5.7 | 3.4 | 0.5 - 13.6 |
| | | CU | 37.3 | 8.2 | 20.3 - 53.0 |
| | | TF | 50.7 | 7.1 | 37.2 - 65.2 |
| Final Mixture | RA | AL | 12.5 | 7.8 | 0.9 - 29.6 |
| | | CR | 63.2 | 10.6 | 39.3 - 81.6 |
| | | CS | 5.5 | 4.4 | 0.2 - 16.0 |
| | | PL | 11.1 | 7.6 | 0.5 - 28.5 |
| | | TJ | 7.8 | 7.0 | 0.2 - 25.9 |

The Cachapoal river sub-catchment was divided in two calculation sections (Table 1). The upstream portion draining the Andes Mountains was evaluated in the first mixture area in the Cachapoal confluence with Claro River (CL sampling site). Details of tracers selected, discarded, etc. can be seen in Table S4. Results at this point indicated that the dominant source at the CL mixing point was the Coya River (CK) with 81.4 ± 4.7 %, followed by the Pangal River (PA), Cachapoal Andes (CH) and Clonqui River (CQ) with contributions of 14.2 ± 5.0, 2.7 ± 1.8, 1.7 ± 1.7 % respectively. These results were in agreement with tributary groupings observed in Fig. S4.a, where CK samples clustered together with CL-M samples. The main sediment source to this point comes from the tributary that has a strong signal of mining activities with significant influence of Cu (see PCA Fig. S4.a). The next mixing area was the Cachapoal at the "La Ratonera" sampling point (CR). A further point and closest to the confluence of the two main rivers was the Cachapoal at "La Junta" (CJ). However, due to the river being canalised for irrigation, a large proportion of fine sediment bearing flow is diverted into the agricultural area. Subsequently, little water reaches the CJ point. For this reason, the CR sampling point was used to represent the integrated sediment input from upstream (with no river intervention). The results from the mixing model to this point showed that the main sources (tributaries) were the Claro River (RO) with 60.5 ± 8.9%, followed by CL with 17.3 ± 4 % and Claro River of Cauquenes (CL-S) with 9.3 ± 4.0%, the rest were distributed between 3.5 to 5.7 % of contribution (CD, ID and ZA). The dominant contribution of RO sediments to CR mixing point is expected as RO is a river that carries sediments from agricultural areas along the whole central valley and delivers them close to the mixing point (~ 10 km). Similarities in geochemical sediment signatures can be confirmed by looking at the PCA score plot where RO samples are overlapping the CR sample cluster (Fig. S4.b). It is important to emphasise that we considered the previous mixing point CL as a source for the second mixing area evaluated and not the individual tributaries from the first mixing area. Also, the contribution of CL to this point is expected to be highly influenced by the sediments coming from the Coya River (CK) whose contribution was more than 80% to mixing point CL. Nevertheless, when performing the full calculation of all tributaries upstream of CR, the main contributions did not change considerably (e.g. RO from 60.5 to 62.5% data not shown). It is not surprising that the tributary RO is the main source as it flows through the whole Central Valley carrying sediments from the agricultural activities and delivering its sediments very close to the mixing point CR (~16 km).

Considering that a tributary transports sediment that integrates the inputs from its catchment area (Haddadchi et al., 2013), we renamed and summed the individual contributions of each tributary to the final point of the river according to the main land uses/activities occurring in the Cachapoal sub-catchment. Results showed that the 73% of the sediment contributions originated from areas that harboured agricultural activities in the Central Valley. The other comes from the mining activity conducted in the Andes with 14%, and finally that which is derived from the Andes attributed to natural erosion processes such as snow melting and glacial retreat inputs contributing 13%.

Regarding the Tinguiririca sub-catchment, at the first mixing zone (sampling point TF, Table 1) the main sediment source was derived from the Claro River at Puente Negro (TR) (45.2 ± 7.7%), followed by Azufre River (AZ) with 32.8 ± 6.3% and finally Tinguiririca Andes at Lo Herrera stream (LH) and Las Damas River (LD) with 14.9 ± 6.2 and 7.1 ± 5.6% respectively. At the second mixing point, La Junta (TJ), results indicated that the main contributor were the previous mixing point area TF with 50.7 ± 7.1%, then tributaries Calleuque stream (CU) (37.3 ± 8.2%), and the others (CO, BO) contributed the remaining 12%. Considering the same assumption applied to the Cachapoal sub-catchment, the integration of results from these ‘type’ tributaries suggested that the dominant sources that contributed to this mixing point in the Tinguiririca river were derived from the Andes tributaries (51%) followed by tributaries that drain agricultural activities in the Coastal Range area (43%) and tributaries that hold agriculture activities in the Central Valley (6%)

