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1 Afforestation of degraded grasslands reduces sediment transport and
2 may contribute to streamflow regulation in small catchments in the short-run

3

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19 1. Abstract

20 Land use change in the Southern Grasslands biome with the introduction of exotic, fast-growing forest
21 species is a controversial topic, because of the potential effect on water and soil resources. The aim
22 of this study was to assess the effects of afforesting degraded grassland on streamflow and
23 sedimentation in subtropical headwater catchments. Two small, headwater catchments were studied,
24 one planted with *Eucalyptus saligna* (EC: 0.83 km²) and another with grasslands and extensive
25 livestock (GC: 1.10 km²). Rainfall, runoff, and sediment discharge were monitored from September
26 2013 to March 2017. The results show two-fold greater surface runoff and sediment yield occurred
27 in GC than in EC. Maximum and mean runoff coefficients were, respectively, 45.5 and 10.2% in GC,
28 and 12.4 and 2.2% in EC; suspended sediment yield was, respectively, 67.9 and 22.4 Mg km⁻² in GC
29 and EC; and bed load sediment yield was only 0.053 and 0.006 Mg km⁻². *El Niño* year with high
30 rainfall produced the greatest annual sediment yield, with values 4.2 times greater in GC (167.7 Mg
31 km⁻²) than in EC (39.9 Mg km⁻²). Hydrographs/sedimentographs synchronization and hysteresis
32 patterns suggest faster sediment delivery in GC than in EC, but further studies are needed to conciliate
33 sediment sources results with sediment fingerprinting. Staggered forest harvest (21% of the planted
34 eucalyptus) led to sediment yield comparable to the observed during the pre-harvest period, and lower
35 than in grassland catchment. In conclusion, well-managed forest plantations, including staggered
36 forest harvesting and preservation of riparian forest, are less prone to sedimentation than degraded
37 grassland under intensive grazing. Well-managed afforestation of degraded lands contributes to soil
38 conservation in the studied region, whereas low ecological performance of degraded grassland calls
39 for soil and pasture management practices to increase forage offer to grazing animals, while
40 conserving soil and water resources in the catchments.

41

42 **Keywords:** Forest hydrology; land use change; runoff; erosion; sediment yield.

43

44

45 2. Introduction

46 One of the largest complexes of grasslands in South America is referred to as the Río de la
47 Plata Grasslands (750,000 km²) located in central-east Argentina, southern Brazil, and Uruguay
48 (Paruelo et al., 2007). In Brazil, the Pampa or Southern Grasslands biome is found in the state of Rio
49 Grande do Sul (RS), where it covers 63% of the state territory, which corresponds to 2% of the
50 national territory (MMA, 2018). This biome consists of large extensions of natural grassland, in which
51 the floristic matrix is composed of forest formations interspersed along watercourses (Boldrini et al.,
52 2010; Andriolo et al., 2017). The Pampa is the least protected biome in Brazil, with only 0.4 % under
53 strictly protected areas (Jenkins et al., 2015). Livestock production is one of the main economic
54 activities since the Iberian Colonization, when the Jesuits introduced the cattle around the year 1634
55 (MMA, 2018).

56 Continuous, extensive overgrazing on natural grasslands contributes to the low pasture
57 productivity, which may have deleterious economic (productivity decrease), social (poverty) and
58 environmental consequences (soil and water degradation). Under extensive livestock production, the
59 farming system has low productivity per animal and per area because of low forage availability where
60 livestock is obliged to walk long distances in search of forage and water. The natural soil fragility
61 combined with overgrazing has led to intense soil degradation by soil erosion (Roesch et al., 2009;
62 Reichert et al., 2016; Ebling et al., 2020; Ferreto et al., 2020; Valente et al., 2020), and loss of soil
63 carbon and grassland species diversity (Overbeck et al., 2007). Intense grazing and trampling by cattle
64 cause soil compaction (Collares et al., 2011; Cecagno et al., 2016; Ambus et al., 2018), through an
65 increase in the soil bulk density and a decrease in porosity.

66 Annual crops (soybean) and cultivated forests were increasingly found, from 1995 to 2005,
67 in Pampa's territory (Oliveira et al., 2017). In 2017, forestry activities were conducted on 780,900 ha
68 (i.e. 2.7% of the RS State territory), with 55% of these forests corresponding to eucalyptus, 34% to
69 pinus, and 11% to acacia plantations (AGEFLOR, 2017). This expansion is a response to the
70 increased demand for forest-based raw-materials, low prices of land in this region, and currently

71 encouraged by carbon offset projects. This change in land use and management of natural resources
72 consequently raised concerns on environmental sustainability associated with the introduction of
73 exotic, fast-growing forest species (Andriollo et al., 2017; Mateus and Padilha, 2017; Reichert et al.,
74 2017), requiring conservation methods to reduce soil erosion and sedimentation (Ebling et al., 2020;
75 Valente et al., 2020), runoff (Ferreto et al., 2020), soil compaction (Holthusen et al., 2018; França et
76 al., 2021), and water consumption (Ferreto et al., 2021). Similar concerns had been raised in Uruguay,
77 where the conversion of native grasslands into croplands and exotic forest plantations (*Eucalyptus*
78 and *Pinus*) has fast-tracked during the last two decades (Brazeiro et al., 2020).

79 Previous catchments studies showed a decrease in streamflow in response to the afforestation
80 of grasslands (Gush et al., 2002; Andréassian, 2004; Germer et al., 2009; Silveira et al., 2016).
81 Among the environmental impacts of afforestation of Pampa, it has been shown that afforestation
82 increases evapotranspiration and induces flow from the grassland areas into the forest plantation
83 (Engel et al., 2015), promote secondary salinization in the flooding Pampas under old-unmanaged
84 plantations (Milione et al., 2020), decrease live plant cover, but increase litter cover (Clavijo et al.,
85 2005), and reduce the taxonomic biodiversity of birds in adult forests, but with no reduction in
86 functional diversity (Jacoboski and Hartz, 2020). Moreover, afforestation promotes changes in the
87 local productive, economic, and social dynamics (Matte and Waquil, 2020).

88 Land use changes impact erosion processes and sediment loads vary significantly throughout
89 the year (Vercruysse et al., 2017). Temporal variability in suspended sediment concentration in
90 function of streamflow can be expressed as hysteresis patterns (Vercruysse et al., 2017), which
91 provide useful insights into the occurrence of feedback mechanisms and thresholds controlling
92 suspended sediment transport (Eder et al., 2010; Krueger et al., 2009; Marttila and Klove, 2010).
93 Moreover, different approaches are used to describe the dynamics of sediment mobilization and
94 transport at the basin scale; for example, the sediment source identifications (fingerprinting approach)
95 to identify the main sediment supplier to the stream network (e.g., Minella et al., 2009; Le Gall et al.,

96 2016; Tiecher et al., 2017; Rodrigues et al., 2018; Valente et al., 2020), and to model erosion in the
97 catchment scale (Bonumá et al., 2014; Rodrigues et al., 2014; Fu et al., 2019; Ebling et al., 2020).

98 Erosion is a size-selective process during the supply, transport and deposition stages of
99 sediment movement (Koiter et al., 2013; Zhang et al., 2020). The total sediment load in rivers consists
100 of suspended sediment and bed load, where the suspended sediment contribution is significantly
101 greater (up to 95%) than that of bed load (Morgan, 2005; Carvalho, 2008). However, in some
102 catchments, the bed load discharge can reach up to 30% of the total sediment export (Carvalho, 2008;
103 Cantalice et al., 2014; Lenzi et al., 2016).

104 Understanding the hydrosedimentological dynamics and differences in sediment yield in
105 response to land use changes is essential for improving our comprehension of transport mechanisms.
106 This knowledge is also necessary to identify vulnerable land, define best management practices, and
107 allocate limited financial resources to conservation projects in the most threatened areas (Minella et
108 al., 2007, 2009a,b; Valente et al., 2020). To the best of our knowledge, few studies used sediment
109 yield to analyze the effects of intensive forest plantation management in Pampas grasslands of
110 southern South America. A review by Modernel et al. (2016) showed land use change and
111 overgrazing led to increased soil erosion, but little quantitative information was found in the literature
112 on water provision and sedimentation control in these environments. Forest plantation management
113 influences the impacts of afforestation on hydrosedimentology; for instance, clear-cut harvest exposes
114 soil thereby encouraging erosion and sediment transport to streams (Rodrigues et al., 2019; Cassiano
115 et al., 2020), and thus staggered harvesting may be an alternative to regulate streamflow and reduce
116 sedimentation.

117 We hypothesized that converting degraded grassland to eucalyptus plantation reduces
118 surface runoff and sedimentation. The aim of this study was to assess the effects of afforesting
119 degraded grassland on streamflow and sedimentation in subtropical headwater catchments. The study
120 covers about four years of monitoring two catchments with eucalyptus plantation and degraded
121 grasslands, that included the harvest of 21% of the planted eucalyptus, which allows us to evaluate

122 the effect of the strategy of partial harvest on sediment load, in the Southern Grasslands biome. The
123 novelty is bridging the processes of streamflow and sedimentation to forest management in the
124 understudied region. Specific needs for implementing appropriate management and effective
125 conservation measures are then discussed.

