2024-01-09

Basin scale sources of siltation in a contaminated hydropower reservoir

Bravo-Linares, C

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

16

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

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.

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

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

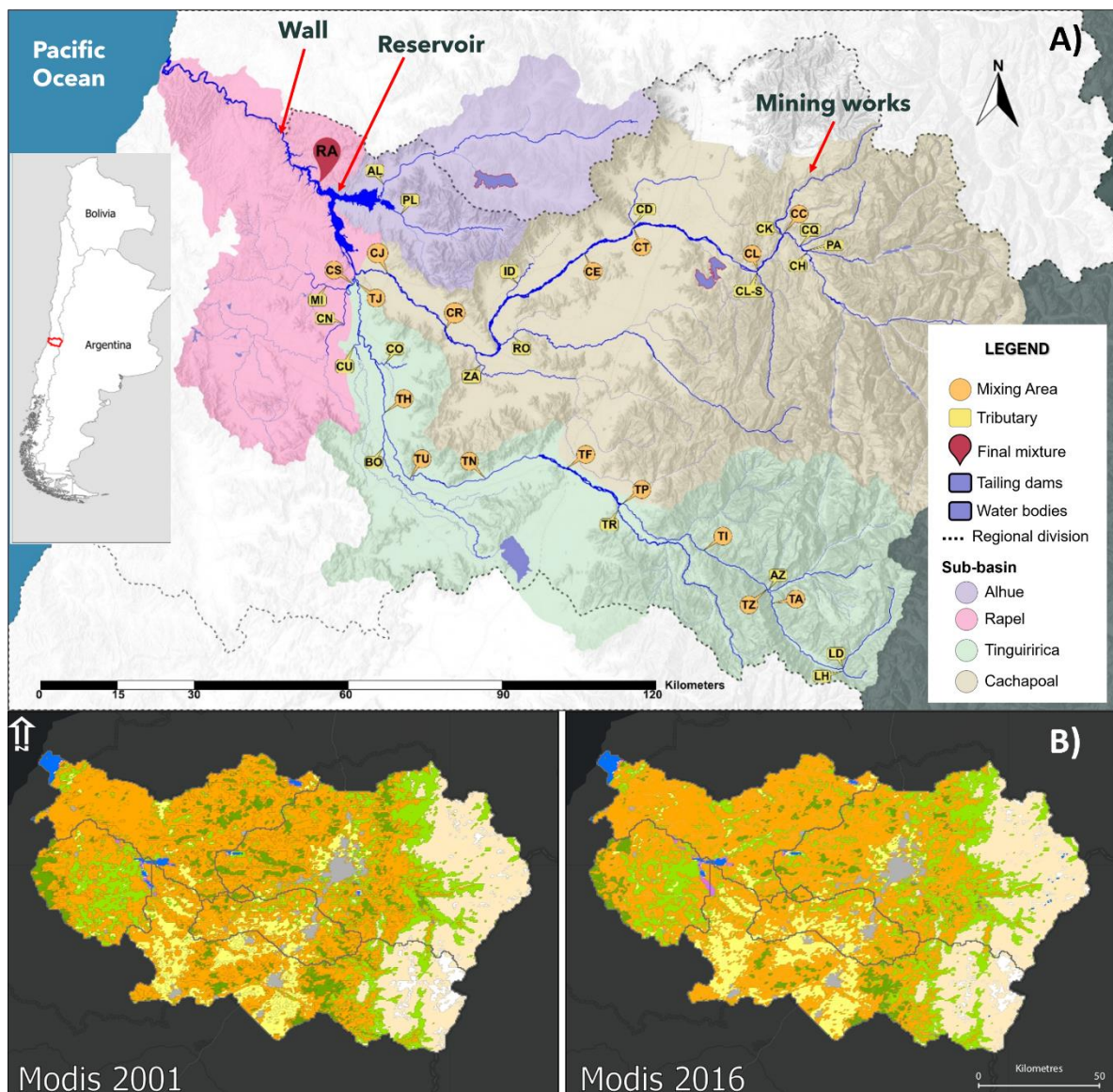126 industrially impacted rivers of similar size across the world.

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 km^2 (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 m^3 s^{-1} during winter and 98 m^3 s^{-1} during summer, while the

138 Tinguiririca River has a primarily pluvial regime with average flow rates of 125 m^3 s^{-1} during winter and 46 m^3

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 m^3 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 m^3 of coarse material have reached the delta, and another 18 million m^3 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).