For the final mixture zone at the Rapel lake (Fig. 1), the source points were selected from the lowermost sampling point of each sub-basin as a representative composite of the sediments derived from each. These included the sampling points at the two main rivers (CR: Cachapoal river and TJ: Tinguiririca river) as well as the independent sources that contribute directly to the Lake (AL: Alhue river, and PL: Las Palmas stream-in

Las Cabras sub-catchment and CS: Las Cadenas stream). Differences in tributaries sediment geochemical composition were clearly observable after the tracer selection procedure (see PCA score plot in Fig. S4.e) which indicates that anthropogenic activities exert a significant influence on the riverine sediment geochemical signatures even in catchments under similar geology (e.g. Alhué and Las Cabras catchments). Apportionment results indicated that the Cachapoal sub-catchment (CR) was the main source to the final mixing area with a 63.2 ± 10.6%, followed by Alhue sub-catchment (AL) with 12.5 ± 7.8%, Las Cabras sub-catchment (PL) with 11.1 ± 7.6%, Tinguiririca sub-catchment (TJ) 7.8 ± 7.0% and finally Las Cadenas sub-catchment (CS) with 5.5 ± 4.4%.

### *3.2.2 Temporal variability in sediment source contributions to the lake*

The activity concentration depth profile of fallout $^{210}$Pb (Fig. S5) showed a profile typical of a dynamic sedimentation rate with a lack of an exponential profile, rather it had notable peaks and troughs linked to depositional episodes. The lack of an independent fallout radionuclide marker (e.g. $^{137}$Cs) led to a key assumption that the deepest part of the core was the date when the dam was built (1968-69). Additional evidence of major flow events from river flow gauge records was then utilised to corroborate the resultant geochronology. This information was derived from four permanent stations within the two main rivers that have records extending back to 2003. Assuming that some physical phenomena that occurs in the upper and lower part of the catchment will affect the water/sediment load of the lake, we plotted some parameters such as stream flow (m$^3$ s$^{-1}$) and sediment concentration (mg L$^{-1}$) alongside the mass accumulation rate (MAR) estimated from the geochronology model (Fig. S1). From these plots, the moving average of the stream flow showed a close relationship with MAR for the dated core (Fig. S1). This coherence in records provided support for the relative derived dates allowing us to proceed in using changes in sedimentation rates to structure sediment data for unmixing purposes. Consequently, the number of core sections considered were 11.

Historical source apportionment of each section showed that the historical dominant sediment source has been the Cachapoal river (CR) contributing from 33-63%, followed by the Alhue river-AL (14-43%) and finally the other three tributaries (CS, PL, and TJ) with minor contributions ranging from 2-24% (See Fig. 2 and Table S3 for summary statistics).