126

127 **3. Materials and methods**

128 *3.1 Catchment characterization*

129 The study was conducted in two paired catchments, one covered with eucalyptus plantation
130 (*Eucalyptus saligna*) referred to as eucalyptus catchment (EC), and the other occupied with natural
131 and exotic grasses, used for livestock farming and referred to as grassland catchment (GC) (Fig. 1;
132 Supplementary material A). These catchments are located in the Rio Grande do Sul State, southern
133 Brazil, and flow into the Vacacaí and Vacacaí-Mirim river basins (SEMA, 2017), which are sub-
134 basins of the Jacuí river basin, which is the main water supply to the metropolitan region of the state
135 (Porto Alegre) with more than 4 million of inhabitants.

136 Climate is Cfa, humid subtropical with no drought, according to the Köppen climate
137 classification, with an average annual temperature of 18.6 °C and average annual precipitation of
138 1,356 mm (Alvares et al., 2013). The soils in both catchments are Ultisols, Inceptisols and Entisols
139 in Soil Taxonomy (USDA, 1999), with area coverage Ultisols >> Cambisols > Entisols (Peláez,
140 2014). These soils are physically-fragile, and of low natural fertility and agricultural potential
141 (Ramgrab et al., 2004).

142 The main watercourses in both catchments are second-order creeks (Strahler, 1957). Mean
143 elevation in EC is 272 m and mean slope is 7.7% (Fig. 2A), whereas in GC the mean elevation is 273
144 m, and the slope is 3.1% (Fig. 2B). Drainage area is 1.10 km², perimeter is 4.32 km, and time of
145 concentration is 107 minutes in GC, whereas in EC these values amount to 0.83 km², 4.17 km, and
146 172 minutes (Reichert et al., 2017).

147 Main land uses in the EC correspond to plantations of *Eucalyptus saligna* (61% in total, with
148 40% planted in 2006 and 21% in 2014 after harvest operations), grassland with brush weeds (22.1%),
149 riparian vegetation (7.9%), unpaved roads (5.8%), and rock outcrops (2.6%) (Fig. 2C). Before
150 afforestation, the catchment was used as pasture, with similar management as our studied GC.
151 Eucalyptus stands had 3.0 m × 3.3 m spacing, and the stand planted in 2006 had an average diameter
152 at breast height of 0.17 m and an average height of 25 m. The grassland is composed of grasses and
153 shrubs, in which *Aloysia gratissima* (Verbenaceae) and *Heterothalamus alienus* (Asteraceae) were
154 the most abundant species. The riparian vegetation is composed by an arboreal stratum of native
155 species, with individuals of 6–8 m in height, including *Sebastiania commersoniana*, *Rollinia*
156 *salicifolia*, *Styrax leprosus*, *Eugenia uniflora*, *Luehea divaricata*, *Casearia decandra*, *Diospyros*
157 *inconstans*, *Myrcianthes pungens* and *Ocotea* spp. (Peláez, 2014).

158 In the GC, the main land uses are degraded native grassland (61.7%), exotic perennial grass
159 (*Avena strigosa*) (31.1%), eucalyptus patches and isolated individuals (3.3%), riparian vegetation
160 (2.1%), reservoir (1.7%), and farm buildings (0.1%) (Fig. 2D). The degraded native grassland is
161 composed of *Saccharum angustifolium*, *Aristida laevis*, *Baccharis riograndensis*, *Andropogon*
162 *lateralis* and *Eryngium pandanifolium*, whereas the lower vegetation stratum consisted of *Paspalum*
163 spp., *Axonopus affinis* and *Fimbristylis autumnalis* (Peláez, 2014). The degraded native grassland
164 produces a low vegetation cover of the soil, because of intensive grazing and low vegetation growth
165 in poor soils. Overgrazing and trampling further exposes the soil to erosion processes. The exotic
166 perennial grass (black oat) is sown between April and May, after lime application and after plowing
167 the soil.

168

169 3.2 Hydrosedimentological monitoring and analysis

170 Monitoring was conducted from September 2013 to March 2017 in gauge stations, one in
171 each catchment, equipped with concrete weir, water pressure sensor to measure water level
172 (limnigraph), turbidimeter, pluviograph, and sediment samplers installed to collect both suspended

173 and bed load sediment (see Supplementary material B for details). The automatic equipment recorded
 174 data at 10-minute intervals. The rate curve between water level and water discharge (streamflow) was
 175 determined for the ranges of values measured up to a maximum vertical height of 1.0 and 0.8 m,
 176 respectively for GC and EC, which correspond to maximum water discharges of 8,131 and 2,711 L
 177 s⁻¹. Of the 150 monitored events, in 51 of them a direct comparison of streamflow data in both
 178 catchments (EC and GC) were compared.

179 Sampling of water-sediment was conducted manually during rainfall events with a USDH-
 180 48 sampler (Supplementary material B) to measure suspended sediment concentration (SSC).
 181 However, due to the need of continuous SSC records, automatic turbidity measurements were used
 182 to establish a continuum. Table 1 shows the sampling frequency of suspended sediment for distinct
 183 flow discharge ranges. During the study period, 15 rainfall-runoff events were monitored
 184 simultaneously in both catchments (Supplementary material C).

185 Suspended sediment concentration (mg L⁻¹) was estimated from turbidity (NTU) based on a
 186 two-step analysis: (i) calibration with standard solutions and (ii) calibration with SSC values obtained
 187 during the events (Minella et al., 2008; Merten et al., 2014). The first method used polymer calibration
 188 solutions at seven concentration levels: 0 (distilled water), 20, 50, 100, 400, 1000 and 3000 NTU.
 189 The second method established the relationship between NTU and sediment concentrations in
 190 samples collected during the events (Fig. 3). The suspended sediment concentration was determined
 191 by the evaporation method (Shreve and Downs, 2005). Suspended sediment concentration was
 192 determined using the sediment concentration and flow data from the monitoring sections (Eq. 1):

$$193 \quad SY = \sum_{i=1}^n k(SSC * Q_i) \quad (1)$$

194 Where: SY is the sediment yield (Mg); SSC is the sediment concentration (g L⁻¹); Q_i is the flow (L s⁻¹);
 195 ¹); k is a unit conversion factor; n is the number of instantaneous measurements of SSC and Q
 196 performed at a given time i.

197 Bed load was monitored using a BLH-84 sampler, following the method proposed by
 198 Edward and Glysson (1999). After drying, samples were passed through a sieve with a mesh opening

199 of 0.063 and 2 mm for the measurement of the corresponding weight of fine (<0.063 mm) and coarse
 200 sediment (0.063-2 mm), respectively. Grain size distribution of sediment samples was analyzed with
 201 a laser granulometer, after oxidation of organic matter with H₂O₂ and dispersion with NaOH (Muggler
 202 et al., 1997). The bottom sediment discharge was cumulated with the suspended sediment discharge
 203 during the events to obtain the total sediment yield for the monitoring period. The values of the bed
 204 sediment discharge were determined by using Eq. 2, established by Gray (2005):

$$205 \quad Q_{bl} = \sum_i^n \left(\frac{m}{(w*t)} \right) * b * 0.0864 \quad (2)$$

206 Where: Q_{bl} is the bed load discharge (Mg day⁻¹); m is the sediment mass (g); w is the nozzle section
 207 (m); t is the sampling time (s); b is the bed width (m); 0.0864 is a unit conversion factor for Mg day⁻¹.
 208

209 The Q and SSC patterns were assessed through the hysteresis analysis, using the methodology
 210 proposed by Lawler et al. (2006). The events were classified based on shape, direction, and index of
 211 hysteresis loop (Eq. 3-5). If the hysteresis curve has a clockwise direction, the hysteresis index (HI)
 212 is positive (Eq. 3), whereas if the hysteresis curve has a counter-clockwise direction, the hysteresis
 213 index (HI) is negative (Eq. 4).

$$214 \quad HI = \left(\frac{SSC_{RL}}{SSC_{FL}} \right) - 1 \quad (3)$$

$$215 \quad HI = \left(\frac{-1}{\left(\frac{SSC_{RL}}{SSC_{FL}} \right)} \right) + 1 \quad (4)$$

216 Values of suspended sediment concentration in the rising limb (SSC RL) and in the falling
 217 limb (SSC FL) can be obtained by interpolation between the points for which SSC and Q
 218 measurements are available. The values of SSC RL and SSC FL are obtained from the central value
 219 (Q_{cen}) of streamflow (hysteresis graph), where Q_{cen} is the central value between the maximum and
 220 minimum flow of the rising limb of the event.

221 To analyze the difference between the studied catchments, we tested the values of
 222 hydrosedimentology parameters for the null hypothesis that data from both catchments are samples
 223 from continuous distributions with equal medians. We used the rank sum function of Matlab to

224 perform the nonparametric Wilcoxon-Mann-Whitney at 5% probability. We did not test the normality
225 of the dataset since the Wilcoxon Rank Sum Test does not assume normal or any known distribution.
226

227 4. Results

228 4.1 Rainfall and runoff

229 Annual rainfall (1823 mm) was greatest in 2015-2016. In 2014-2016, the Rio Grande do Sul
230 State was strongly affected by the *El Niño* phenomenon, which contributed to a rainfall increase of
231 25% compared with the historical annual rainfall average. The mean annual rainfall for the study
232 period was 1736 mm, while for the last 30 years (1986-2016) this mean was 1530 mm for the
233 municipality of São Gabriel (HIDROWEB, 2020).