A)

Pacific Ocean

Wall Reservoir

Mining works

LEGEND

- Mixing Area
- Tributary
- Final mixture
- Tailing dams
- Water bodies
- ···· Regional division

Sub-basin
- Alhue
- Rapel
- Tinguiririca
- Cachapoal

0 15 30 60 90 120 Kilometers

B)

Modis 2001

Modis 2016

0 50 Kilometres

C)

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

155

8

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   Nancahua, TU: Tinguiririca river at Cunaco, BO: Chimbarongo stream, TH: Tinguiririca river at Huique. CO:

166   La Condenada 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 (Laceby 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 (Laceby 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

10

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 (H_2O_2) was added to

236   every sample to remove organic matter over a 12-hour period. Subsequently, another aliquot of 6% of H_2O_2

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 ^{210}Pb (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. ^{210}Pb, ^{214}Pb, ^{137}Cs and ^{241}Am activity concentrations were

259   measured using a well type HPGe gamma spectrometer. Total ^{210}Pb was determined via its gamma emissions

260   at 46.5 keV, ^{226}Ra (to determine the supported ^{210}Pb component) by the 295 keV and 352 keV gamma rays

261   emitted by its daughter isotope ^{214}Pb, and ^{137}Cs 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   ^{137}Cs 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 (Laceby 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.

3   Results and discussion

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; Laceby 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

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
		CH	2.7	1.8	0.1 - 6.9
	CL	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
Cachapoal		CD	3.5	3.2	0.1 - 11.9
		CL	17.3	4.0	8.9 - 24.9
	CR	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
		AZ	32.8	6.3	19.6 - 44.5
Tinguiririca	TF	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

337

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

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

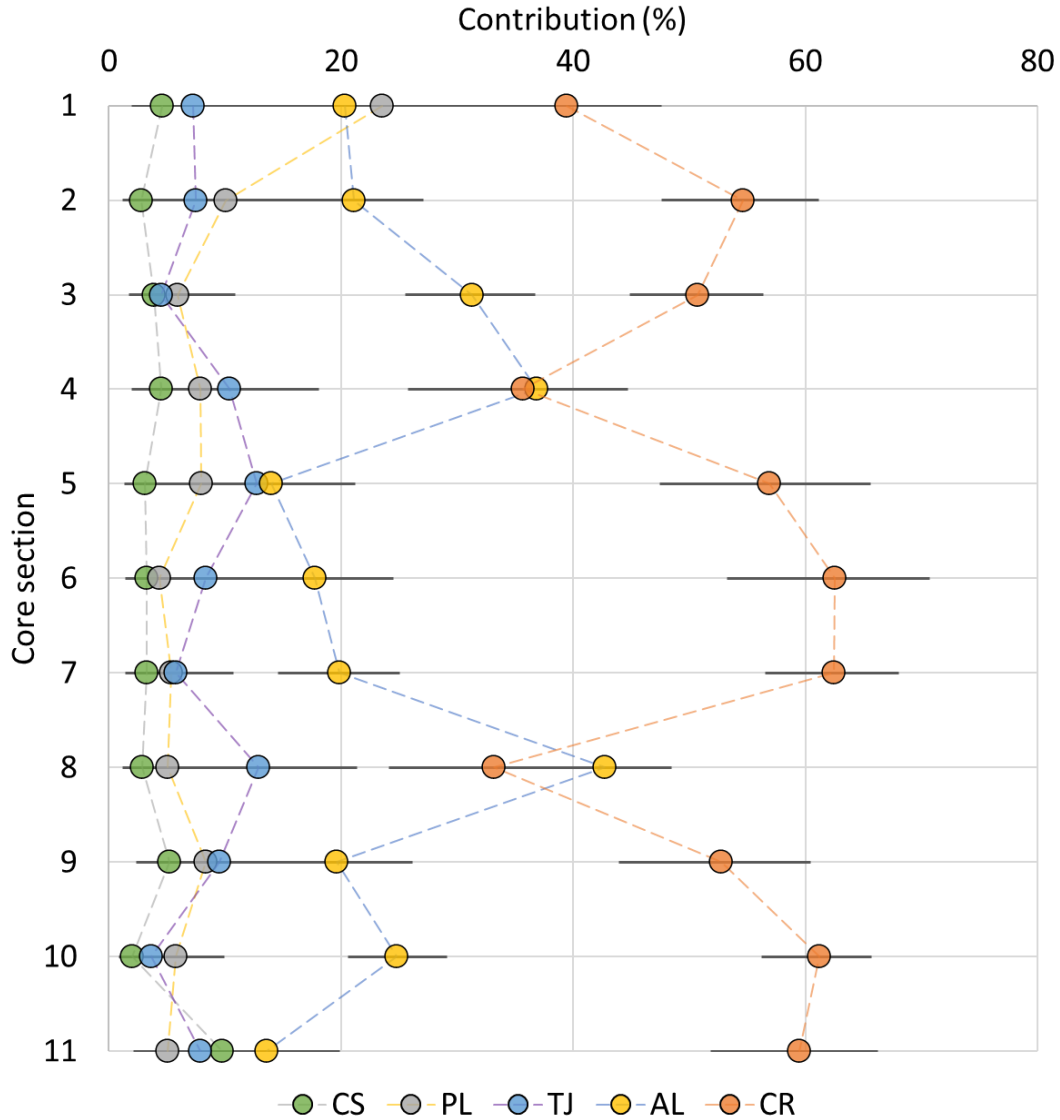411   and Table S3 for summary statistics).

