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2 hydropower reservoir
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16

17 **Abstract**

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

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

41 **Highlights**

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

47 CRediT:

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

58 **1 Introduction**

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

65 Land degradation is a global problem affecting more than 2 billion people (UNCCD and World Bank,
66 2016). Soil erosion, promoted by multiple factors, has negative impacts on the future of soil fertility, water
67 quality, food safety and power generation (Dercon et al., 2012; Li and Fang, 2016; Owens, 2020). Future land-
68 management decisions and water quality assessments in reservoirs require quantification of the anthropogenic
69 amplification of runoff and soil erosion processes, so that hotspots of soil erosion can be targeted and controlled
70 by mitigation measures (Collins et al., 2020; Poesen, 2018; Walling, 2013). Erosion processes not only lead to
71 loss of critical soil resources, but also the associated transport of contaminants that are bound to soil particles,
72 such as pesticides, heavy metals, radionuclides, and phosphorus, that may have detrimental impacts on the
73 healthy functioning of water bodies such as rivers, wetlands, lakes and estuarine ecosystems (Rodgers et al.,
74 2020; Wharton et al., 2017; Wohl, 2015). Fine sediment ($< 63 \mu\text{m}$) has the potential to travel long distances
75 from source to sink and impact local ecology by reducing water clarity and light penetration (Davies-Colley
76 and Smith, 2001; Thrush et al., 2004). The identification of sediment sources is therefore essential to implement
77 focused mitigation actions. Given the widespread impacts of soil erosion and siltation on the food-water-
78 energy-security nexus (Blake et al., 2021) and the increasingly recognition of multiple co-benefits of natural
79 capital, basin-wide solutions integrating upstream soil erosion and downstream sediment loading processes are
80 required to develop sustainable management plans.

81 The impact of agricultural and forestry operations on sediment yields and consequent water pollution
82 from different catchments, and the effectiveness of potential mitigation measures are traditionally assessed by
83 measuring the sediment yield from the outlets of small experimental catchments that are subject to different
84 levels of disturbance/management practices (Stenberg et al., 2015). Whilst this experimental approach can

85 provide an effective basis for assessing the changes in sediment yields, it cannot provide specific information
86 on the sources of eroded soils, especially when attempting to support collective and integrated decision making
87 at the basin scale. Such information is essential to design and implement effective sediment control strategies
88 and to target mitigation measures in the most cost-effective way. The sediment fingerprinting technique has
89 emerged as a valuable tool in this regard, and it has been applied in a variety of landscapes and settings (Collins
90 et al., 2020; Owens et al., 2016; Walling, 2013). This technique characterises potential sources according to
91 their chemical and physical properties. These properties are also measured in sediment samples collected from
92 areas of accumulation/receiving environments (e.g. dams, rivers, wetlands, lakes, estuaries). Then, potential
93 sources and mixtures data are analysed, and subsequently mixing models are used to estimate the relative
94 proportion of soils in a mixture from the various sources in a contemporary and historical perspective (Collins
95 et al., 2017; Walling, 2013). The tracer properties used must be measurable in both sources and mixtures, be
96 representative of their sources and must also have conservative signatures from their source, during transport
97 and deposition (Collins et al., 2017; Owens et al., 2016; Walling, 2013). Several tracer properties have been
98 used to this purpose: major and trace elements (Blake et al., 2018a; Kitch et al., 2019; Muñoz-Arcos et al.,
99 2021), Fallout Radionuclides (FRNs) (Schuller et al., 2013), mineral magnetism (Blake et al., 2006; Gaspar et
100 al., 2019), colour coefficients (Martinez-Carreras et al., 2010) and compound-specific stable isotopes (Blake et
101 al., 2012; Bravo-Linares et al., 2018). These sediment tracing techniques have been used to determine both
102 contemporary and historical sources of eroded soils and across a range of spatial scales (Bravo-Linares et al.,
103 2020; Wynants et al., 2020). However, its application in large river basins, using a tributary tracing approach,
104 has received little attention to date. This basin scale is critically important when tackling the food-water-energy-
105 environment challenge, especially in the context of major silted hydropower basin projects in the global South
106 (Zhang et al., 2018). The present study aims to close this critical gap.

107 Similarly, in many other Latin American contexts, soil erosion constitutes one of the most significant
108 water pollution problems in Chile. A recent study concluded that 36.8 million hectares of land is subject to
109 some degree of soil degradation, an equivalent to 49% of the national territory and a further ~7 million hectares
110 is subject to severe soil erosion. On the other hand, agriculture in Chile has intensified in the past decades
111 causing changes in the physical properties of soils and consequently creating a potential risk for water erosion
112 (Fleige et al., 2016). Increased erosion can thus augment sediment loads in waterbodies, especially in

113 hydropower reservoirs, affecting water quality, energy production and inducing reservoir siltation (Bronstert et
114 al., 2014). In Chile there are 60 dams, which are widely distributed throughout the national territory. This
115 indeed, has the potential to affect the power generation capacity, as the dams are losing water storage capacity
116 with increasing sedimentation rates (Kondolf et al., 2014). Particularly in central Chile, the area has eroded
117 soils (classified into categories of light, moderate, severe and very severe erosion) of 861 thousand hectares,
118 which represents 52.5% of the regional surface.