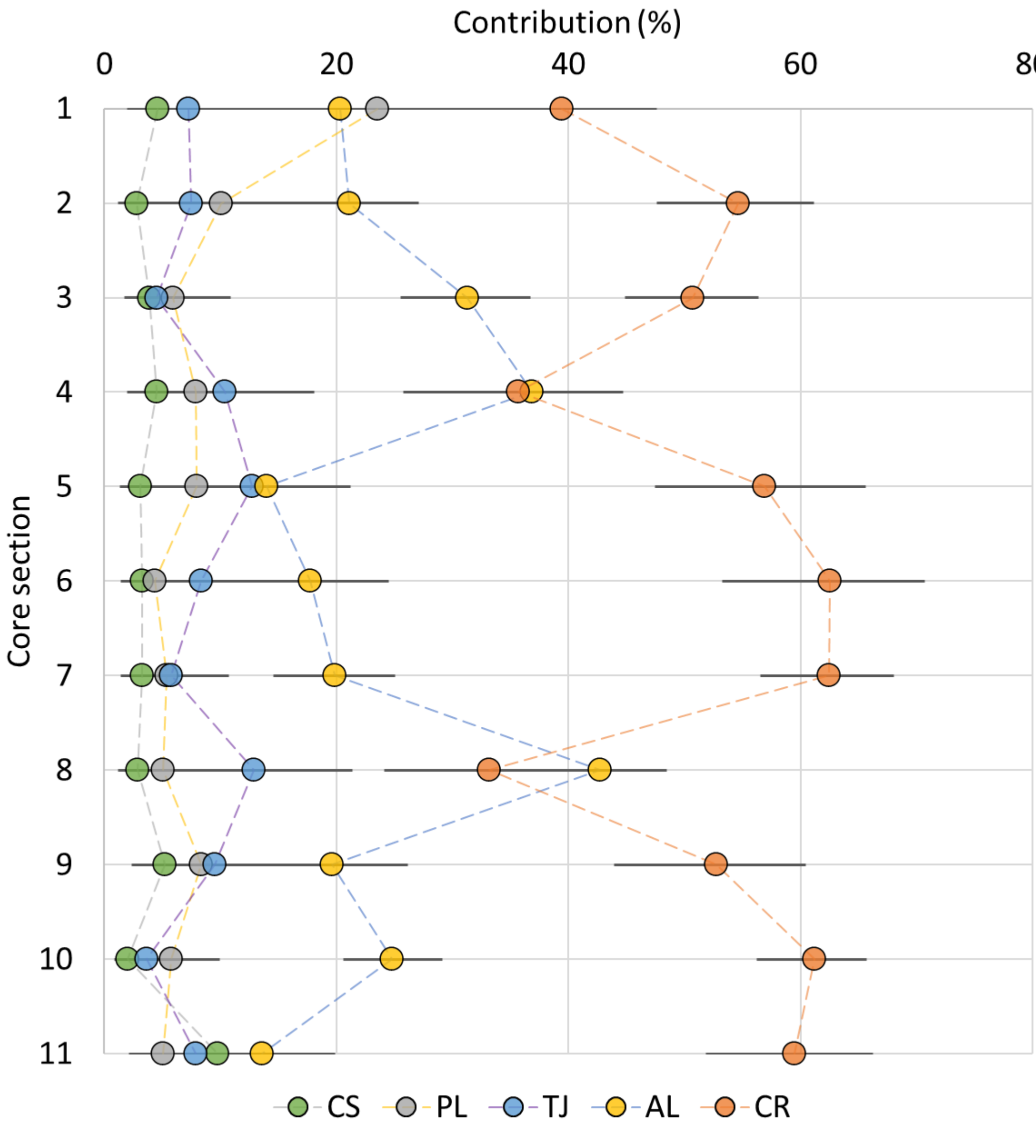



**Fig. 2**. Historical tributary source contribution (%) for each core section obtained from MixSIAR mixing model. Points indicates median relative contribution and error bars 50% credible intervals. Core sections indicate cumulative depth from bottom: 108-120cm (11) to top 0-6cm (1) with deposition period from 1968-1976 (11) to 2017-2019 (1). CS: Las Cadenas stream, PL: Las Palmas stream, TJ: Tinguririca river at La Junta, AL: Alhué river and CR: Cachapoal river at La Ratonera.

Tributaries were classified according to the main activities performed in their catchment area (Andes-Natural erosion, Mining and agricultural activities, as in section 3.2.1). Here, the historical contributions (see Fig. S6) are much clearer and in agreement with catchment antecedents, as historical contributions from the

Andes have been relatively constant, mining has decreased, and agricultural inputs have increased throughout the years.

### *3.2.3 Causes of siltation in the Rapel lake.*

There are various causes of siltation in the Rapel Lake. On the one hand, field observations suggest that both main rivers (i.e. Cachapoal and Tinguiririca rivers) carry substantial amount of sediment from the Andes mountains (especially in summer), with significant contributions from tributaries along the central valley and the coastal range where changes in sediment colour and turbidity are clearly observable. Of the two main rivers, the Cachapoal river contributed the majority of sediment (~63%), at least 8 times higher than the Tinguiririca river (~8%) in a contemporary context. This is in agreement with overall flow regimes of each river (see river flows information on section 2.1 and Fig. S1). Nevertheless, natural sediment sources represented by glacier retreat and snow melting from the Andes, in light of the current findings, were low. On the other hand, the Alhué river (AL) contribution is associated to agricultural activities and, to a lesser extent, copper mining refining activities. Even though, it is much closer to the mixing area selected it is not expected to contribute sediments significantly due to low river flow compared to Cachapoal and Tinguiririca rivers (Alhué stream contributes around 12% of the total flow into the lake and it has an mean annual flow of 2.0 $m^3$ $s^{-1}$ (Rossel and de la Fuente, 2015)). However, its contribution resulted higher than the Tinguiririca river.