18

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

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 m^3 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

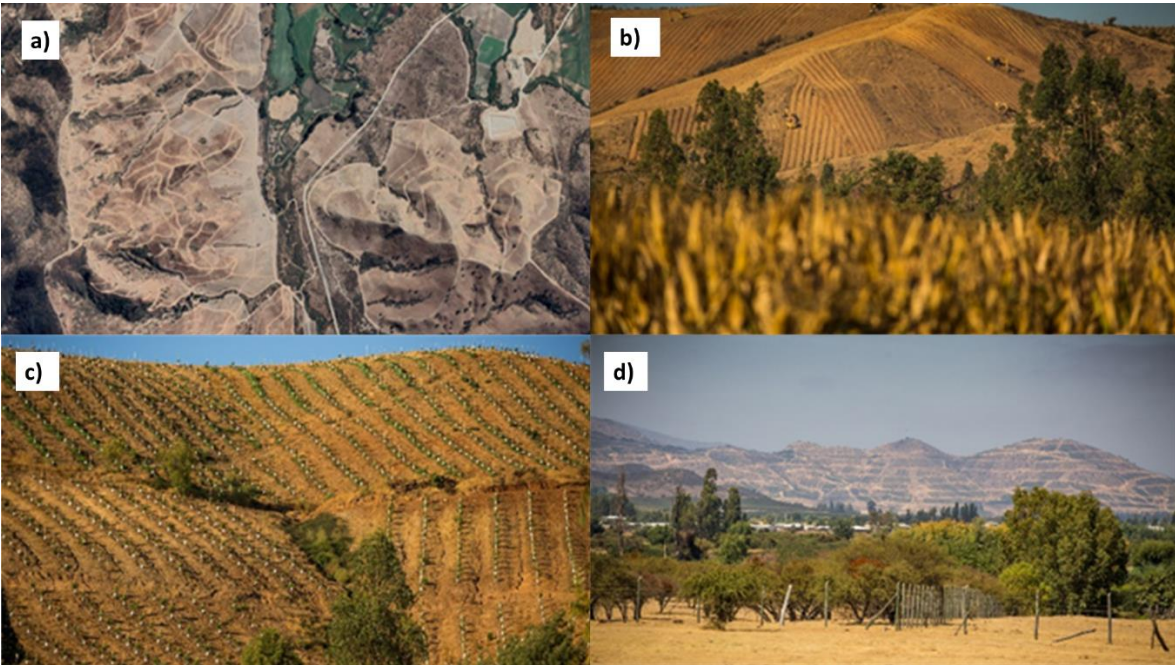451   of bare soil with compacted tracks aligned with the steep slopes leaving a landscape highly prone to erosion.

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 ± 109 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 ± 87 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 ± 14 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 ± 3 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 ± 74 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 ± 2 mg kg^{-1}; at least sixteen time

482   less concentrated that 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 ± 1 mg kg^{-1}). These values can also be considered