234 Total rainfall per event ranged between 4.6 and 154.5 mm, with a mean of 31.2 mm
235 (Supplementary material D) and the maximum rainfall intensity observed reached 54.1 mm h⁻¹. Five
236 rainfall events presented rainfall intensity greater than 40 mm h⁻¹ (events of 01/13/2014, 03/03/2014,
237 07/04/2014, 09/22/2015 and 12/18/2015). As the study period was affected by the *El Niño*
238 phenomenon, the analysis of high-volume events is important, since they have a significant impact
239 of crop growth and development (Goulart et al., 2021), and their frequency has been increasing during
240 the last years, Furthermore, rainfall in the study area is well distributed throughout the year, the types
241 of precipitation (frontal vs. convective rainfall) and temperature vary throughout the year. The
242 analysis of the results by season could indicate seasonal patterns of flow generation, particularly if
243 more years were available for analysis. Therefore, we will continue collecting data to be able, in the
244 future, to separate the analysis in dry and wet seasons.

245 For the 51 monitored events that allow a direct comparison of sediment yield in both
246 catchments (EC and GC), the mean rainfall was 21 mm and the mean rainfall intensity was 15.8 mm
247 h⁻¹. The minimum, maximum and mean values of discharge peak, runoff, and runoff coefficient were
248 greater in GC than in EC (Table 2; Fig. 4). The medians were statistically different between the

249 catchments for all these variables by the Wilcoxon-Mann-Whitney test at a significant level of 5%
250 (Supplementary material E). Stream discharge exceeded 500 L s^{-1} in thirty-seven events presented
251 during the monitoring period in GC, compared with only seven events in EC (Supplementary material
252 D). Events with rainfall lower than 30 mm had a mean runoff coefficient of 6.62% in GC compared
253 with 1.33% in EC.

254 Streamflow exceedance for each monitored year is depicted in Fig. 5. The 5% time
255 streamflow (Q_5) over the entire study period (2013–2017) varied from 65 to $90 \text{ L s}^{-1} \text{ km}^{-2}$ in GC,
256 and from 20 to $45 \text{ L s}^{-1} \text{ km}^{-2}$ in EC. The Q_5 during the wettest year (2015-2016) was approximately
257 double ($90 \text{ L s}^{-1} \text{ km}^{-2}$) in GC than in EC ($45 \text{ L s}^{-1} \text{ km}^{-2}$), and approximately 40% greater in EC and
258 26% in GC compared with the other drier years. The 50% time streamflow (Q_{50}) was approximately
259 60% greater in GC ($10 \text{ L s}^{-1} \text{ km}^{-2}$) than in EC ($4 \text{ L s}^{-1} \text{ km}^{-2}$) in 2013-2014, 33% in 2014-2015, 36%
260 in 2015-2016, and 25% in 2016-2017. The 75% time streamflow (Q_{75}) was similar between GC and
261 EC (values around 3 and $8 \text{ L s}^{-1} \text{ km}^{-2}$). Finally, the 95% time streamflow (Q_{95}) was greater in EC
262 than in GC (values around 1 and $4 \text{ L s}^{-1} \text{ km}^{-2}$). The results in Fig. 5A show greater water availability
263 in EC during low streamflow compared with GC.

264

265 4.2 Sedimentation

266 Streamflow and suspended sediment concentration dynamics during the events resulted in
267 hydrographs and sedimentographs that were rarely synchronized in time. The sedimentographs
268 demonstrate the occurrence of maximum suspended sediment concentrations before the streamflow
269 peak in GC, and after this peak in EC (Fig. 6). The streamflow peak in GC was 3.3 to 10.6 times
270 greater than in EC, depending on rainfall volume and intensity. Consequently, sediment yield was 12
271 times greater in GC compared with EC, as illustrated during events that took place on 03/29/2015 and
272 04/25/2016 (Fig. 6D, H), with very-steep rising and falling hydrograph limbs.

273 Some low rainfall events in GC (Fig. 6H, J) had small magnitude hydrographs, but with high
274 sedimentographs magnitude. For the events on 07/04/2014, 10/14/2015 and 04/25/2016 (Fig. 6B, G,

275 K), a streamflow increase did not provide a similar increase in suspended sediment concentration.
276 This behavior could be associated with sediment available for transport in GC, where less suspended
277 sediment is transported even with a high amount of runoff. In contrast, for the same events in EC, the
278 streamflow peak was accompanied by greater suspended sediment concentration, which possibly
279 indicates greater transport capacity and/or greater availability of sediment.

280 For the 51 monitored events that allow a direct comparison of sediment yield in both
281 catchments (EC and GC), the minimum, maximum and mean values of maximum concentration and
282 yield suspended sediment (SSC_{max}; SSY), and hysteresis loop index (HI) of the events were greater
283 in GC than in EC (Table 2). The medians were statistically different between the catchments for all
284 these variables (Wilcoxon-Mann-Whitney test at a significant level of 5%). Sediment yield was
285 greatest on 07/04/2014, after 77 mm (53.1 mm h⁻¹) of cumulative rainfall, which resulted in greater
286 streamflow peak (6,206.0 and 1,513.6 L s⁻¹, respectively, in GC and EC), suspended sediment
287 concentration (2,290.2 and 1,046.8 mg L⁻¹, respectively) and total sediment yield (26.8 and 11.8 Mg
288 km⁻², respectively) (Fig. 6; Supplementary material D). This event occurred a few weeks after
289 eucalyptus harvesting in 21% of the catchment area, followed by soil tillage and new planting.
290 Therefore, even under these potentially highly erosive conditions, the sediment yield in EC was lower
291 than in GC.

292 The staggered forest harvest led to a similar sediment yield which remained comparable to
293 that observed during the pre-harvest period. For this event, sediment yield corresponded to 65 and
294 83% of the total annual yield recorded in 2013-2014 in GC and EC catchments, respectively.
295 Considering the surface runoff coefficient, the GC lost much more water than EC, with respective
296 values of 45% and 11%. Moreover, even after the partial harvesting and subsequent soil tillage
297 between March and May of 2014, greater streamflow and sediment yield were recorded in GC
298 (Supplementary material F).

299 Sediment yield was similar between catchments on 04/25/2016 (Fig. 6), although the
300 maximum discharge for EC was five times lower than for GC (1,513.6 and 5,169.0 L s⁻¹, respectively).

301 Consequently, the suspended sediment concentration peak was lower in EC (692.6 mg L⁻¹) compared
302 with GC (829.6 mg L⁻¹). The granulometric characteristics (sand, silt and clay percentages) of a
303 selection of sediment samples collected in their respective study areas are shown in Table 3. The
304 granulometric characteristics were similar in both catchments, covered with similar soil types.

305 Bed load was only recorded during high magnitude events, and more frequently observed in
306 GC (Fig. 4E) than in EC, which may reflect the greater sensitivity of GC to this type of sediment
307 transport. For the five events with measurements, bed load mean, maximum and minimum were,
308 respectively, 0.019, 0.053, and 0.002 Mg km⁻² in GC and 0.002, 0.006, and 0.000 Mg km⁻² in EC,
309 without statistical differences between the catchments (Wilcoxon-Mann-Whitney test at a significant
310 level of 5%). Thus, bed load was one order magnitude greater in GC compared with EC, but bed load
311 represented only a small fraction of the total solid discharge, with less than 1% of the total sediment
312 yield. By contrast, the maximum suspended sediment concentration was three times greater in GC
313 than in EC.

314 Annual sediment yield (Fig. 7) was greatest for the year of 2015-2016 in both catchments
315 with 39.9 Mg km⁻² for EC and 167.7 Mg km⁻² for GC. During this year, there were many successive
316 rainfall events with a high volume of precipitation (Supplementary material C). Nonetheless, when
317 the maximum values of streamflow and suspended sediment concentration were compared among
318 years, the greatest values were found in 2013-2014, due to a particularly intense event that occurred
319 on 04/07/2014.

320 Linear regressions between rainfall, runoff and erosive variables considering all events to
321 demonstrate the distinct behavior between the catchments are presented in Fig. 8. For all the
322 relationships, the dependent variables (SSC, sediment yield, and bed load) were more sensitive to the
323 independent variables (streamflow) in GC than in EC.

324

325 4.3 *Hysteresis analysis*

326 Table 4 summarizes the hysteresis analyses, and Fig. 9 shows the hysteresis loops of the
327 monitored events. In GC, most of the hysteresis patterns were clockwise (CW), i.e. the
328 sedimentograph is advanced in relation to the hydrograph, which indicates fast delivery of sediment
329 sources at hillslopes with high connectivity (possibly because of animal trails) or from erosion in
330 fluvial channel. Bank collapses were visually observed in both catchments, although they were more
331 frequently found in GC.

332 Hysteresis had counter-clockwise (ACw) shape in EC, which could mean the eroded
333 sediment is mainly supplied by remote sources, although sediment deposits were observed in the
334 stream channel (Supplementary material A). Hysteresis loop index (HI) ranged from -2.5 to 9.2 and
335 -1.4 to 1.2 for GC and EC, respectively (more information in Supplementary Material D), and Table
336 2 shows the hysteresis loop index was different for GC (0.67) than for EC (-0.32) for the 51 events
337 with complete data.