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

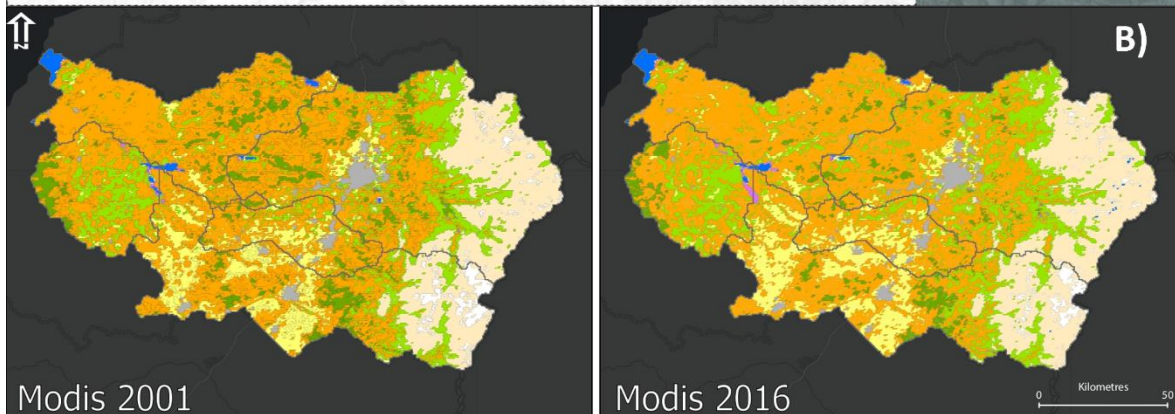
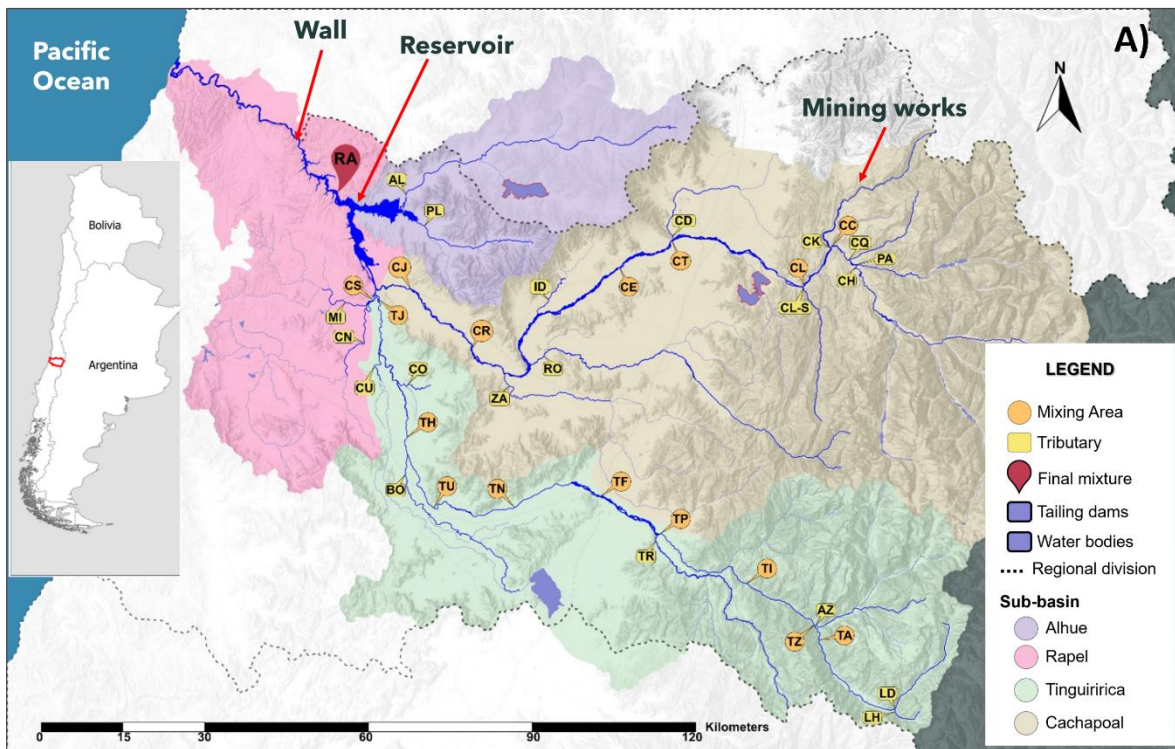
127 **2 Materials and Methods**

128 *2.1 Study Site*

129 This study is based in the Rapel basin, located in Central Chile (VI Region), between the parallels
130 33°53' and 35°01' south latitude, and with a surface area of 13,695 km² (Fig. 1). The annual rain fall by the
131 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
132 significance, during the winter of 2023 (July and August) normal rainfall patterns changed dramatically and
133 two massive flooding events occurred during this period. Here, rainfall concentrated in the Andes produced
134 dramatic changes in river flow leading to floods that left urban and agricultural areas with substantial amounts
135 of deposited material. Average temperatures for the coast are 14°C and 9.6°C by the mountains. The catchment
136 is fed by two main river systems (Cachapoal and Tinguiririca). Cachapoal River is characterised by a nivo-
137 pluvial regime, with an average flow rate of 172 m³ s⁻¹ during winter and 98 m³ s⁻¹ during summer, while the
138 Tinguiririca River has a primarily pluvial regime with average flow rates of 125 m³ s⁻¹ during winter and 46 m³
139 s⁻¹ during summer. Estimated sediment loads, for the Cachapoal and Tinguiririca rivers are approximately 1.11

140 and 0.71 million tons/year, respectively (Pepin et al., 2010). The confluence of these two rivers forms the Rapel
141 River, which flows into an artificial lake system (Rapel Lake) impounded by a dam (built in 1968) with an
142 initial water storage capacity of 695 million m³ and power generation capacity of 350,000 kW.

143 In terms of socio-economic activities, the catchment has been used for hydropower generation since
144 1968. Upstream, intensive agricultural and forestry activities, mining industry, sand and gravel extraction, land
145 use changes of hilly zones and recreational uses have severely affected the water storing capacity of the dam
146 (36% loss by 2010, data provided by national energy company, ENEL). It has been reported that to 2010, 159
147 million m³ of coarse material have reached the delta, and another 18 million m³ of fine sediment have reached
148 the dam's wall (Lecaros Sánchez, 2011); the latter being the input that provokes greater siltation in the reservoir
149 and most severe consequences for hydropower generation. These processes have arguably been enhanced by
150 natural erosion produced by glacial retreat, and the consequences can be seen in the lake environmental quality.
151 Alongside the sediment load issues, water quality has also been affected by runoff from livestock activities
152 resulted in a significant input of particulate phosphorous, generating eutrophication and associated issues (Vila
153 et al., 2000). Moreover, urban development indicated an increase in organic matter and boron (B) (Lacassie and
154 Ruiz-Del-Solar, 2021).