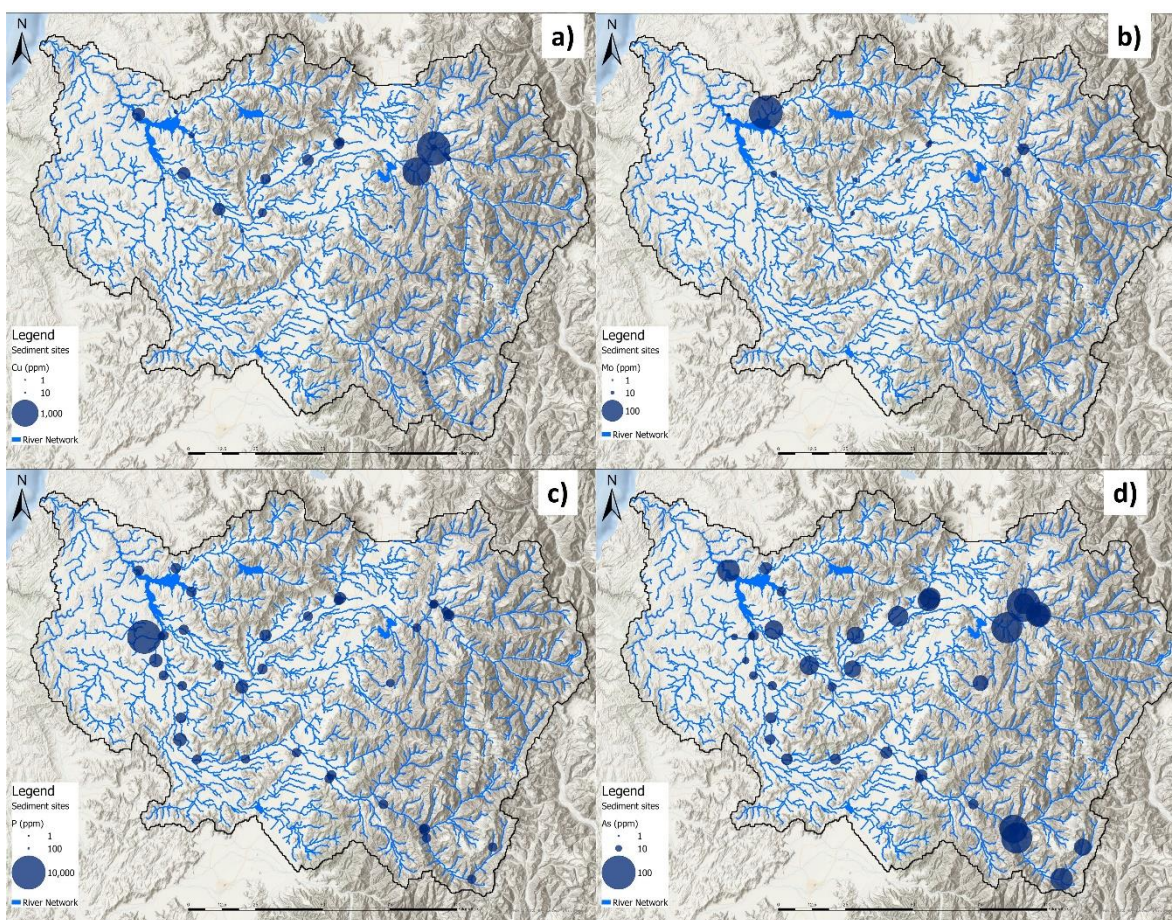484   as background levels within the catchment.

485

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

498　　both rivers average values were 24 ± 11 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 stablish 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-bases 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).
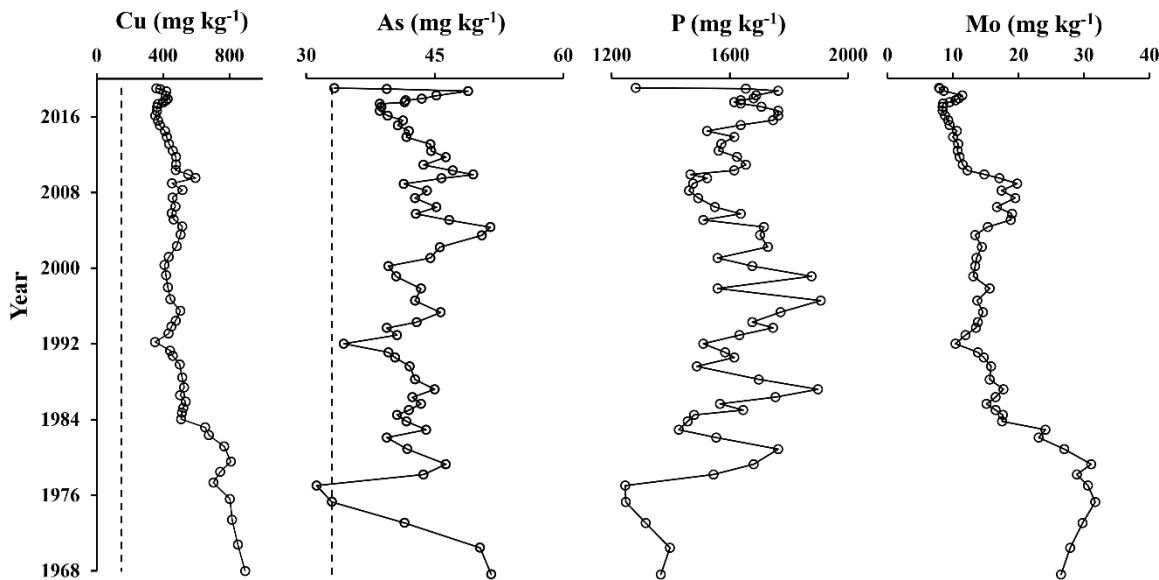
535

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.

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 steeplands. 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

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.

Acknowledgments

5   References

Aitken, D., Rivera, D., Godoy-Faundez, A., Holzapfel, E., 2016. Water Scarcity and the Impact of the Mining and Agricultural Sectors in Chile. Sustainability 8, 128. https://doi.org/10.3390/su8020128

Appleby, P.G., 2001. Chronostratigraphic Techniques in Recent Sediments, in: Last, W.M., Smol, J.P. (Eds.), Tracking Environmental Change Using Lake Sediments: Basin Analysis, Coring, and Chronological Techniques. Springer Netherlands, Dordrecht, pp. 171–203. https://doi.org/10.1007/0-306-47669-X_9

Appleby, P.G., Oldfield, F., 1978. The calculation of lead-210 dates assuming a constant rate of supply of unsupported 210Pb to the sediment. CATENA 5, 1–8. https://doi.org/10.1016/S0341-8162(78)80002-2

Armitage, D., Berkes, F., Doubleday, N., 2010. Adaptive co-management: collaboration, learning, and multi-level governance. UBC Press.