338

339 5. Discussion

340 5.1 *Runoff*

341 The lower streamflow and runoff coefficient observed in EC compared with GC can be
342 attributed to the greater capacity of rainwater interception by the eucalyptus canopy, which was
343 estimated to approximately 13% of rainfall (Peláez, 2014). Furthermore, our results confirm those of
344 Reichert et al. (2017) showing that the high values of streamflow reflect lower infiltration and greater
345 surface runoff in GC. Baumhardt (2014) also observed greater peak flows in a catchment covered
346 with grassland than in eucalyptus stands in the Pampa biome, whereas Almeida et al. (2014) observed
347 greater water infiltration under eucalyptus plantations compared with grazing areas. Even during the
348 periods with the greatest amount of rainfall (in September 2015 and in August 2016) when a large
349 number of consecutive rainfall events occurred, less runoff and lower sediment yields were recorded
350 in EC.

351 Forest canopy acts as a barrier against rainfall energy and reduces the volume that reaches the
352 ground given the partial interception of rainfall by tree canopy, trunk and litter. These processes
353 reduce the amount of precipitation and redistribute it to the ground (Chang, 2012). Although the soil
354 surface is more protected under forest plantations than under cropland or grassland, soil degradation
355 may also occur during tillage operations implemented during tree plantation, harvest and road
356 construction and maintenance (Sheridan et al., 2006; Ferreira et al., 2008; Oliveira et al., 2014). In
357 areas with extensive livestock, soil compaction and animal trampling may limit infiltration and
358 accelerate runoff and erosion (Holt et al., 1996; Müller et al., 2001). In the grassland catchment, we
359 observed many cattle trails were observed in the field and towards the creek. Although not quantified,
360 these trails might increase hydraulic connectivity from the slope to the creek, potentially increasing
361 runoff and sedimentation.

362

363 5.2 *Sedimentation*

364 Forest harvesting of 21% of the catchment area and subsequent replanting did not lead to
365 major changes in sediment yield in EC. This observation demonstrates that the staggered forest
366 harvest may provide a sustainable option to manage tree plantations. Almeida et al. (2016) suggested
367 that the partial harvest of eucalyptus plantations reduces the impact on streamflow compared with
368 clear cutting the entire catchment area. Furthermore, unpaved roads in forests may provide a major
369 sediment source in forested catchments (Croke et al., 1999; Croke and Mockler, 2001; Hairsine et al.,
370 2002). Unpaved roads were shown to contribute more than 90% of the sediment yield (Grace et al.
371 1998; Madej 2001). High values of runoff (coefficients of 44%) and soil loss (22 Mg ha⁻¹) were
372 observed on forest unpaved roads by Oliveira et al. (2014).

373 In forestry sites, the effects of weathering, groundwater seepage, geotechnical instability, and
374 erosion conditions on local bank collapse are not well understood, and this topic requires further
375 research (Martilla and Klove, 2010). Besides being a significant source of sediment, bank erosion can

376 cause structural damage because particles eroded from the bank cannot be replaced. In larger stream
377 systems bank sediment can account for over 50% of the total sediment input (Knighton, 1998).

378

379 5.3 *Hysteresis*

380 Counter-clockwise hysteresis as the most frequent pattern in EC suggests lack of sediment
381 available for resuspension in the channel (Martilla and Klove, 2010), and thus sediments should be
382 mobilized from remote sources (Minella et al., 2011; Seeger et al., 2004). Furthermore, lower bed
383 load transport was observed in EC compared with GC, although lag widespread deposits were
384 observed in the stream channel of GC. For this same catchment, using fingerprinting techniques
385 Valente et al. (2020) found 63% of the sediment derived from stream channel and only 30 % from
386 eucalyptus stands. Rodrigues et al. (2018) also identified stream channel as the main sediment source
387 in forested catchments with stream bank collapse.

388 Clockwise hysteresis, most frequent in GC, is generally attributed to rapid delivery of
389 sediment from channel banks or sources located in the outlet vicinity (Smith and Dragovich, 2009),
390 whereas sediment deposited into the channel is easily available to transport during events by
391 resuspension (Martilla and Klove, 2010). Bank collapse was visually observed in both catchments,
392 although more frequently in GC, in line with the hysteresis findings. Channel bank erosion in GC was
393 accelerated by the cattle access and passage in these areas, which led to the formation of trails.
394 Furthermore, trampling and vegetation grazing led to the removal of the protective vegetation cover
395 in these areas. Using fingerprinting techniques, Valente et al. (2020) found only 14% of the sediment
396 derived from stream channel in the forested catchment, compacted with 84 % in the degraded
397 grasslands.

398

399 5.4 *Land use and management implications for the Pampa biome*

400 The study region is located within the Pampa biome or Southern Grasslands. This biome
401 corresponds to 750,000 km² in South America and 176,496 km² in Brazil, concentrated in the

402 southernmost state of Rio Grande do Sul (RS), corresponding to 63% of the state territory and 90%
403 of the state rangelands (Roesch et al., 2009; Brasil, 2016). The biome is rich in biodiversity and
404 traditionally used for cattle raising (Boldrini et al., 2010), but this scenario has been changing with
405 the introduction of annual crops and forest plantations.

406 Forest plantations impact the environment (Andriollo et al., 2017; Reichert et al., 2017;
407 Cavalli et al., 2020); for instance, eucalyptus have high water consumption than grasslands and
408 consequently decrease the soil moisture, streamflow and groundwater, with more significant effect in
409 regions with low rainfall (Jackson et al., 2005). The effect of forest plantations on water resources is
410 usually significant in small watersheds, but not in large ones (Van Dijk, 2007). Changes in hydrology
411 affect soil production potential, water quality and downstream water supply, and cause conflicts
412 among water uses (Calder, 2007; Vanclay, 2009). Forest harvesting might increase streamflow in dry
413 seasons, but may increase the risk of flooding during rainy seasons (Hamilton, 2008). Therefore,
414 long-term studies are needed to evaluate the effect of multiple rotations on the sustainability of soil
415 productive potential, including soil fertility and biodiversity, and water use by commercial forests,
416 such as eucalyptus monocropping in the Pampa biome.

417 By shifting the comparison between commercial forests and native grasslands towards an
418 enlarged focus on green-blue pools and flows such as soil water and stream discharge, rather than
419 concentrating on stream discharge, water management will be rainfall-based, including land use as
420 an important variable and seeking catchment environmental resilience and ecosystem roles
421 (Falkenmark and Rockström, 2010). Soil and forest/grassland management should be oriented
422 towards protecting or restoring soil moisture and manage the green water flux for increased water
423 productivity. An improved soil structure allows for increased infiltration and retention in the soil
424 (Cavalli et al., 2020; Ferreto et al., 2020).

425 In the areas with forest plantations, the post-harvest period and the earlier stages of
426 eucalyptus plantation are the ones with the greatest attention in terms of soil loss, since the soil is
427 uncovered (Oliveira et al., 2013). Limiting the size of clear-cut harvest areas and interrupted the

428 linkage between harvest areas and drainage network are classic approaches to mitigate erosion and
429 sedimentation processes in streams post-harvesting (Gimenez et al., 2019). The retention of harvest
430 residue also reduced soil loss and water runoff the first years of plantation establishment (Nambiar et
431 al., 2000). These approaches were observed in the studied area, since only 21% of the planted area
432 was harvested and the riparian vegetation is preserved, as required by Brazilian law. Therefore, the
433 adopted forest management contribute to making sediment yield comparable to the observed during
434 the pre-harvest period.

435 Although the current local management systems for eucalyptus and grasslands, in our study,
436 favor the former in terms of regulating streamflow and decreasing sedimentation, there are many
437 opportunities to improve degraded grassland watersheds. The management of the Pampa's natural
438 grasslands to pastures usually included overgrazing, low productivity, low technology and low
439 economic results (Nabinger et al., 2009). Practices to increase the resilience and functions of degraded
440 grasslands catchments include improving forage availability and quality for grazing, improve soil
441 fertility, adjusting the number of animals per hectare, controlling weeds, and grazing deferral
442 (Nabinger et al., 2009), along with reducing compaction from cattle trampling (Collares et al., 2011),
443 augmenting water infiltration by runoff reduction and soil structure amelioration, and increasing and
444 protecting riparian vegetation, among others.

445

446 **6. Conclusions**

447 Streamflow, suspended sediment concentration, and sediment yield were greater in degraded
448 grassland (GC) than in eucalyptus catchment (EC), as hypothesized, particularly during *El Niño* year.
449 Most sediment was lost in suspension, whereas bed load was recorded only during high magnitude
450 events, represented only a small fraction (less than 1%) of the total sediment yield.

451 Staggered forest-harvesting (21% of the EC) led to sediment yield comparable to the observed
452 during the pre-harvest period. Even after partial harvesting and subsequent soil tillage, greater
453 streamflow and sediment yield were lower than the recorded in GC. Thus, well-managed forest

454 plantations may be less prone to sedimentation than degraded grassland under intensive grazing, and
455 contribute for increased land use sustainability in terms of soil erosion and degradation.

456 Hydrographs/sedimentographs synchronization and hysteresis patterns suggest faster
457 sediment delivery in GC than in EC, signifying sediment sources in the former are hillslopes with
458 high connectivity and/or erosion in fluvial channels, whereas in the latter the eroded sediment is
459 mainly supplied by remote sources. However, further studies are needed to conciliate sediment
460 sources results with sediment fingerprinting.