Rapel Catchment: Changes in land cover 2001 - 2016

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

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

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

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

190 2.2 *Sampling strategy to evaluate sediment source dynamics: the tributary tracing approach.*

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

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

205 Approximately 300 sediment samples were collected at different locations in the Rapel lake and its
206 main tributaries during 2019 (Fig. 1): Alhué, Las Palmas, Cachapoal, Tinguiririca and Las Cadenas. At every
207 tributary, 7 to 10 spatially integrated channel bed deposited sediment samples were taken upstream of the
208 confluence where recent sediment deposits were clearly evidenced. With regards to Tinguiririca and Cachapoal
209 sub-catchments, low order tributaries to their higher order mainstem were also sampled before the confluence
210 to capture characteristics of lowland feeder streams (Fig. 1). Surficial sediments were sampled using plastic

211 spatulas to avoid cross contamination, placed in double plastic bags and the central point of each sampling area
212 was recorded using a GPS. Hence, the surficial sediment samples were used in the following context: First:
213 Recently deposited sediments collected at the main rivers and tributaries were employed to characterise the
214 signature of dominant sources within the tributary catchment. Second: Sediment samples collected in the lake
215 (sediment mixing/accumulation zone) were used to estimate the contribution of the different upstream tributary
216 sediments. Of note, Additional sampling points presented in the map were only considered when evaluating the
217 contaminant distribution along the river channel and not for sediment source apportionment.

218 Sediment cores (n = 6) were taken in the upper zone of the reservoir during 2020 (around 12 km from
219 the wall and around 22 km from the confluence of the two main tributaries). The sampling was done in a
220 meander area that was selected as a natural and representative sediment accumulation zone (Fig.1. Lat. -
221 34.113278°; Long. -71.516222°). The average water depth was 30 m. This location represented the maximum
222 feasible depth for coring with depths at the centre of the dam and places near to the wall reaching greater than
223 50 m. The samples were collected from an anchored zodiac using a Ubitech gravity core equipped with acrylic
224 tubes of 2.5 m length and 6 cm of internal diameter. Sediment core depths ranged from 88 to 122 cm. In parallel,
225 surficial sediment samples (3 samples with 3 to 5 sub-samples for analysis) were taken using a Van Veen grab
226 sampler and short-cores (about 10 – 15 cm of sediment depth) to assess contemporary sediment source
227 contributions and spatial variability therein.

228 2.3 *Sample preparation and analysis*

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

234 A subsample of approximately 1 g of dried and sieved particles was used for particle size analysis
235 using a Malvern 2000 series laser granulometer. Herein, 3 ml of 6% Hydrogen Peroxide (H₂O₂) was added to
236 every sample to remove organic matter over a 12-hour period. Subsequently, another aliquot of 6% of H₂O₂
237 was added to each sample to assess if effervescence still took place. Samples were then placed into a water bath

238 at 80°C for 2 to 4 hours and cooled at room temperature. Before laser diffraction analysis, sodium
239 hexametaphosphate was added to every sample as a dispersant. Particle size distribution analyses were
240 performed in triplicate and results were assessed statistically to check for large dispersion between particle size
241 distributions percentiles in which a coefficient of variation < 10 % between percentiles was accepted as a valid
242 result.

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

255 From the six cores taken, the deepest (122 cm) was selected to perform sediment geochronology and
256 thereby estimating sedimentation rates via fallout ²¹⁰Pb (Appleby, 2001). This deepest core was selected for
257 profiling to minimise the risk of disturbance due to seasonal emptying and filling of the lake for
258 recreational/hydropower generation purposes. ²¹⁰Pb, ²¹⁴Pb, ¹³⁷Cs and ²⁴¹Am activity concentrations were
259 measured using a well type HPGe gamma spectrometer. Total ²¹⁰Pb was determined via its gamma emissions
260 at 46.5 keV, ²²⁶Ra (to determine the supported ²¹⁰Pb component) by the 295 keV and 352 keV gamma rays
261 emitted by its daughter isotope ²¹⁴Pb, and ¹³⁷Cs was measured by its emissions at 662 keV (Appleby, 2001).
262 The CRS (Constant Rate of Supply) model (Appleby and Oldfield, 1978) was applied to construct core
263 chronologies and sedimentation rates from activity concentrations. Since the reservoir was constructed after the
264 ¹³⁷Cs peak fallout in the 1960s, there was a lack of an independent chronological marker. The CRS-model dates

265 were benchmarked to the basal core base date by reservoir construction in 1968 and the subsequent
266 chronological estimate was independently evaluated by comparing sedimentation rates patterns through time to
267 floods records from the main tributaries in the upper part of the core (Fig. S1). All samples were analysed at
268 the ISO9001 certified University of Plymouth Consolidated Radioisotope Facility (CoRIF), UK.

269 2.4 *Statistical analysis and mixing modelling*

270 2.4.1 *Tracer selection*

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

289 2.4.2 *Mixing model structure*

290 The mixing model employed was MixSIAR (Stock et al., 2018). The model was run with
291 uninformative prior, error structure as 'residual error' and chain length of 'very long'. Convergence of the model

292 was evaluated using the Gelman-Rubin diagnostic, in which case all variables were below 1.05. This in accord
293 with accepted limits in this field of application (Blake et al., 2018a; Smith et al., 2018). All results are provided
294 as the mean and standard deviations as well as credible intervals from the posterior distributions.