Ayala-Orozco, B., Rosell, J.A., Merçon, J., Bueno, I., Alatorre-Frenk, G., Langle-Flores, A., Lobato, A., 2018. Challenges and strategies in place-based multi-stakeholder collaboration for sustainability: Learning from experiences in the Global South. Sustainability 10, 3217.

Blake, W.H., Boeckx, P., Stock, B.C., Smith, H.G., Bode, S., Upadhayay, H.R., Gaspar, L., Goddard, R., Lennard, A.T., Lizaga, I., Lobb, D.A., Owens, P.N., Petticrew, E.L., Kuzyk, Z.Z.A., Gari, B.D., Munishi, L., Mtei, K., Nebiyu, A., Mabit, L., Navas, A., Semmens, B.X., 2018a. A deconvolutional Bayesian mixing model approach for river basin sediment source apportionment. Sci Rep 8, 13073. https://doi.org/10.1038/s41598-018-30905-9

Blake, W.H., Ficken, K.J., Taylor, P., Russell, M.A., Walling, D.E., 2012. Tracing crop-specific sediment sources in agricultural catchments. Geomorphology 139–140, 322–329. https://doi.org/10.1016/j.geomorph.2011.10.036

Blake, W.H., Kelly, C., Wynants, M., Patrick, A., Lewin, S., Lawson, J., Nasolwa, E., Page, A., Nasseri, M., Marks, C., Gilvear, D., Mtei, K., Munishi, L., Ndakidemi, P., 2021. Integrating land-water-people connectivity concepts across disciplines for co-design of soil erosion solutions. Land Degradation & Development 32, 3415–3430. https://doi.org/10.1002/ldr.3791

Blake, W.H., Rabinovich, A., Wynants, M., Kelly, C., Nasseri, M., Ngondya, I., Patrick, A., Mtei, K., Munishi, L., Boeckx, P., 2018b. Soil erosion in East Africa: an interdisciplinary approach to realising pastoral land management change. Environmental Research Letters 13, 124014.

Blake, W.H., Wallbrink, P.J., Doerr, S.H., Shakesby, R.A., Humphreys, G.S., 2006. Magnetic enhancement in wildfire-affected soil and its potential for sediment-source ascription. Earth Surf. Process. Landf. 31, 249–264. https://doi.org/10.1002/esp.1247

Bravo-Linares, C., Schuller, P., Castillo, A., Ovando-Fuentealba, L., Munoz-Arcos, E., Alarcon, O., de los Santos-Villalobos, S., Cardoso, R., Muniz, M., dos Anjos, R.M., Bustamante-Ortega, R., Dercon, G., 2018. First use of a compound-specific stable isotope (CSSI) technique to trace sediment transport in upland forest catchments of Chile. Sci. Total Environ. 618, 1114–1124. https://doi.org/10.1016/j.scitotenv.2017.09.163

Bravo-Linares, C., Schuller, P., Castillo, A., Salinas-Curinao, A., Ovando-Fuentealba, L., Muñoz-Arcos, E., Swales, A., Gibbs, M., Dercon, G., 2020. Combining isotopic techniques to assess historical sediment delivery in a forest catchment in central Chile. Journal of Soil Science and Plant Nutrition 20, 83–94. https://doi.org/10.1007/s42729-019-00103-1

Bronstert, A., de Araújo, J.-C., Batalla, R.J., Costa, A.C., Delgado, J.M., Francke, T., Foerster, S., Guentner, A., López-Tarazón, J.A., Mamede, G.L., Medeiros, P.H., Mueller, E., Vericat, D., 2014. Process-based modelling of erosion, sediment transport and reservoir siltation in mesoscale semi-arid catchments. Journal of Soils and Sediments 14, 2001–2018. https://doi.org/10.1007/s11368-014-0994-1

Collins, A.L., Blackwell, M., Boeckx, P., Chivers, C.-A., Emelko, M., Evrard, O., Foster, I., Gellis, A., Gholami, H., Granger, S., Harris, P., Horowitz, A.J., Laceby, J.P., Martinez-Carreras, N., Minella, J., Mol, L., Nosrati, K., Pulley, S., Silins, U., da Silva, Y.J., Stone, M., Tiecher, T., Upadhayay, H.R., Zhang, Y., 2020. Sediment source fingerprinting: benchmarking recent outputs, remaining challenges and emerging themes. Journal of Soils and Sediments 20, 4160–4193. https://doi.org/10.1007/s11368-020-02755-4