Agricultural activities across the system are also an important source of fine sediment to the lake. For instance, around 60% of CR mixing point contribution comes mainly from agriculture (RO tributary, see section 3.2.1) and the historical contribution of PL stream (dominated by agricultural activities) presented an increment in the top two sections of the core (Fig. 2). This contribution goes against local expectation of low input from agricultural land which historically occupies the low gradient/flat terrain of the basin floor. Recent years, however, have seen important land use changes, where steeper terrain of the Coastal Range has been cleared of natural vegetation and planted with vineyards and avocado, olive and citrus trees (Fig. 3a, c and d). These land use changes can be seen on Fig. 1b and c. where croplands have increased by ~7% and forest have decreased ~35%. These changes have been observed predominantly on hilly areas by the Coastal Range (Fig. 3b, c and d). This conversion of steeper terrain from natural vegetation to agriculture, in response to economic drivers to increase production, have generated an important sediment load transferred into the lake. Clearance of the secondary forest of hilly zones and conversion to agricultural use relies on heavy machinery leading to compaction wherein plough lines are required to be parallel to the slope due to the steep gradient and safe operation of machinery (Fig. 3b). Overall, conversion of land from natural to agricultural usage generates areas of bare soil with compacted tracks aligned with the steep slopes leaving a landscape highly prone to erosion.

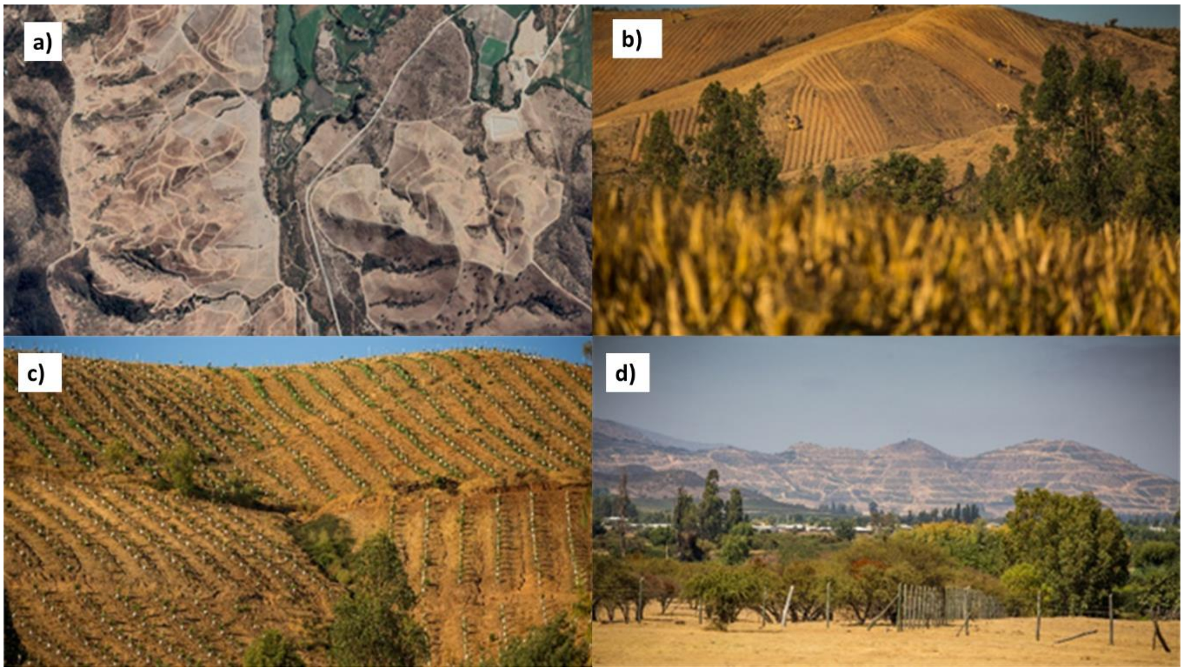



**Fig. 3**. Images of the land use changes in the Coastal Range area. a) Satellite Google Earth image showing hillslope clearances; b) Land reconversion for cropping; c) and d) Planted hillslopes. Note that direction of the ploughing is parallel to the slope.