295 **3 Results and discussion**

296 *3.1 Tracer selection*

297 The potential influence of particle sorting effects on sediment fingerprinting properties (e.g. element
298 concentrations) have been widely debated in the literature (Gaspar et al., 2022; Lacey et al., 2017; Smith and
299 Blake, 2014). Differences in grain size between soil and sediment particles can bias mixing model results
300 (Gaspar et al., 2022). To address the potential effects, several approaches have been applied such as sample
301 fractionation and particle size correction factors. However, it has been reported that the use of particle size
302 correction factors can lead to overcorrection (Smith and Blake, 2014). Here, a similar approach to that reported
303 in Smith et al., (2018) and Gaspar et al., (2022) where evaluation of potential grain size effects was carried out.
304 Differences in the median particle size distribution (D50) between tributary and lake sediments were found (see
305 Fig. S2). Therefore, a correlation analysis between the Specific Surface Area (SSA) and elemental
306 concentrations of all tributary samples was performed (Table S2). Only one element (out of 27) showed strong
307 correlation with SSA (i.e. Cr, $R = -0.55$, see Table S2) while four presented moderate association (i.e. V, Mn,
308 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
309 correlations (i.e. Fe, Ni, Zn, Zr, Pb and U. R between -0.2 and -0.3 or 0.2 and 0.3 , $p < 0.05$, $n = 60$).
310 Consequently, due to low association between these elements and SSA, instead of removing them (or going
311 further in particle size effects assessment) we tested them within the tracer selection procedure (see section
312 2.4.1). Here, the inclusion of some weak-to-moderate correlated tracers was not considered likely to increase,
313 or significantly influence the source apportionment results.

314 It is noteworthy that when performing historical sediment source apportionment, it may not be
315 appropriate to use elements coming from pollution point-sources (such as Cu from mining in this study) due to
316 changes in concentrations along the core profile in relation to pollutant dynamics compared to contemporary
317 source samples used to characterise the input. This implies that those elements may be discarded during the
318 tracer selection procedure for some mixtures and included in others. For instance, range test evaluation of Cu

319 concentrations at the top of the core suggest inclusion of this tracer in the modelling, while the opposite occurs
 320 when evaluating the fit of sediment from the bottom (Fig. S3). The influence on the inclusion of Cu in the tracer
 321 set for modelling can be seen in sections 6 and 7 of the core, where results did not differ significantly when Cu
 322 is included (section 7) and not included (section 6) (see Table S5), possibly due to the number of tracers
 323 employed in a single calculation (between 12 to 18). Here, it is expected that one rogue tracer would not
 324 significantly drive or influence mixing model outputs.

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

330 *3.2 Sediment source apportionment*

331 *3.2.1 Contemporary source apportionment*

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

335 **Table 1.** Spatial contemporary source apportionment summary statistics obtained from MixSIAR for
 336 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
		BO	6.3	3.9	0.5 - 15.1
	TJ	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
		AL	12.5	7.8	0.9 - 29.6
		CR	63.2	10.6	39.3 - 81.6
Final Mixture	RA	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

337

338 The Cachapoal river sub-catchment was divided in two calculation sections (Table 1). The upstream
339 portion draining the Andes Mountains was evaluated in the first mixture area in the Cachapoal confluence with
340 Claro River (CL sampling site). Details of tracers selected, discarded, etc. can be seen in Table S4. Results at
341 this point indicated that the dominant source at the CL mixing point was the Coya River (CK) with 81.4 ± 4.7
342 %, followed by the Pangal River (PA), Cachapoal Andes (CH) and Clonqui River (CQ) with contributions of
343 14.2 ± 5.0 , 2.7 ± 1.8 , 1.7 ± 1.7 % respectively. These results were in agreement with tributary groupings
344 observed in Fig. S4.a, where CK samples clustered together with CL-M samples. The main sediment source to
345 this point comes from the tributary that has a strong signal of mining activities with significant influence of Cu
346 (see PCA Fig. S4.a). The next mixing area was the Cachapoal at the “La Ratonera” sampling point (CR). A
347 further point and closest to the confluence of the two main rivers was the Cachapoal at “La Junta” (CJ).
348 However, due to the river being canalised for irrigation, a large proportion of fine sediment bearing flow is
349 diverted into the agricultural area. Subsequently, little water reaches the CJ point. For this reason, the CR
350 sampling point was used to represent the integrated sediment input from upstream (with no river intervention).
351 The results from the mixing model to this point showed that the main sources (tributaries) were the Claro River
352 (RO) with $60.5 \pm 8.9\%$, followed by CL with 17.3 ± 4 % and Claro River of Cauquenes (CL-S) with $9.3 \pm 4.0\%$,
353 the rest were distributed between 3.5 to 5.7 % of contribution (CD, ID and ZA). The dominant contribution of
354 RO sediments to CR mixing point is expected as RO is a river that carries sediments from agricultural areas
355 along the whole central valley and delivers them close to the mixing point (~ 10 km). Similarities in geochemical
356 sediment signatures can be confirmed by looking at the PCA score plot where RO samples are overlapping the
357 CR sample cluster (Fig. S4.b). It is important to emphasise that we considered the previous mixing point CL as
358 a source for the second mixing area evaluated and not the individual tributaries from the first mixing area. Also,

359 the contribution of CL to this point is expected to be highly influenced by the sediments coming from the Coya
360 River (CK) whose contribution was more than 80% to mixing point CL. Nevertheless, when performing the
361 full calculation of all tributaries upstream of CR, the main contributions did not change considerably (e.g. RO
362 from 60.5 to 62.5% data not shown). It is not surprising that the tributary RO is the main source as it flows
363 through the whole Central Valley carrying sediments from the agricultural activities and delivering its sediments
364 very close to the mixing point CR (~16 km).