461 Well-managed forest plantations, including staggered forest harvesting and preservation of
462 riparian forest, are less prone to sedimentation than degraded grassland under intensive grazing.
463 Moreover, the low ecological performance of the degraded grassland catchment calls for soil and
464 pasture management practices to increase forage offer to grazing animals, while conserving soil and
465 water resources. Along with long-term studies to ameliorate degraded grasslands and on multiple
466 forest rotations, further evaluations are recommended to estimate the contribution of sediment
467 deposition in the channel from runoff, cattle trampling in grassland catchment and collapse of channel
468 banks in eucalyptus catchment, and of hydraulic connectivity for fast-delivery of sediment from
469 the landscape to the stream channel.

470

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476

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773 **Table 1**

774 Sampling frequency of the mixture water and sediments, based on discharge, in the grassland (GC)
 775 and eucalyptus catchments (EC), during the study period (Sep/2013–Mar/2017).

Discharge interval (L s ⁻¹)	Frequency of sampling (%)			
	Fine sediment		Coarse sediment	
	GC	EC	GC	EC
0-100	39	91	4	54
101-200	25	15	4	21
201-300	4	18	13	8
301-400	3	5	4	13
401-500	9	4	13	0
501-600	2	0	4	4
601-700	2	0	4	0
701-800	2	0	0	0
801-900	0	0	9	0
901-1000	3	0	17	0
1001-2501	6	0	13	0
2501-4501	3	-	13	0
4501-6500	1	-	0	0
6501-8500	1	-	0	0

776

Table 2

Minimum, maximum and mean value of studied runoff and sedimentation variables in grassland (GC) and eucalyptus watersheds (EC) for the 51 events with complete data.

Variable	Unit	Minimum		Maximum		Mean	
		GC	EC	GC	EC	GC	EC
Maximum streamflow*	L s ⁻¹	105.0	9.0	6206.0	1282.0	2908.0	209.1
Streamflow*	mm	1.0	1.0	15.0	9.0	5.5	3.0
Runoff coefficient*	%	6.0	1.0	22.0	3.0	13.5	1.4
Maximum suspended sediment concentration*	mg L ⁻¹	17.3	10.1	2290.2	1046.8	253.0	141.8
Bed load ⁽¹⁾	kg km ⁻²	2.0	0.0	53.0	30.0	0.02	0.00
Sediment yield*	kg km ⁻²	0.0	20.0	26700.0	11830.0	1.9	0.6
Hysteresis loop index*		-1.0	-0.8	2.7	1.3	0.67	-0.32

* Medians for EC and GC are different when compared with the Wilcoxon-Mann-Whitney test at 5% probability.

⁽¹⁾ Only for five events. No significant difference.

Table 3

Percentage of sand, silt and clay in the different types of sediment samples (including suspended matter and lag deposits) collected in the eucalyptus (EC) and grassland (GC) catchments, during the study period (Sep/2013–Mar/2017).

Sediment (yy.mm.dd)	Eucalyptus catchment			Grassland catchment		
	Sand	Silt	Clay	Sand	Silt	Clay
	%			%		
Event - 14/07/04	8.2	73.6	18.2	7.5	76.0	16.5
Event - 14/10/30	0	91.5	8.5	5.6	81.4	13.0
Event - 14/12/21	0.8	81.6	17.6	5.9	81.5	12.7
Event - 15/10/08	7.9	75.7	16.4	6.3	81.2	12.5
Event - 16/10/07	5.8	74.5	19.7	5.6	77.3	17.1
Event - 16/10/19	12.7	71.7	15.7	7.0	80.5	12.5
Lag deposit - 14/07/05	24.6	70.0	5.4	17.7	75.6	6.7
Lag deposit - 14/08/20	17.2	75.1	7.6	13.5	78.9	7.6
Lag deposit - 14/09/20	26.0	69.2	4.9	-	-	-
Lag deposit - 14/12/20	23.2	71.5	5.3	-	-	-
Lag deposit - 15/03/12	22.4	72.1	5.5	-	-	-
Lag deposit - 15/06/18	22.0	72.1	5.9	18.4	73.9	7.7
Lag deposit - 15/10/15	23.5	71.1	5.3	-	-	-
Lag deposit - 15/12/03	25.6	69.3	5.1	18.7	73.1	8.1
Lag deposit - 16/02/03	15.3	76.5	8.2	10.2	78.8	11.0
Lag deposit - 16/06/23	25.3	69.5	5.2	-	-	-
Lag deposit - 16/09/03	9.8	82.0	8.2	-	-	-
Lag deposit - 16/10/16	22.7	71.6	5.7	-	-	-
Lag deposit - 16/11/15	24.3	70.3	5.4	-	-	-
Trap - 14/02/12	7.7	76.2	16.2	-	-	-
Trap - 15/07/17	2.6	89.6	7.8	6.5	83.3	10.2
Trap - 16/03/31	5.1	83.8	11.1	6.9	79.8	13.3
Trap - 16/10/12	7.6	81.6	10.8	8.2	77.1	14.8
Mean Event	5.9	78.1	16.0	6.3	79.7	14.0
Mean Lag deposit	21.7	72.3	6.0	15.7	76.1	8.2
Mean Trap	5.7	82.8	11.5	7.2	80.1	12.7

Table 4

Summary of SSC-Q hysteresis patterns monitored at the outlet of both grassland and eucalyptus catchments, during the study period (Sep/2013–Mar/2017).

Hysteresis pattern	Number of flow peaks	Mean peak discharge ($L^{-1} s^{-1} km^{-2}$)	Mean peak suspended sediment ($mg L^{-1}$)	Mean event rainfall (mm)	Mean rainfall intensity ($mm h^{-1}$)
Grassland catchment					
Counter-clockwise	23	265.9	153.6	23.3	13.0
Clockwise	61	558.4	246.2	31.7	15.4
Eucalyptus catchment					
Counter-clockwise	67	175.7	169.2	38.7	18.3
Clockwise	7	185.6	185.9	44.6	23.1

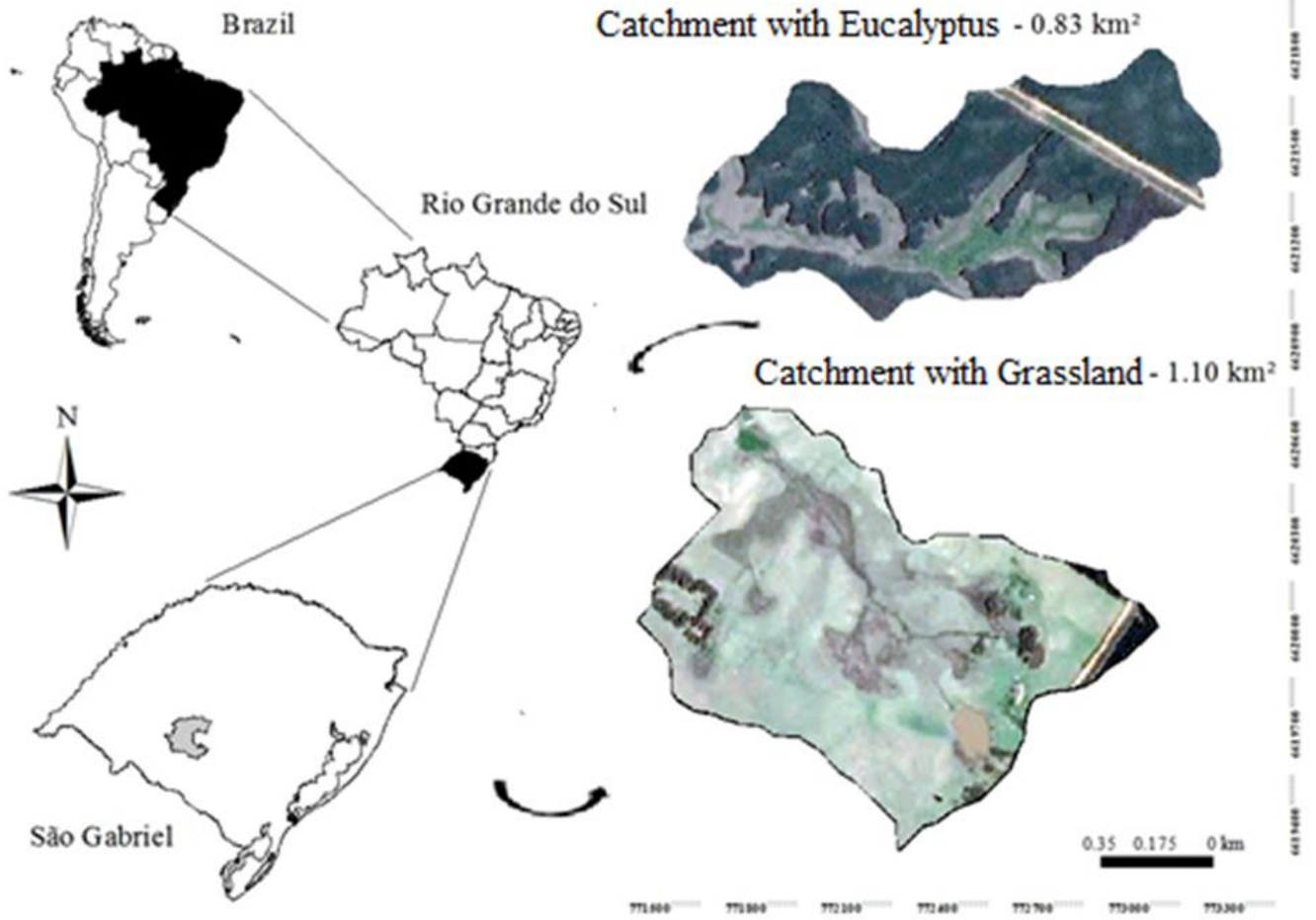


Fig. 1. Location of the municipality of São Gabriel-RS, Brazil, and delineation of the study catchments covered with eucalyptus and grassland.