### *3.3 Spatio-temporal patterns in sediment-associated contaminants*

Sediment associated contaminants includes a series of chemicals that form part of a legacy of current and past natural and anthropogenic activities within the catchment. Sediment quality is highly influenced by the presence of contaminants in the water (Förstner, 1987). The Rapel river system contains high concentrations of Cu due to mining activities, as well as Mo. In addition, As is also present as a naturally-occurring contaminant associated with volcanic origins and enhanced in riverine sediment by selective transport of fine materials. Phosphorus is a typical contaminant of the area associated to the agricultural activities (Lacassie and Ruiz-Del-Solar, 2021). The spatial distribution of measured elements within the catchment (sediment in tributaries and main river courses, Fig. 4) offers evidence that enhanced Cu concentrations (Fig. 4a) have a dominant tributary source that is likely influenced by mining activities that deliver sediment and leachates. These subsequently interact with sediment along the river Cachapoal from the Andes through to the Coya River, with Cu concentrations averaging 1282 ± 109 mg $kg^{-1}$. Concentrations remain high in sediments near to the confluence of this tributary with the main river, values at this point "CL" (distanced around 9 km from the confluence) presented average concentrations of 1218 ± 87 mg $kg^{-1}$. Moving downstream, Cu concentrations are diluted through the river Cachapoal and ultimately sediment is released into the lake with concentrations of 453 ± 102 mg $kg^{-1}$ (around 3 times less compared with its origin in the mine-affected tributary). Herein, sediment pulses from the tributary source (CK) to the material recovered from the lake covers a distance of 124 km along which a range of additional sediment sources are active and contributing to the mix. In comparison, Cu concentrations in the Tinguiririca river varied from values of 35 to 118 mg $kg^{-1}$ (average 64 ± 14 mg $kg^{-1}$). These values can be considered as baseline levels in the wider basin area, as in this river there is no evidence of mining activities. Similarly, the behaviour of Mo appears to be as a secondary product of mining activities (Fig. 4b), with average concentrations in mountain reaches (CK) of 50 ± 3 mg $kg^{-1}$. However, Mo concentrations in this case are not exclusively from sources derived from the Andes, as the main source of Mo into the lake comes from another mining activity located in the coastal range area in the Alhue sub-catchment (AL), where concentrations in sediments averaged 162 ± 74 mg $kg^{-1}$, possibly derived from a tailing dam that accumulates mining waste for the operations taking place in the Andes. Concentrations in the lake were 10 ± 2 mg $kg^{-1}$; at least sixteen time less concentrated that the Alhue river (AL). As stated with Cu, the concentrations of Mo in the Tinguiririca River and tributaries ranged from 1 to 8 mg $kg^{-1}$ (average 3 ± 1 mg $kg^{-1}$). These values can also be considered as background levels within the catchment.