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

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

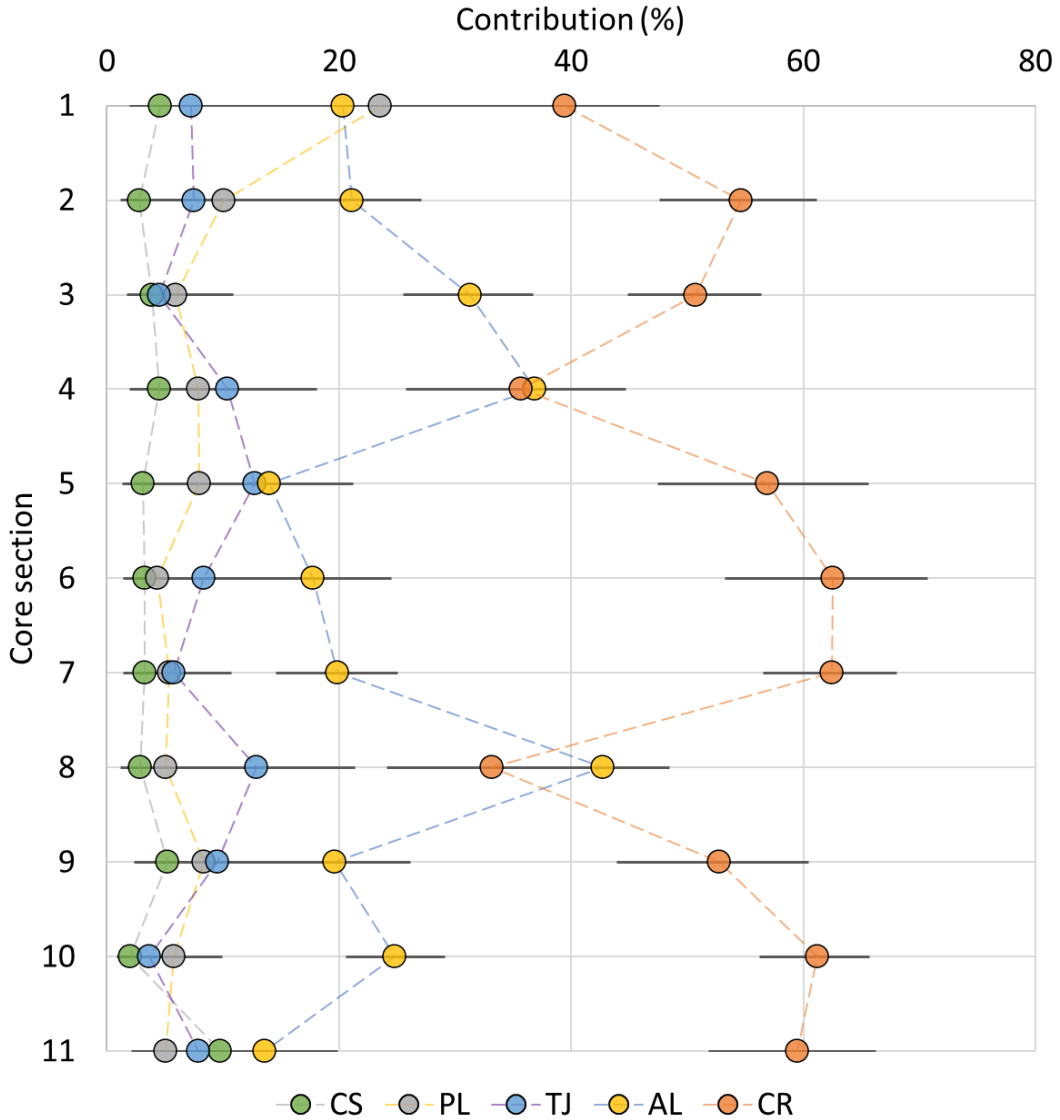
382 For the final mixture zone at the Rapel lake (Fig. 1), the source points were selected from the lowermost
383 sampling point of each sub-basin as a representative composite of the sediments derived from each. These
384 included the sampling points at the two main rivers (CR: Cachapoal river and TJ: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

386 Las Cabras sub-catchment and CS: Las Cadenas stream). Differences in tributaries sediment geochemical
387 composition were clearly observable after the tracer selection procedure (see PCA score plot in Fig. S4.e) which
388 indicates that anthropogenic activities exert a significant influence on the riverine sediment geochemical
389 signatures even in catchments under similar geology (e.g. Alhué and Las Cabras catchments). Apportionment
390 results indicated that the Cachapoal sub-catchment (CR) was the main source to the final mixing area with a
391 $63.2 \pm 10.6\%$, followed by Alhue sub-catchment (AL) with $12.5 \pm 7.8\%$, Las Cabras sub-catchment (PL) with
392 $11.1 \pm 7.6\%$, Tinguiririca sub-catchment (TJ) $7.8 \pm 7.0\%$ and finally Las Cadenas sub-catchment (CS) with 5.5
393 $\pm 4.4\%$.

394 3.2.2 *Temporal variability in sediment source contributions to the lake*

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

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



412

413 **Fig. 2.** Historical tributary source contribution (%) for each core section obtained from MixSIAR mixing
 414 model. Points indicates median relative contribution and error bars 50% credible intervals. Core sections
 415 indicate cumulative depth from bottom: 108-120cm (11) to top 0-6cm (1) with deposition period from 1968-
 416 1976 (11) to 2017-2019 (1). CS: Las Cadenas stream, PL: Las Palmas stream, TJ: Tinguririca river at La Junta,
 417 AL: Alhué river and CR: Cachapoal river at La Ratонера.