Collins, A.L., Pulley, S., Foster, I.D.L., Gellis, A., Porto, P., Horowitz, A.J., 2017. Sediment source fingerprinting as an aid to catchment management: A review of the current state of knowledge and a methodological decision-tree for end-users. J. Environ. Manage. 194, 86–108. https://doi.org/10.1016/j.jenvman.2016.09.075

Davies-Colley, R.J., Smith, D.G., 2001. Turbidity, suspended sediment, and water clarity: A review. J. Am. Water Resour. Assoc. 37, 1085–1101. https://doi.org/10.1111/j.1752-1688.2001.tb03624.x

del Valle, A., 2023. Participatory Innovation Praxis: A trans-disciplinary method for conducting high-complexity social transformations. Societal Impacts 100004.

Dercon, G., Mabit, L., Hancock, G., Nguyen, M.L., Dornhofer, R., Bacchi, O.O.S., Benmansour, M., Bernard, C., Froehlich, W., Golosov, V.N., Haciyakupoglu, S., Hai, P.S., Klik, A., Li, Y., Lobb, D.A., Onda, Y., Popa, N., Rafiq, M., Ritchie, J.C., Schuller, P., Shakhashiro, A., Wallbrink, P., Walling, D.E., Zapata, F., Zhang, X., 2012. Fallout radionuclide-based techniques for assessing the impact of soil conservation measures on erosion control and soil quality: an overview of the main lessons learnt under an FAO/IAEA Coordinated Research Project. J. Environ. Radioact. 107, 78–85. https://doi.org/10.1016/j.jenvrad.2012.01.008

Fleige, H., Beck-Broichsitter, S., Dorner, J., Goebel, M.-O., Bachmann, J., Horn, R., 2016. Land use and soil development in southern Chile: Effects on physical properties. J. Soil Sci. Plant Nutr. 16, 818–831.

Förstner, U., 1987. Sediment-associated contaminants—an overview of scientific bases for developing remedial options. Hydrobiologia 149, 221–246.

Gaspar, L., Blake, W.H., Lizaga, I., Latorre, B., Navas, A., 2022. Particle size effect on geochemical composition of experimental soil mixtures relevant for unmixing modelling. Geomorphology 403, 108178. https://doi.org/10.1016/j.geomorph.2022.108178

Gaspar, L., Lizaga, I., Blake, W.H., Latorre, B., Quijano, L., Navas, A., 2019. Fingerprinting changes in source contribution for evaluating soil response during an exceptional rainfall in Spanish pre-pyrenees. Journal of Environmental Management 240, 136–148. https://doi.org/10.1016/j.jenvman.2019.03.109

Gerrits, L., Edelenbos, J., 2004. Management of sediments through stakeholder involvement. Journal of Soils and Sediments 4, 239–246. https://doi.org/10.1007/BF02991120

Haddadchi, A., Ryder, D.S., Evrard, O., Olley, J., 2013. Sediment fingerprinting in fluvial systems: review of tracers, sediment sources and mixing models. International Journal of Sediment Research 28, 560–578. https://doi.org/10.1016/S1001-6279(14)60013-5

Horowitz, A.J., Elrick, K.A., 1987. The relation of stream sediment surface area, grain size and composition to trace element chemistry. Applied Geochemistry 2, 437–451. https://doi.org/10.1016/0883-2927(87)90027-8

Jager, N.W., Challies, E., Kochskämper, E., Newig, J., Benson, D., Blackstock, K., Collins, K., Ernst, A., Evers, M., Feichtinger, J., 2016. Transforming European water governance? Participation and river basin management under the EU Water Framework Directive in 13 member states. Water 8, 156.

Kelly, C., Wynants, M., Munishi, L.K., Nasseri, M., Patrick, A., Mtei, K.M., Mkilema, F., Rabinovich, A., Gilvear, D., Wilson, G., 2020. 'Mind the Gap': reconnecting local actions and multi-level policies to bridge the governance gap. An example of soil erosion action from East Africa. Land 9, 352.

Kitch, J.L., Phillips, J., Peukert, S., Taylor, A., Blake, W.H., 2019. Understanding the geomorphic consequences of enhanced overland flow in mixed agricultural systems: sediment fingerprinting demonstrates the need for integrated upstream and downstream thinking. J. Soils Sediments 19, 3319–3331. https://doi.org/10.1007/s11368-019-02378-4

Kondolf, G.M., Gao, Y., Annandale, G.W., Morris, G.L., Jiang, E., Zhang, J., Cao, Y., Carling, P., Fu, K., Guo, Q., 2014. Sustainable sediment management in reservoirs and regulated rivers: Experiences from five continents. Earth's Future 2, 256–280.