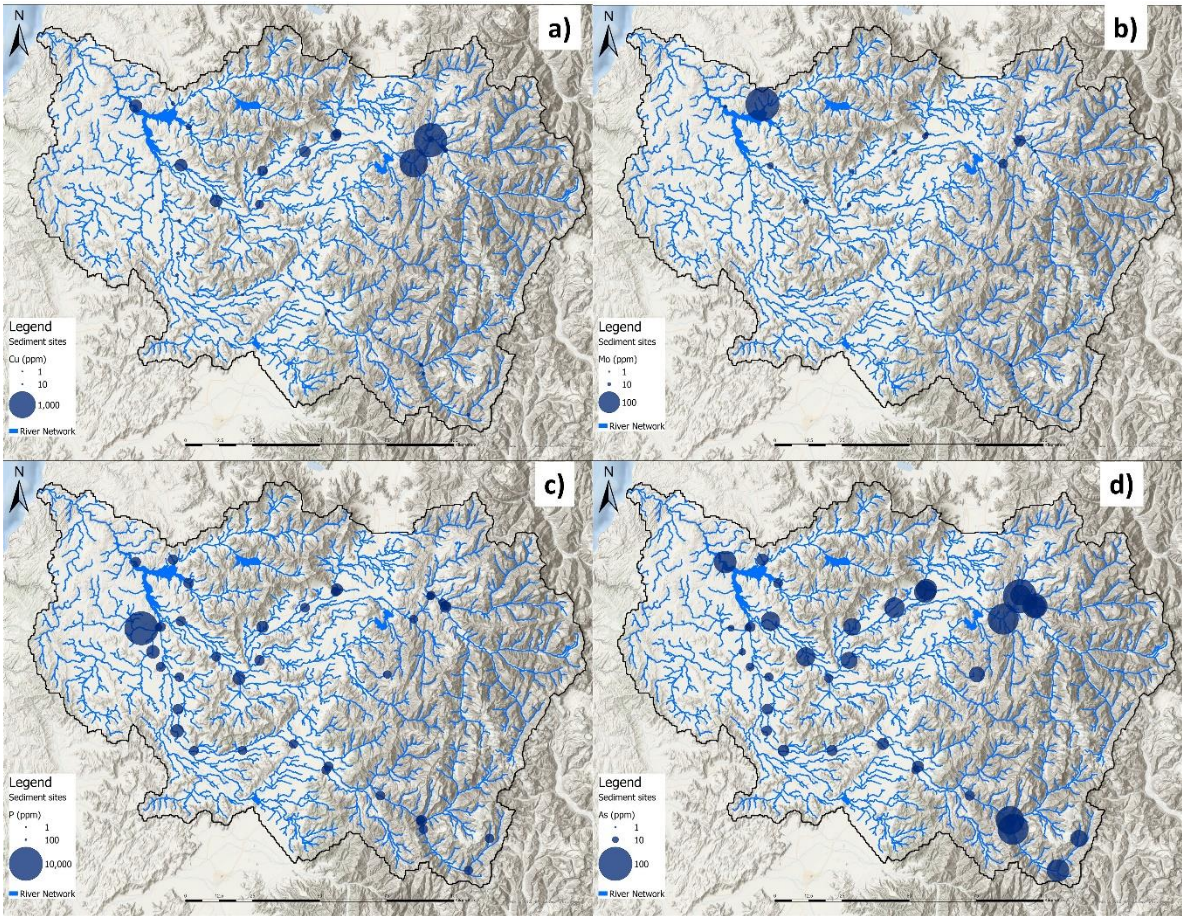



**Fig. 4.** Spatial distribution of some sediment associated contaminants (SAC) in the Rapel Catchment. a) Cu, b) Mo, c) P and d) As.

Elevated phosphorus concentrations of note appear to emanate from the dryland area (Coastal Range) with a maximum value of 8255 mg kg$^{-1}$ at the San Miguel stream (MI) sampling point (average value of 5283 ± 1335 mg kg$^{-1}$). This sampling area is located near to 'pig farms' and possibly highly influenced by this activity. Concentrations from the rest of the sampling points within the catchment were similar (~1400 mg kg$^{-1}$). The concentrations detected in the lake averaged 1443 ± 116 mg kg$^{-1}$. In a eutrophic lake in China P values ranged from 544 to 932 mg kg$^{-1}$ indicating the risk of P-related events in this system (Wang and Liang, 2015), suggesting that the Rapel lake may be also be at risk of eutrophication if P is not bound stably in the sediment column.

Arsenic concentrations (Fig. 4d) were higher in proximity to the Andes with average values for both rivers of 35 ± 20 mg kg$^{-1}$. Values at the lake were around 42 ± 2 mg kg$^{-1}$. For the rest of the sampling points in both rivers average values were 24 ± 11 mg $kg^{-1}$. Therefore, it was concluded that the sources of As in the catchment are derived from naturally occurring As within the Andes (Lacassie and Ruiz-Del-Solar, 2021) with some enhancement by fluvial sorting (Horowitz and Elrick, 1987). Previous studies in the Rapel catchment detected maximum As concentrations of 72 and 21 mg $kg^{-1}$ in the Cachapoal and Tinguiririca Rivers respectively. Both rivers contribute sediment-associated As to the lake due to high background As concentrations in Andean geology and consequently concentrations of this element in fine sediments can reach the lake and influence its accumulation due to natural processes (Lacassie and Ruiz-Del-Solar, 2021), as well as being enhanced by mining (Figure 4d).

In order to stablish if concentrations reported within the sediments have the potential to produce certain effect on living organisms, measured concentrations were compared to Probable Effect Concentrations (PEC) (MacDonald et al., 2000). The consensus-bases PEC for elements of interest in this study are: Cu: 149 mg $kg^{-1}$; As: 33 mg $kg^{-1}$. Regarding the Cu values, in the Tinguiririca River all concentrations in sediment samples were below the threshold suggested for this element. The other tributaries that contribute sediments directly to the lake "Las Palmas" and "Alhue" were over and below this threshold respectively. Regarding the sediment in the lake itself, concentrations were above this threshold. The samples, and as stated above, from the Cachapoal river and tributaries showed high concentrations of Cu and most of them were above the PEC. Nevertheless, samples collected at the higher part of the Andes mountains were below (CH, PA, CQ). This demonstrates the potential negative effect of Cu (present in sediments that are derived from the processing of minerals containing Cu) on aquatic living organisms in the receptor aquatic environments.

Concerning Arsenic values, the PEC threshold was mostly surpassed by all the sediment samples within the Cachapoal river, and only in samples collected by the Andes in the Tinguiririca River (TZ, TA, ID). Samples from sediment collected at AL and PL were below this threshold. Values of the Lake sediments also exceeded the PEC. This evidenced that natural sources of As by the mountains put in potential risk the living organisms present in this area.

Regarding the temporal inputs of sediment-associated contaminants into the lake, the profile for the four elements of interest (Fig. 5) and how they behaved since 1968 offer important temporal perspectives in pollution dynamics in the system. Profiles of Cu and Mo presented strong correlation (R: 0.92, $p < 0.05$), this is expected as those elements demonstrated to come from the same origin (mining activities). Additionally, the mining company during the year 1985 began to use a different rock that was depleted in Cu and consequently the amount of Cu present in the mineral was much lower than the one used before 1985. This process can be clearly seen in the profiles for Cu and Mo with a drop in concentration in the sediment profile since 1985. Also, the mining activity since 1985 invested in the improvements of their environmental impacts as it is clear that Cu inputs from 1985 also tended to decrease. Nevertheless, all the values were higher than the PEC for this element. The same is true for As concentrations where values were over the threshold and relatively constant throughout the years. Phosphorus, on the other hand, followed a step change upward in the late 1970s, where concentrations have remained relatively constant from 1981 potentially reflecting sustained but noting also that the non-conservative properties of this element can also influence profiles (Tiecher et al., 2019).