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

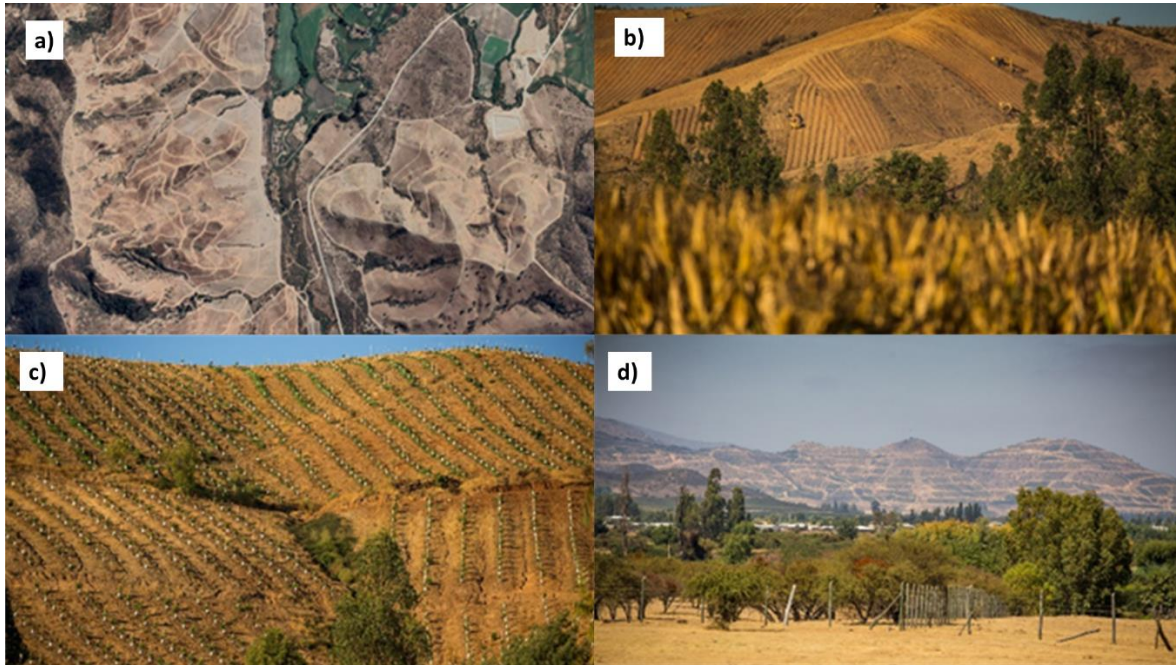
421 Andes have been relatively constant, mining has decreased, and agricultural inputs have increased throughout
422 the years.

423 3.2.3 *Causes of siltation in the Rapel lake.*

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

437 Agricultural activities across the system are also an important source of fine sediment to the lake. For
438 instance, around 60% of CR mixing point contribution comes mainly from agriculture (RO tributary, see section
439 3.2.1) and the historical contribution of PL stream (dominated by agricultural activities) presented an increment
440 in the top two sections of the core (Fig. 2). This contribution goes against local expectation of low input from
441 agricultural land which historically occupies the low gradient/flat terrain of the basin floor. Recent years,
442 however, have seen important land use changes, where steeper terrain of the Coastal Range has been cleared of
443 natural vegetation and planted with vineyards and avocado, olive and citrus trees (Fig. 3a, c and d). These land
444 use changes can be seen on Fig. 1b and c. where croplands have increased by ~7% and forest have decreased
445 ~35%. These changes have been observed predominantly on hilly areas by the Coastal Range (Fig. 3b, c and
446 d). This conversion of steeper terrain from natural vegetation to agriculture, in response to economic drivers to
447 increase production, have generated an important sediment load transferred into the lake. Clearance of the

448 secondary forest of hilly zones and conversion to agricultural use relies on heavy machinery leading to
449 compaction wherein plough lines are required to be parallel to the slope due to the steep gradient and safe
450 operation of machinery (Fig. 3b). Overall, conversion of land from natural to agricultural usage generates areas
451 of bare soil with compacted tracks aligned with the steep slopes leaving a landscape highly prone to erosion.