Lacassie, J.P., Ruiz-Del-Solar, J., 2021. Integrated mineralogical and geochemical study of the Rapel fluvial system, central Chile: An application of multidimensional analysis to river sedimentation. Journal of South American Earth Sciences 109, 103289. https://doi.org/10.1016/j.jsames.2021.103289

Laceby, J.P., Evrard, O., Smith, H.G., Blake, W.H., Olley, J.M., Minella, J.P.G., Owens, P.N., 2017. The challenges and opportunities of addressing particle size effects in sediment source fingerprinting: A review. Earth-Sci. Rev. 169, 85–103. https://doi.org/10.1016/j.earscirev.2017.04.009

Lecaros Sánchez, M.H., 2011. Estudio de sedimentación en el embalse Rapel (Undergraduate dissertation). Universidad de Chile, https://repositorio.uchile.cl/handle/2250/104338.

Li, Z., Fang, H., 2016. Impacts of climate change on water erosion: A review. Earth-Science Reviews 163, 94–117. https://doi.org/10.1016/j.earscirev.2016.10.004

MacDonald, D.D., Ingersoll, C.G., Berger, T.A., 2000. Development and evaluation of consensus-based sediment quality guidelines for freshwater ecosystems. Archives of environmental contamination and toxicology 39, 20–31.

Martinez-Carreras, N., Krein, A., Udelhoven, T., Gallart, F., Iffly, J.F., Hoffmann, L., Pfister, L., Walling, D.E., 2010. A rapid spectral-reflectance-based fingerprinting approach for documenting suspended sediment sources during storm runoff events. J. Soils Sediments 10, 400–413. https://doi.org/10.1007/s11368-009-0162-1

Muñoz-Arcos, E., Castillo, A., Cuevas-Aedo, A., Ovando-Fuentealba, L., Taylor, A., Bustamante-Ortega, R., Blake, W.H., Bravo-Linares, C., 2021. Sediment source apportionment following wildfire in an upland commercial forest catchment. Journal of Soils and Sediments 21, 2432–2449. https://doi.org/10.1007/s11368-021-02943-w

Mussehl, M.L., Horne, A.C., Webb, J.A., Poff, N.L., 2022. Purposeful stakeholder engagement for improved environmental flow outcomes. Frontiers in Environmental Science 9, 749864.

O'Ryan, R., Del Valle, A., 1996. Managing air quality in Santiago: what needs to be done? Estudios de Economía 23, 155–191.

Owens, P.N., 2020. Soil erosion and sediment dynamics in the Anthropocene: a review of human impacts during a period of rapid global environmental change. J. Soils Sediments 20, 4115–4143. https://doi.org/10.1007/s11368-020-02815-9

Owens, P.N., Blake, W.H., Gaspar, L., Gateuille, D., Koiter, A.J., Lobb, D.A., Petticrew, E.L., Reiffarth, D.G., Smith, H.G., Woodward, J.C., 2016. Fingerprinting and tracing the sources of soils and sediments: Earth and ocean science, geoarchaeological, forensic, and human health applications. Earth-Sci. Rev. 162, 1–23. https://doi.org/10.1016/j.earscirev.2016.08.012

Pahl-Wostl, C., Lebel, L., Knieper, C., Nikitina, E., 2012. From applying panaceas to mastering complexity: toward adaptive water governance in river basins. Environmental Science & Policy 23, 24–34.

Pepin, E., Carretier, S., Guyot, J.-L., Escobar, F., 2010. Specific suspended sediment yields of the Andean rivers of Chile and their relationship to climate, slope and vegetation. Hydrological Sciences Journal–Journal des Sciences Hydrologiques 55, 1190–1205.

Perera, D., Williams, S., Smakhtin, V., 2023. Present and future losses of storage in large reservoirs due to sedimentation: a country-wise global assessment. Sustainability 15, 219.

Pizarro, J., Rubio, M.A., Castillo, X., 2003. Study of chemical speciation in sediments: An approach to vertical metals distribution in Rapel reservoir (Chile). Journal of the Chilean Chemical Society 48, 45–50.

Poesen, J., 2018. Soil erosion in the Anthropocene: Research needs. Earth Surface Processes and Landforms 43, 64–84. https://doi.org/10.1002/esp.4250

Reed, M.S., 2008. Stakeholder participation for environmental management: a literature review. Biological conservation 141, 2417–2431.

Rodgers, K., McLellan, I., Peshkur, T., Williams, R., Tonner, R., Knapp, C.W., Henriquez, F.L., Hursthouse, A.S., 2020. The legacy of industrial pollution in estuarine sediments: spatial and temporal variability implications for ecosystem stress. Environmental Geochemistry & Health 42, 1057–1068.

Rossel, V., de la Fuente, A., 2015. Assessing the link between environmental flow, hydropeaking operation and water quality of reservoirs. Ecological Engineering 85, 26–38.