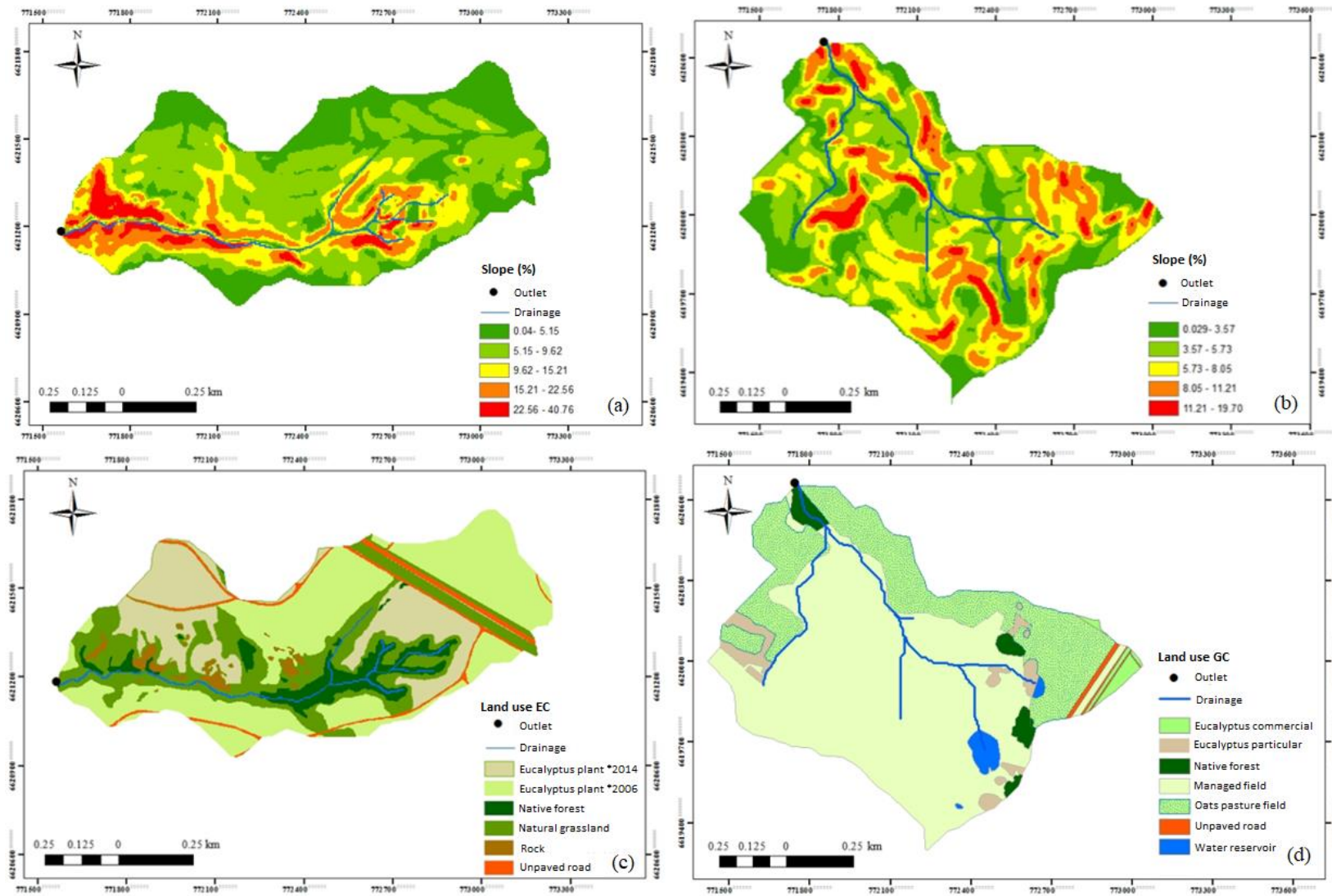


Fig. 2. Maps of slope for eucalyptus (a) and grassland catchment (b), and land use for eucalyptus (c) and grassland catchment (d).

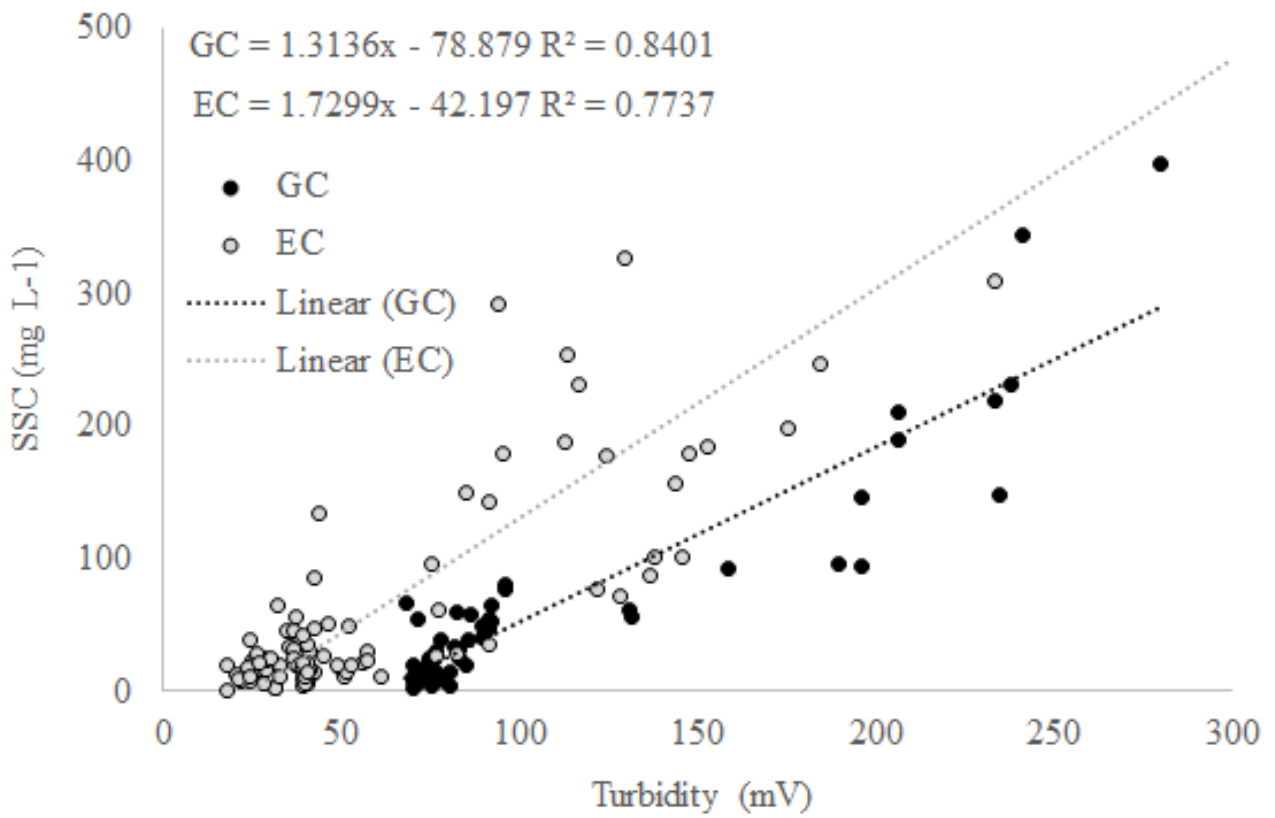


Fig. 3. Relationship between suspended sediment concentrations (SSC) and turbidity in the study catchments covered with eucalyptus and grassland.