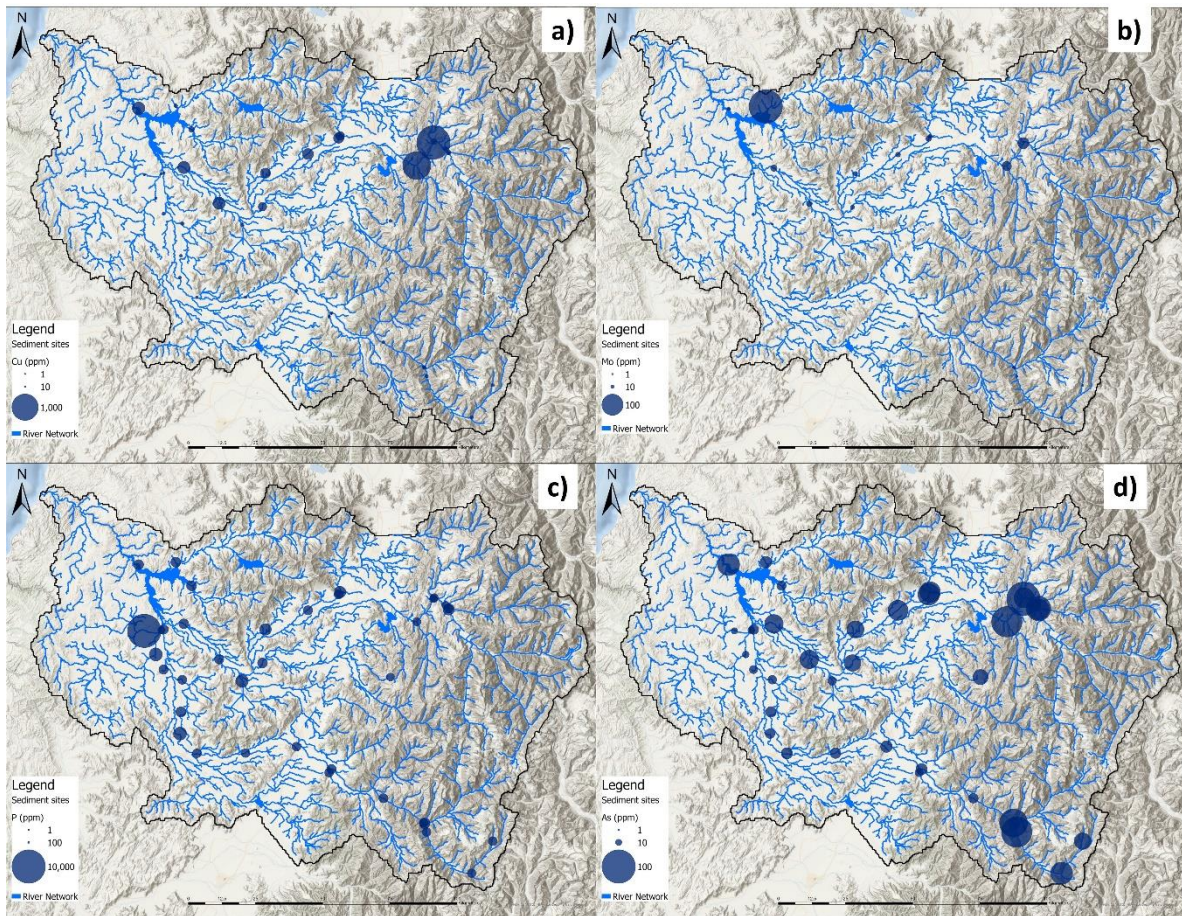
452

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

456 3.3 Spatio-temporal patterns in sediment-associated contaminants

457 Sediment associated contaminants includes a series of chemicals that form part of a legacy of current
458 and past natural and anthropogenic activities within the catchment. Sediment quality is highly influenced by
459 the presence of contaminants in the water (Förstner, 1987). The Rapel river system contains high concentrations
460 of Cu due to mining activities, as well as Mo. In addition, As is also present as a naturally-occurring contaminant
461 associated with volcanic origins and enhanced in riverine sediment by selective transport of fine materials.
462 Phosphorus is a typical contaminant of the area associated to the agricultural activities (Lacassie and Ruiz-Del-
463 Solar, 2021). The spatial distribution of measured elements within the catchment (sediment in tributaries and
464 main river courses, Fig. 4) offers evidence that enhanced Cu concentrations (Fig. 4a) have a dominant tributary

465 source that is likely influenced by mining activities that deliver sediment and leachates. These subsequently
466 interact with sediment along the river Cachapoal from the Andes through to the Coya River, with Cu
467 concentrations averaging $1282 \pm 109 \text{ mg kg}^{-1}$. Concentrations remain high in sediments near to the confluence
468 of this tributary with the main river, values at this point “CL” (distanced around 9 km from the confluence)
469 presented average concentrations of $1218 \pm 87 \text{ mg kg}^{-1}$. Moving downstream, Cu concentrations are diluted
470 through the river Cachapoal and ultimately sediment is released into the lake with concentrations of 453 ± 102
471 mg kg^{-1} (around 3 times less compared with its origin in the mine-affected tributary). Herein, sediment pulses
472 from the tributary source (CK) to the material recovered from the lake covers a distance of 124 km along which
473 a range of additional sediment sources are active and contributing to the mix. In comparison, Cu concentrations
474 in the Tinguiririca river varied from values of 35 to 118 mg kg^{-1} (average $64 \pm 14 \text{ mg kg}^{-1}$). These values can
475 be considered as baseline levels in the wider basin area, as in this river there is no evidence of mining activities.
476 Similarly, the behaviour of Mo appears to be as a secondary product of mining activities (Fig. 4b), with average
477 concentrations in mountain reaches (CK) of $50 \pm 3 \text{ mg kg}^{-1}$. However, Mo concentrations in this case are not
478 exclusively from sources derived from the Andes, as the main source of Mo into the lake comes from another
479 mining activity located in the coastal range area in the Alhue sub-catchment (AL), where concentrations in
480 sediments averaged $162 \pm 74 \text{ mg kg}^{-1}$, possibly derived from a tailing dam that accumulates mining waste for
481 the operations taking place in the Andes. Concentrations in the lake were $10 \pm 2 \text{ mg kg}^{-1}$; at least sixteen time
482 less concentrated than the Alhue river (AL). As stated with Cu, the concentrations of Mo in the Tinguiririca
483 River and tributaries ranged from 1 to 8 mg kg^{-1} (average $3 \pm 1 \text{ mg kg}^{-1}$). These values can also be considered
484 as background levels within the catchment.



485

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

488 Elevated phosphorus concentrations of note appear to emanate from the dryland area (Coastal Range)
 489 with a maximum value of 8255 mg kg^{-1} at the San Miguel stream (MI) sampling point (average value of 5283
 490 $\pm 1335 \text{ mg kg}^{-1}$). This sampling area is located near to ‘pig farms’ and possibly highly influenced by this activity.
 491 Concentrations from the rest of the sampling points within the catchment were similar ($\sim 1400 \text{ mg kg}^{-1}$). The
 492 concentrations detected in the lake averaged $1443 \pm 116 \text{ mg kg}^{-1}$. In a eutrophic lake in China P values ranged
 493 from 544 to 932 mg kg^{-1} indicating the risk of P-related events in this system (Wang and Liang, 2015),
 494 suggesting that the Rapel lake may be also be at risk of eutrophication if P is not bound stably in the sediment
 495 column.

496 Arsenic concentrations (Fig. 4d) were higher in proximity to the Andes with average values for both
 497 rivers of $35 \pm 20 \text{ mg kg}^{-1}$. Values at the lake were around $42 \pm 2 \text{ mg kg}^{-1}$. For the rest of the sampling points in

