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

Mirian Lago Valente, José Miguel Reichert, Rosane Barbosa Lopes Cavalcante, Jean Paolo Gomes Minella, Olivier Evrard, Raghavan Srinivasan

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Mirian Lago Valente; Universidade Federal de Santa Maria, Santa Maria, RS, Brazil.
José Miguel Reichert*; Universidade Federal de Santa Maria, Santa Maria, RS, Brazil.
Rosane Barbosa Lopes Cavalcante; Instituto Tecnológico Vale - Desenvolvimento Sustentável, Belém, PA, Brazil.
Jean Paolo Gomes Minella; Universidade Federal de Santa Maria, Santa Maria, RS, Brazil
Olivier Evrard; Laboratoire des Sciences du Climat et de l'Environnement (LSCE/IPSL), UMR 8212 (CEA/CNRS/UVSQ), Université Paris-Saclay, Gif-sur-Yvette, France.
Raghavan Srinivasan; Department of Ecology and Conservation Biology, Texas A&M University, College Station, TX, USA.

*Corresponding author:
José Miguel Reichert
reichert@ufsm.br Tel: +55 55 32208918 Fax. +55 55 32208295
Soils Department, Federal University of Santa Maria, Avenida Roraima, 1000, Bairro Camobi, Santa Maria-RS 97105-900, Brazil.

1. Abstract

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

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

2. Introduction

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

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

Annual crops (soybean) and cultivated forests were increasingly found, from 1995 to 2005, in Pampa's territory (Oliveira et al., 2017). In 2017, forestry activities were conducted on 780,900 ha (i.e. 2.7% of the RS State territory), with 55% of these forests corresponding to eucalyptus, 34% to pinus, and 11% to acacia plantations (AGEFLOR, 2017). This expansion is a response to the increased demand for forest-based raw-materials, low prices of land in this region, and currently encouraged by carbon offset projects. This change in land use and management of natural resources consequently raised concerns on environmental sustainability associated with the introduction of exotic, fast-growing forest species (Andriollo et al., 2017; Mateus and Padilha, 2017; Reichert et al., 2017), requiring conservation methods to reduce soil erosion and sedimentation (Ebling et al., 2020; Valente et al., 2020), runoff (Ferreto et al., 2020), soil compaction (Holthusen et al., 2018; França et al., 2021), and water consumption (Ferreto et al., 2021). Similar concerns had been raised in Uruguay, where the conversion of native grasslands into croplands and exotic forest plantations (*Eucalyptus* and *Pinus*) has fast-tracked during the last two decades (Brazeiro et al., 2020).

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

Land use changes impact erosion processes and sediment loads vary significantly throughout the year (Vercruysse et al., 2017). Temporal variability in suspended sediment concentration in function of streamflow can be expressed as hysteresis patterns (Vercruysse et al., 2017), which provide useful insights into the occurrence of feedback mechanisms and thresholds controlling suspended sediment transport (Eder et al., 2010; Krueger et al., 2009; Marttila and Klove, 2010). Moreover, different approaches are used to describe the dynamics of sediment mobilization and transport at the basin scale; for example, the sediment source identifications (fingerprinting approach) to identify the main sediment supplier to the stream network (e.g., Minella et al., 2009; Le Gall et al., 2016; Tiecher et al., 2017; Rodrigues et al., 2018; Valente et al., 2020), and to model erosion in the catchment scale (Bonumá et al., 2014; Rodrigues et al., 2014; Fu et al., 2019; Ebling et al., 2020).

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

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

We hypothesized that converting degraded grassland to eucalyptus plantation reduces surface runoff and sedimentation. The aim of this study was to assess the effects of afforesting degraded grassland on streamflow and sedimentation in subtropical headwater catchments. The study covers about four years of monitoring two catchments with eucalyptus plantation and degraded grasslands, that included the harvest of 21% of the planted eucalyptus, which allows us to evaluate the effect of the strategy of partial harvest on sediment load, in the Southern Grasslands biome. The novelty is bridging the processes of streamflow and sedimentation to forest management in the understudied region. Specific needs for implementing appropriate management and effective conservation measures are then discussed.

3. Materials and methods

3.1 Catchment characterization

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

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

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

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

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

3.2 Hydrosedimentological monitoring and analysis

Monitoring was conducted from September 2013 to March 2017 in gauge stations, one in each catchment, equipped with concrete weir, water pressure sensor to measure water level (limnigraph), turbidimeter, pluviograph, and sediment samplers installed to collected both suspended and bed load sediment (see Supplementary material B for details). The automatic equipment recorded data at 10-minute intervals. The rate curve between water level and water discharge (streamflow) was determined for the ranges of values measured up to a maximum vertical height of 1.0 and 0.8 m, respectively for GC and EC, which correspond to maximum water discharges of 8,131 and 2,711 L s^{-1}. Of the 150 monitored events, in 51 of them a direct comparison of streamflow data in both catchments (EC and GC) were compared.