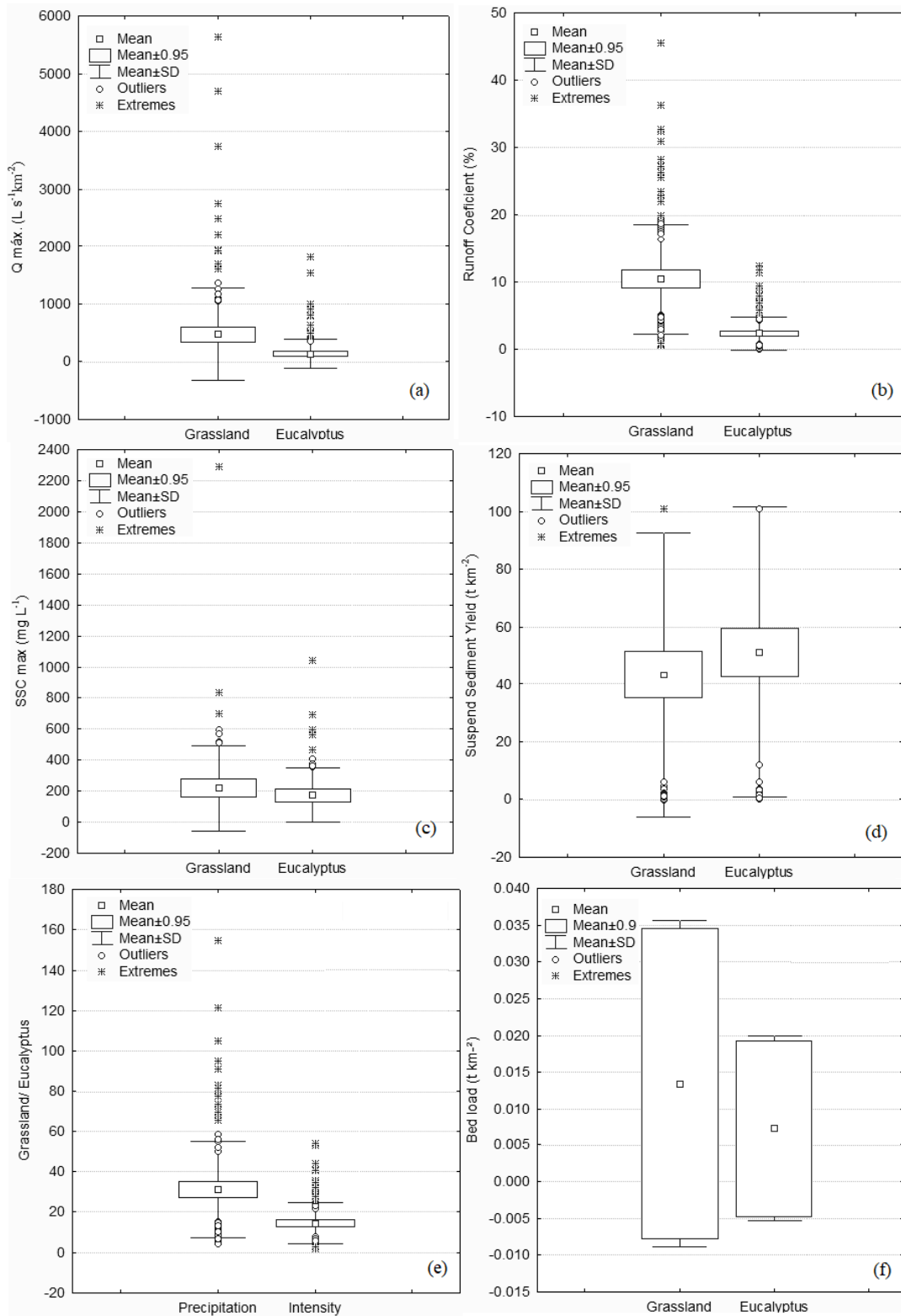


Fig. 4. Box-plot of hydro-sedimentologic variables: maximum streamflow (a), runoff or streamflow (b), maximum suspended sediment concentration (c), suspended sediment yield (d), and bed load (e) of the events monitored in the grassland and eucalyptus catchments. As shown in Table 2, medians of the different runoff and hydrology parameters were different, except for bed load, when comparing GC and EC with the Wilcoxon-Mann-Whitney test at 5% probability.

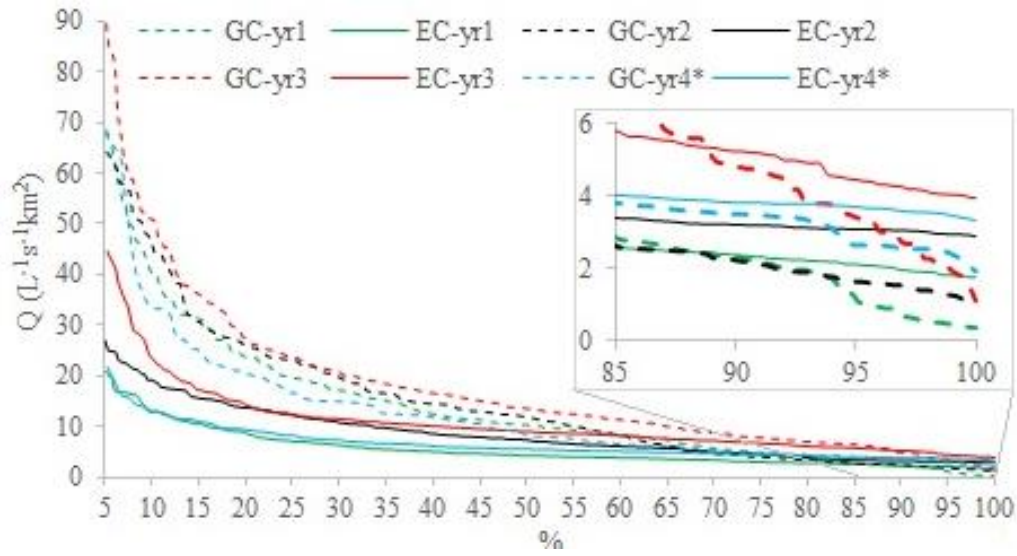


Fig. 5. Exceedance probability curves for streamflow in the grassland (GC) and eucalyptus catchments (EC), during the study period (Sep/2013–Mar/2017).

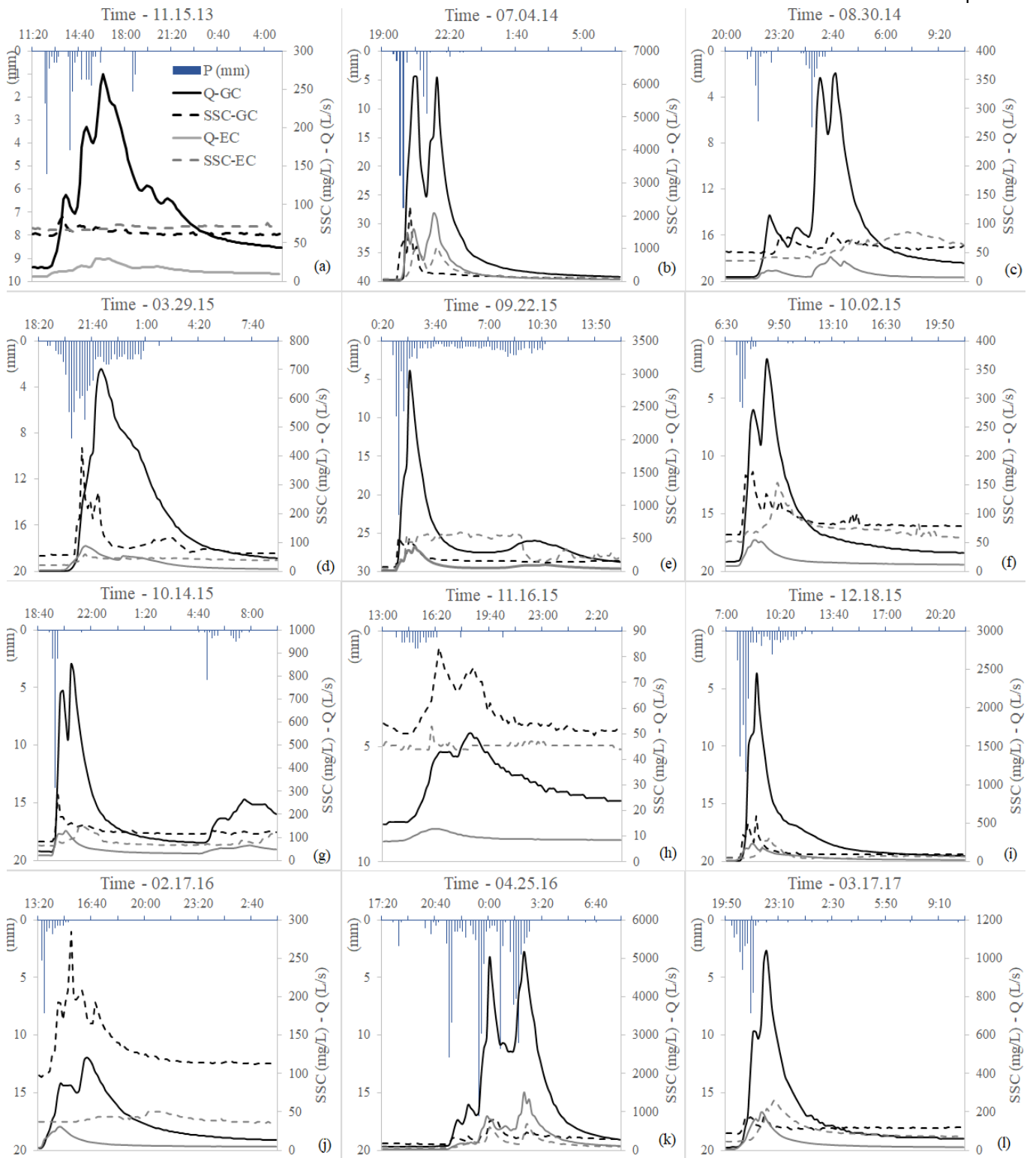


Fig. 6. Time series for events with different water flow (Q), suspended sediment concentration (SSC) and rainfall, selected to represent the typical peak events in the grassland (GC) eucalyptus catchments (EC).