Schuller, P., Walling, D.E., Iroume, A., Quilodran, C., Castillo, A., Navas, A., 2013. Using Cs-137 and Pb-210(ex) and other sediment source fingerprints to document suspended sediment sources in small forested catchments in south-central Chile. J. Environ. Radioact. 124, 147–159. https://doi.org/10.1016/j.jenvrad.2013.05.002

Schulz, J.J., Cayuela, L., Echeverria, C., Salas, J., Rey Benayas, J.M., 2010. Monitoring land cover change of the dryland forest landscape of Central Chile (1975–2008). Applied Geography 30, 436–447. https://doi.org/10.1016/j.apgeog.2009.12.003

Smith, H.G., Blake, W.H., 2014. Sediment fingerprinting in agricultural catchments: A critical re-examination of source discrimination and data corrections. Geomorphology 204, 177–191. https://doi.org/10.1016/j.geomorph.2013.08.003

Smith, H.G., Karam, D.S., Lennard, A.T., 2018. Evaluating tracer selection for catchment sediment fingerprinting. Journal of Soils and Sediments 18, 3005–3019. https://doi.org/10.1007/s11368-018-1990-7

Stenberg, L., Finer, L., Nieminen, M., Sarkkola, S., Koivusalo, H., 2015. Quantification of ditch bank erosion in a drained forested catchment. Boreal Environ. Res. 20, 1–18.

Stock, B.C., Jackson, A.L., Ward, E.J., Parnell, A.C., Phillips, D.L., Semmens, B.X., 2018. Analyzing mixing systems using a new generation of Bayesian tracer mixing models. PeerJ 6, e5096. https://doi.org/10.7717/peerj.5096

798 Syvitski, J.P., Vörösmarty, C.J., Kettner, A.J., Green, P., 2005. Impact of humans on the flux of terrestrial
799   sediment to the global coastal ocean. science 308, 376–380.
800 Thrush, S.F., Hewitt, J.E., Cummings, V., Ellis, J.I., Hatton, C., Lohrer, A., Norkko, A., 2004. Muddy waters:
801   elevating sediment input to coastal and estuarine habitats. Front. Ecol. Environ. 2, 299–306.
802   https://doi.org/10.2307/3868405
803 Tiecher, T., Ramon, R., Laceby, J.P., Evrard, O., Minella, J.P.G., 2019. Potential of phosphorus fractions to
804   trace sediment sources in a rural catchment of Southern Brazil: Comparison with the conventional
805   approach based on elemental geochemistry. Geoderma 337, 1067–1076.
806   https://doi.org/10.1016/j.geoderma.2018.11.011
807 UNCCD, World Bank, 2016. Land for life. Create Wealth, Transform lives.
808 Upadhayay, H.R., Bodé, S., Griepentrog, M., Bajracharya, R.M., Blake, W., Cornelis, W., Boeckx, P., 2018.
809   Isotope mixing models require individual isotopic tracer content for correct quantification of sediment
810   source contributions. Hydrological Processes 32, 981–989. https://doi.org/10.1002/hyp.11467
811 Vila, I., Contreras, M., Montecino, V., Pizarro, J., Adams, D.D., 2000. Rapel: A 30 years temperate reservoir.
812   Eutrophication or contamination? Ergebnisse der Limnologie 55, 31–44.
813 Walling, D.E., 2013. The evolution of sediment source fingerprinting investigations in fluvial systems. J. Soils
814   Sediments 13, 1658–1675. https://doi.org/10.1007/s11368-013-0767-2
815 Wang, L., Liang, T., 2015. Distribution characteristics of phosphorus in the sediments and overlying water of
816   Poyang lake. PLoS One 10, e0125859. https://doi.org/10.1371/journal.pone.0125859
817 Wharton, G., Mohajeri, S.H., Righetti, M., 2017. The pernicious problem of streambed colmation: a multi-
818   disciplinary reflection on the mechanisms, causes, impacts, and management challenges: The
819   pernicious problem of streambed colmation. Wiley interdisciplinary reviews. Water 4, e1231.
820   https://doi.org/10.1002/wat2.1231
821 Wohl, E., 2015. Legacy effects on sediments in river corridors. Earth-science reviews 147, 30–53.
822   https://doi.org/10.1016/j.earscirev.2015.05.001
823 Wynants, M., Millward, G., Patrick, A., Taylor, A., Munishi, L., Mtei, K., Brendonck, L., Gilvear, D., Boeckx,
824   P., Ndakidemi, P., Blake, W.H., 2020. Determining tributary sources of increased sedimentation in
825   East-African Rift Lakes. Science of The Total Environment 717, 137266.
826   https://doi.org/10.1016/j.scitotenv.2020.137266
827 Zhang, X., Li, H.-Y., Deng, Z.D., Ringler, C., Gao, Y., Hejazi, M.I., Leung, L.R., 2018. Impacts of climate
828   change, policy and Water-Energy-Food nexus on hydropower development. Renewable Energy 116,
829   827–834. https://doi.org/10.1016/j.renene.2017.10.030
830