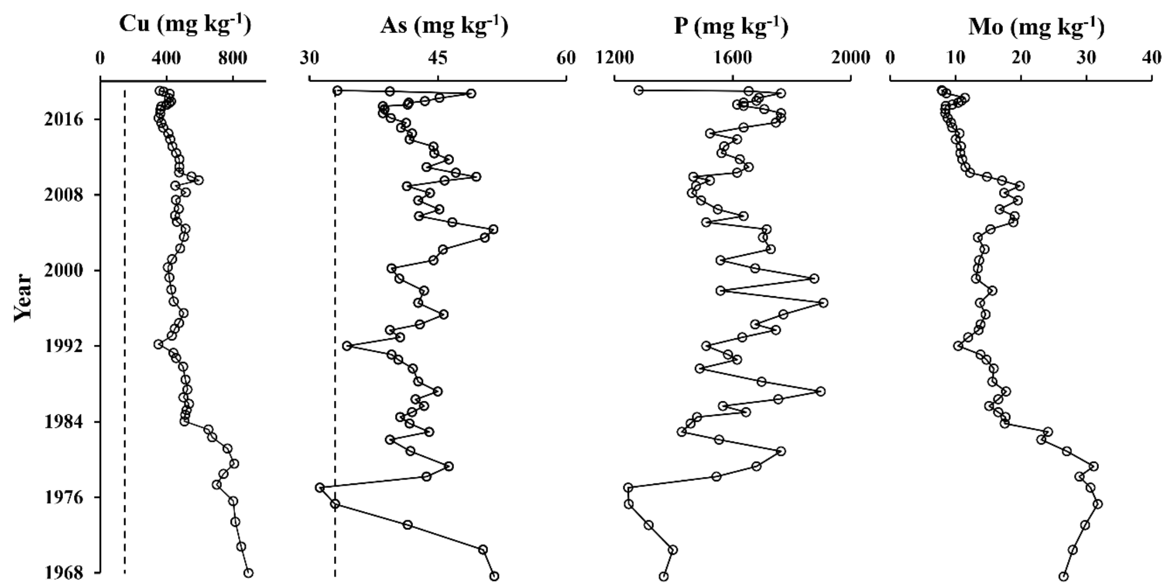



**Fig. 5**. Profiles of concentrations of four SAC. For Cu and As segmented lines are indicative of their PEC values.

*3.4 Challenges and opportunities for sediment management in major hydropower basins*

Adopting a holistic ‘systems-thinking’ approach is paramount in tackling hydropower sediment management challenges that comprise the food-water-energy-environment nexus (Blake et al., 2018b). Hydropower basins are complex systems where numerous factors, including dynamic change in land use practice, climatic conditions and hydro-geomorphological process influence sediment dynamics. Understanding these interconnected factors and their interactions, through interdisciplinary and cross-sectoral working, is crucial for devising comprehensive management plans (del Valle, 2023). Collaborative efforts between stakeholders can enhance the effectiveness of sediment management strategies and facilitate sustainable land management decisions. The Rapel River basin serves as an exemplar to demonstrate the complexity and urgency of addressing hydropower siltation where stakeholder participation and co-design will be an essential part of the solution.

Our analysis of sediment sources in the Rapel basin reveals that conversion of steep land from native vegetation to agriculture promotes soil erosion, making agriculture the dominant contributor of sediment to the reservoir. Mining activities also contribute to the total sediment deposited in the lake but here it is the sediment-associated contaminants such as Cu and Mo that pose significant risks to reservoir water and broader environmental quality. Industrial livestock production further intensifies the environmental quality problem, particularly through particulate P input. Our evaluation of these challenges emphasises the need for context-specific, locally tailored solutions that address the unique interplay of factors contributing to soil erosion and downstream siltation.