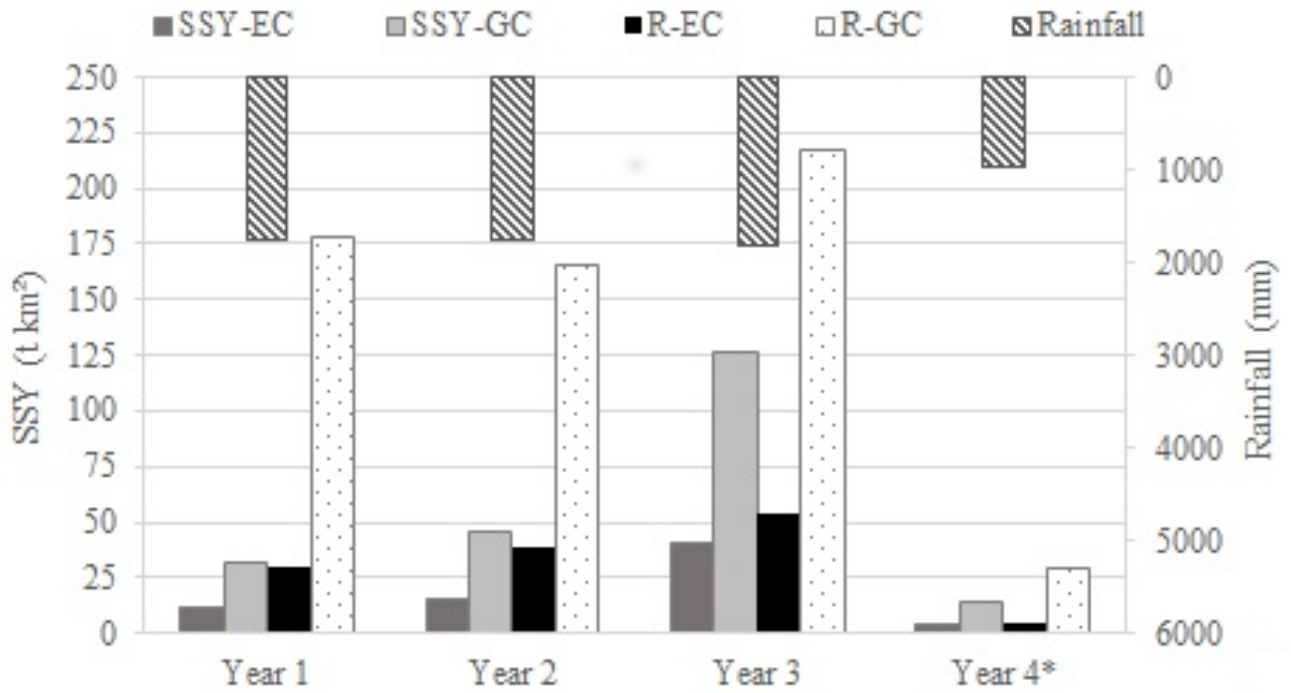


Fig. 7. Annual suspended sediment yield (SSY), streamflow or runoff (R) and rainfall were selected to represent the typical peak events in the eucalyptus and grassland catchments, during the study period (Sep/2013–Mar/2017). *Year 4 corresponds to a half-year, i.e. from Sep/2016 to Mar/2017.

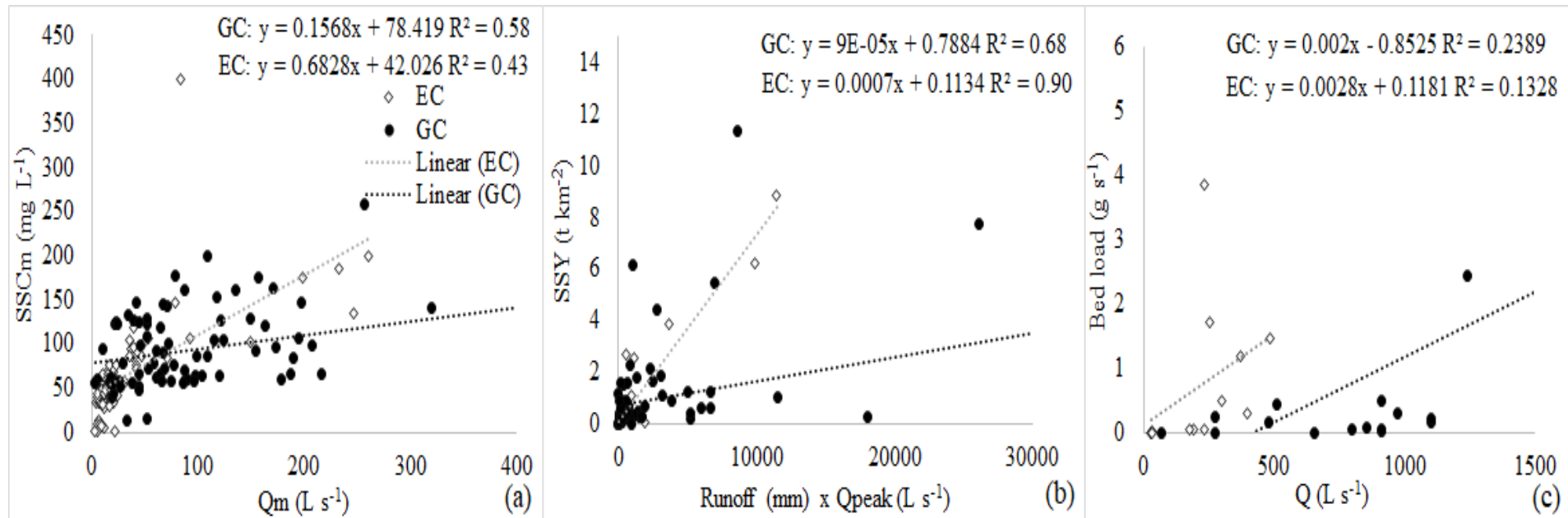


Fig. 8. Relationship between mean of suspended sediment concentration (SSC_m) and mean discharge (Q_m) (a), suspended sediment yield (SSY) with product of runoff and peak discharge (b), and bed load and discharge (c) in grassland (GC) eucalyptus catchments (EC).

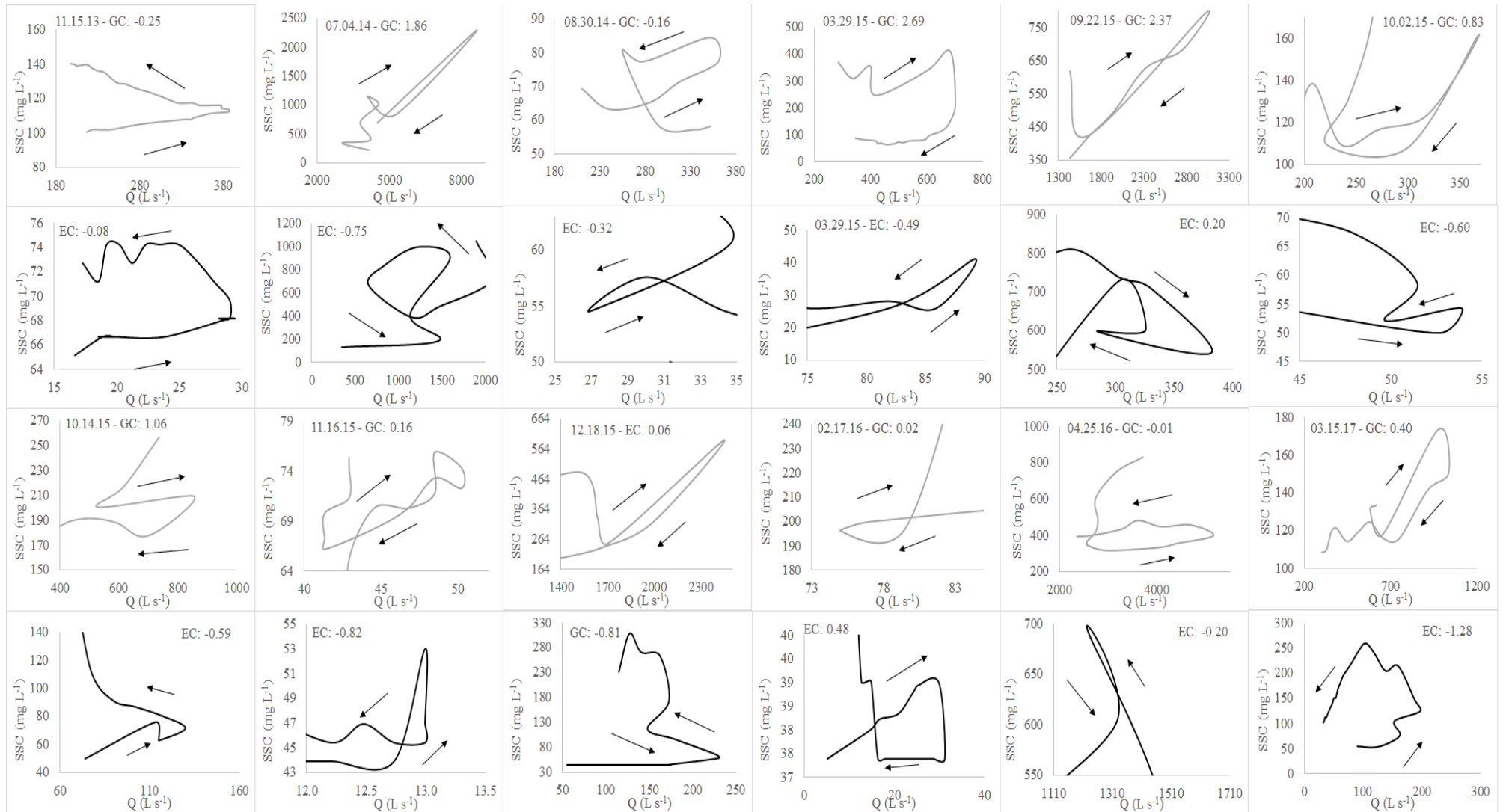


Fig. 9. Hysteresis loops of events in the grassland and eucalyptus catchments.