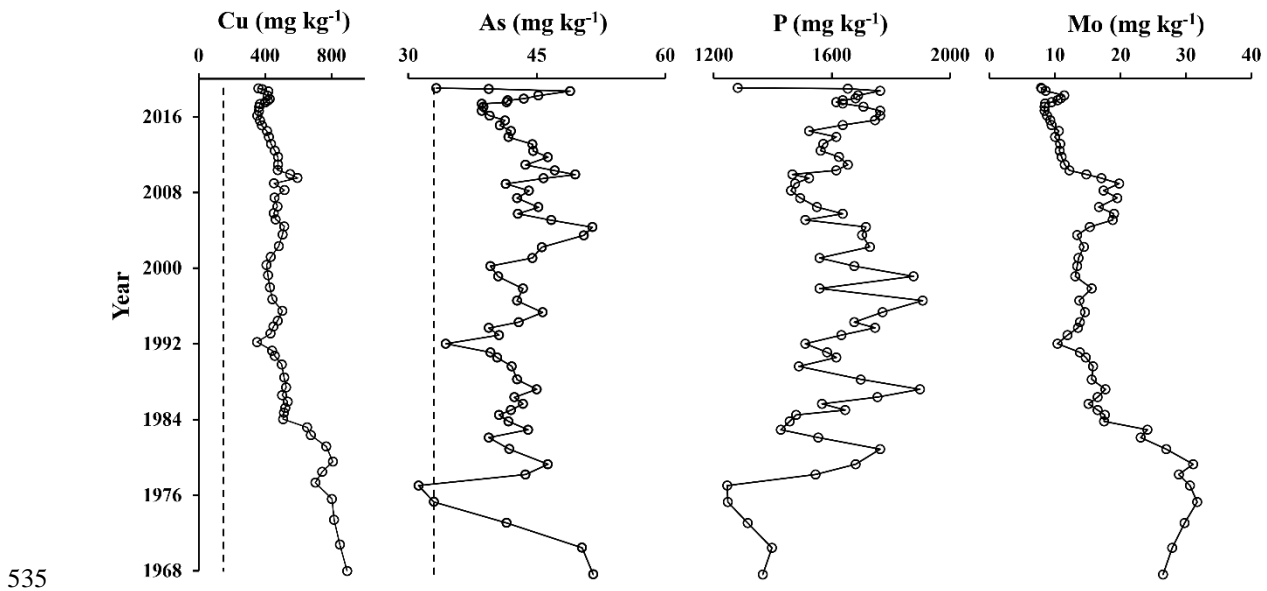
498 both rivers average values were $24 \pm 11 \text{ mg kg}^{-1}$. Therefore, it was concluded that the sources of As in the
499 catchment are derived from naturally occurring As within the Andes (Lacassie and Ruiz-Del-Solar, 2021) with
500 some enhancement by fluvial sorting (Horowitz and Elrick, 1987). Previous studies in the Rapel catchment
501 detected maximum As concentrations of 72 and 21 mg kg^{-1} in the Cachapoal and Tinguiririca Rivers
502 respectively. Both rivers contribute sediment-associated As to the lake due to high background As
503 concentrations in Andean geology and consequently concentrations of this element in fine sediments can reach
504 the lake and influence its accumulation due to natural processes (Lacassie and Ruiz-Del-Solar, 2021), as well
505 as being enhanced by mining (Figure 4d).

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

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

522 Regarding the temporal inputs of sediment-associated contaminants into the lake, the profile for the
523 four elements of interest (Fig. 5) and how they behaved since 1968 offer important temporal perspectives in
524 pollution dynamics in the system. Profiles of Cu and Mo presented strong correlation ($R: 0.92, p < 0.05$), this

525 is expected as those elements demonstrated to come from the same origin (mining activities). Additionally, the
 526 mining company during the year 1985 began to use a different rock that was depleted in Cu and consequently
 527 the amount of Cu present in the mineral was much lower than the one used before 1985. This process can be
 528 clearly seen in the profiles for Cu and Mo with a drop in concentration in the sediment profile since 1985. Also,
 529 the mining activity since 1985 invested in the improvements of their environmental impacts as it is clear that
 530 Cu inputs from 1985 also tended to decrease. Nevertheless, all the values were higher than the PEC for this
 531 element. The same is true for As concentrations where values were over the threshold and relatively constant
 532 throughout the years. Phosphorus, on the other hand, followed a step change upward in the late 1970s, where
 533 concentrations have remained relatively constant from 1981 potentially reflecting sustained but noting also that
 534 the non-conservative properties of this element can also influence profiles (Tiecher et al., 2019).



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

538 *3.4 Challenges and opportunities for sediment management in major hydropower basins*

539 Adopting a holistic ‘systems-thinking’ approach is paramount in tackling hydropower sediment
 540 management challenges that comprise the food-water-energy-environment nexus (Blake et al., 2018b).
 541 Hydropower basins are complex systems where numerous factors, including dynamic change in land use
 542 practice, climatic conditions and hydro-geomorphological process influence sediment dynamics. Understanding

543 these interconnected factors and their interactions, through interdisciplinary and cross-sectoral working, is
544 crucial for devising comprehensive management plans (del Valle, 2023). Collaborative efforts between
545 stakeholders can enhance the effectiveness of sediment management strategies and facilitate sustainable land
546 management decisions. The Rapel River basin serves as an exemplar to demonstrate the complexity and urgency
547 of addressing hydropower siltation where stakeholder participation and co-design will be an essential part of
548 the solution.

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

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

566 In the Rapel basin, stakeholder collaboration faces potential challenges through the conflicting
567 interests of agriculture, mining, and environmental conservation. This requires a proactive dialogue and guided
568 negotiation to find mutually beneficial solutions. The basin comprises diverse communities and industrial sector
569 actors with varying economic and socio-cultural contexts, necessitating equitable representation and inclusive

570 decision-making processes (del Valle, 2023). Overcoming these challenges involves building trust, fostering
571 cooperation, and addressing power dynamics among stakeholders. Strengthening governance mechanisms and
572 promoting institutional coordination will also be importance considerations for effective implementation of
573 sediment management strategies in the Rapel basin. Co-designing solutions with stakeholders promotes
574 adaptive management and iterative decision-making, allowing for the development of tailored strategies that
575 consider the unique geomorphological, hydrological, and socio-economic characteristics of the basin (Mussehl
576 et al., 2022).

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

590 Sediment management in major hydropower basins presents a global challenge exemplified by the
591 Rapel basin (Syvitski et al., 2005). The specific challenges faced here highlight the need for sensitive co-design
592 to produce evidence – led and context-specific solutions that address the complexity and dynamic nature of the
593 drivers within and external to the basin and its communities. Achieving these co-designed solutions, alongside
594 an equitable, stakeholder-led governance system for the basin is the only way to enhance the social acceptability
595 of sediment management strategies. The research outcomes from sediment source apportionment studies have
596 the potential to inform international best practices in sediment management for sustainable hydropower

597 development and ecosystem protection. As a representative exemplar, the Rapel Basin challenge highlights the
598 transferability of such research findings to other hydropower basins globally, reducing the impact of soil erosion
599 on water, food and energy security.

600 **4 Conclusions**

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

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

621

622 **Declaration of Competing Interest**

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

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