Addressing sedimentation challenges requires a collaborative participatory approach that engages stakeholders at all levels (Gerrits and Edelenbos, 2004). Stakeholder participation will play a crucial role in co-designing effective sediment management strategies. Engaging stakeholders across sectors fosters a sense of ownership of the problem and enables the integration of multiple perspectives, knowledge, and needs into decision-making processes (Armitage et al., 2010; Jager et al., 2016; Pahl-Wostl et al., 2012). Collaborative platforms, such as participatory workshops, stakeholder dialogues, and multi-stakeholder partnerships, facilitate inclusive and informed discussions, leading to more effective and socially acceptable solutions (Ayala-Orozco et al., 2018; del Valle, 2023; Reed, 2008) and empirical data such as that gathered in this study have a key role to play in eliciting discussion and supporting co-design (Blake et al., 2018b; Kelly et al., 2020).

In the Rapel basin, stakeholder collaboration faces potential challenges through the conflicting interests of agriculture, mining, and environmental conservation. This requires a proactive dialogue and guided negotiation to find mutually beneficial solutions. The basin comprises diverse communities and industrial sector actors with varying economic and socio-cultural contexts, necessitating equitable representation and inclusive decision-making processes (del Valle, 2023). Overcoming these challenges involves building trust, fostering cooperation, and addressing power dynamics among stakeholders. Strengthening governance mechanisms and promoting institutional coordination will also be importance considerations for effective implementation of sediment management strategies in the Rapel basin. Co-designing solutions with stakeholders promotes adaptive management and iterative decision-making, allowing for the development of tailored strategies that consider the unique geomorphological, hydrological, and socio-economic characteristics of the basin (Mussehl et al., 2022).

The research reported in this paper was a part of a broader project in the Rapel Basin that also involved an intervention process guided by the trans-disciplinary method for conducting social transformations called Participatory Innovation Praxis (del Valle, 2023; O'Ryan and Del Valle, 1996). In this process representative actors of the basin generated a consensus understanding of the soil erosion condition and defined a comprehensive strategy to face it over the short, medium and long term. According to this understanding, soil erosion is the main symptom of a social-cultural condition existing because practically no effective action exists in any or the following social dimensions: integrated management of the basin; incentives and promotion of sustainable soil management; cultural transformation towards understanding soil as an ecosystem; dissemination of agricultural and forestry practices that activate soil's life; mechanisms for technological innovation, development and demonstration; fostering legal instruments to promote conservation and sustainable use of soil in Chile; and sufficient and effective enforcement of norms for soil conservation and use in Chile. The strategy specifies 41 social, institutional, and technological innovations in all dimensions, along with a governance system for their participatory design and implementation.

Sediment management in major hydropower basins presents a global challenge exemplified by the Rapel basin (Syvitski et al., 2005). The specific challenges faced here highlight the need for sensitive co-design to produce evidence – led and context-specific solutions that address the complexity and dynamic nature of the drivers within and external to the basin and its communities. Achieving these co-designed solutions, alongside an equitable, stakeholder-led governance system for the basin is the only way to enhance the social acceptability of sediment management strategies. The research outcomes from sediment source apportionment studies have the potential to inform international best practices in sediment management for sustainable hydropower development and ecosystem protection. As a representative exemplar, the Rapel Basin challenge highlights the transferability of such research findings to other hydropower basins globally, reducing the impact of soil erosion on water, food and energy security.

## 4 Conclusions

Evidence of spatio-temporal dynamics of sediment and pollutant source from the Rapel Basin provides an important new framework for evaluating the global hydropower siltation risk. Illuminating the complexity of basin response to land management decisions, and the complex socioeconomic and cultural factors that drive them, demonstrates the clear need for holistic, system thinking approach to derive evidence-led, co-designed solutions.

With the changing global climate and, in particular rainfall patterns and emerging climates of extreme, national and inter-governmental bodies are rightly focusing on adaptation and mitigation strategies to reinforce food, water and energy security through sustainable environmental management. Herein, the interplay between agriculture and mining against the background of natural sediment production in the Rapel Basin is highly relevant. Despite glacial retreat and increase in the intensity of extreme rainfall in the region, the changes in natural sediment production in the high relief Andean sub-catchments are dwarfed by agricultural and mining inputs. The coincidence of observed consequences of conversion of steep lands in the Coastal Mountain range to orchards and vineyards with changes in rainfall patterns threatens a step-change in sediment production. This non-linear response in siltation to rainfall patterns is not readily understood by stakeholders and will be seen to intensify with continued conversion of naturally-vegetated steeplands. In this exemplar case, the complexity of governance issues and the combined concerns of sediment and mining pollution as well as agricultural co-contaminant further demonstrates the need for system-wide participatory solution co-design. The importance of blending scientific evidence with local environmental knowledge to inform an inclusive participatory approach to resolving basin-wide hydropower siltation problems cannot be over-emphasised as we move into new and less predictable climate regimes.

## 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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