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

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

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

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

Bed load was monitored using a BLH-84 sampler, following the method proposed by Edward and Glysson (1999). After drying, samples were passed through a sieve with a mesh opening of 0.063 and 2 mm for the measurement of the corresponding weight of fine (<0.063 mm) and coarse sediment (0.063-2 mm), respectively. Grain size distribution of sediment samples was analyzed with a laser granulometer, after oxidation of organic matter with H_2O_2 and dispersion with NaOH (Muggler et al., 1997). The bottom sediment discharge was cumulated with the suspended sediment discharge during the events to obtain the total sediment yield for the monitoring period. The values of the bed sediment discharge were determined by using Eq. 2, established by Gray (2005):

$$Qbl = \sum_{i}^{n} \left(\frac{m}{(w*t)}\right) * b * 0.0864 \qquad (2)$$

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

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

$$HI = \left(\frac{SSC\ RL}{SSC\ FL}\right) - 1 \qquad (3)$$

$$HI = \left(\frac{-1}{\left(\frac{SSC\ RL}{SSC\ FL}\right)}\right) + 1 \qquad (4)$$

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

To analyze the difference between the studied catchments, we tested the values of hydrosedimentology parameters for the null hypothesis that data from both catchments are samples from continuous distributions with equal medians. We used the rank sum function of Matlab to perform the nonparametric Wilcoxon-Mann-Whitney at 5% probability. We did not test the normality of the dataset since the Wilcoxon Rank Sum Test does not assume normal or any known distribution.

4. Results

4.1 Rainfall and runoff

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

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

For the 51 monitored events that allow a direct comparison of sediment yield in both catchments (EC and GC), the mean rainfall was 21 mm and the mean rainfall intensity was 15.8 mm h^{-1}. The minimum, maximum and mean values of discharge peak, runoff, and runoff coefficient were greater in GC than in EC (Table 2; Fig. 4). The medians were statistically different between the catchments for all these variables by the Wilcoxon-Mann-Whitney test at a significant level of 5% (Supplementary material E). Stream discharge exceeded 500 L s^{-1} in thirty-seven events presented during the monitoring period in GC, compared with only seven events in EC (Supplementary material D). Events with rainfall lower than 30 mm had a mean runoff coefficient of 6.62% in GC compared with 1.33% in EC.

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

4.2 Sedimentation

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

Some low rainfall events in GC (Fig. 6H, J) had small magnitude hydrographs, but with high sedimentographs magnitude. For the events on 07/04/2014, 10/14/2015 and 04/25/2016 (Fig. 6B, G, K), a streamflow increase did not provide a similar increase in suspended sediment concentration. This behavior could be associated with sediment available for transport in GC, where less suspended sediment is transported even with a high amount of runoff. In contrast, for the same events in EC, the streamflow peak was accompanied by greater suspended sediment concentration, which possibly indicates greater transport capacity and/or greater availability of sediment.

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

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

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

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

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

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

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

4.3 Hysteresis analysis

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

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

5. Discussion

5.1 Runoff

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

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

5.2 Sedimentation

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

In forestry sites, the effects of weathering, groundwater seepage, geotechnical instability, and erosion conditions on local bank collapse are not well understood, and this topic requires further research (Martilla and Klove, 2010). Besides being a significant source of sediment, bank erosion can cause structural damage because particles eroded from the bank cannot be replaced. In larger stream systems bank sediment can account for over 50% of the total sediment input (Knighton, 1998).

5.3 Hysteresis

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

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

5.4 Land use and management implications for the Pampa biome

The study region is located within the Pampa biome or Southern Grasslands. This biome corresponds to 750,000 km^2 in South America and 176,496 km^2 in Brazil, concentrated in the southernmost state of Rio Grande do Sul (RS), corresponding to 63% of the state territory and 90% of the state rangelands (Roesch et al., 2009; Brasil, 2016). The biome is rich in biodiversity and traditionally used for cattle raising (Boldrini et al., 2010), but this scenario has been changing with the introduction of annual crops and forest plantations.

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

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

In the areas with forest plantations, the post-harvest period and the earlier stages of eucalyptus plantation are the ones with the greatest attention in terms of soil loss, since the soil is uncovered (Oliveira et al., 2013). Limiting the size of clear-cut harvest areas and interrupted the linkage between harvest areas and drainage network are classic approaches to mitigate erosion and sedimentation processes in streams post-harvesting (Gimenez et al., 2019). The retention of harvest residue also reduced soil loss and water runoff the first years of plantation establishment (Nambiar et al., 2000). These approaches were observed in the studied area, since only 21% of the planted area was harvested and the riparian vegetation is preserved, as required by Brazilian law. Therefore, the adopted forest management contribute to making sediment yield comparable to the observed during the pre-harvest period.

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

6. Conclusions

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

Staggered forest-harvesting (21% of the EC) led to sediment yield comparable to the observed during the pre-harvest period. Even after partial harvesting and subsequent soil tillage, greater streamflow and sediment yield were lower than the recorded in GC. Thus, well-managed forest plantations may be less prone to sedimentation than degraded grassland under intensive grazing, and contribute for increased land use sustainability in terms of soil erosion and degradation.

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

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

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Table 1

Sampling frequency of the mixture water and sediments, based on discharge, in the grassland (GC) 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

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

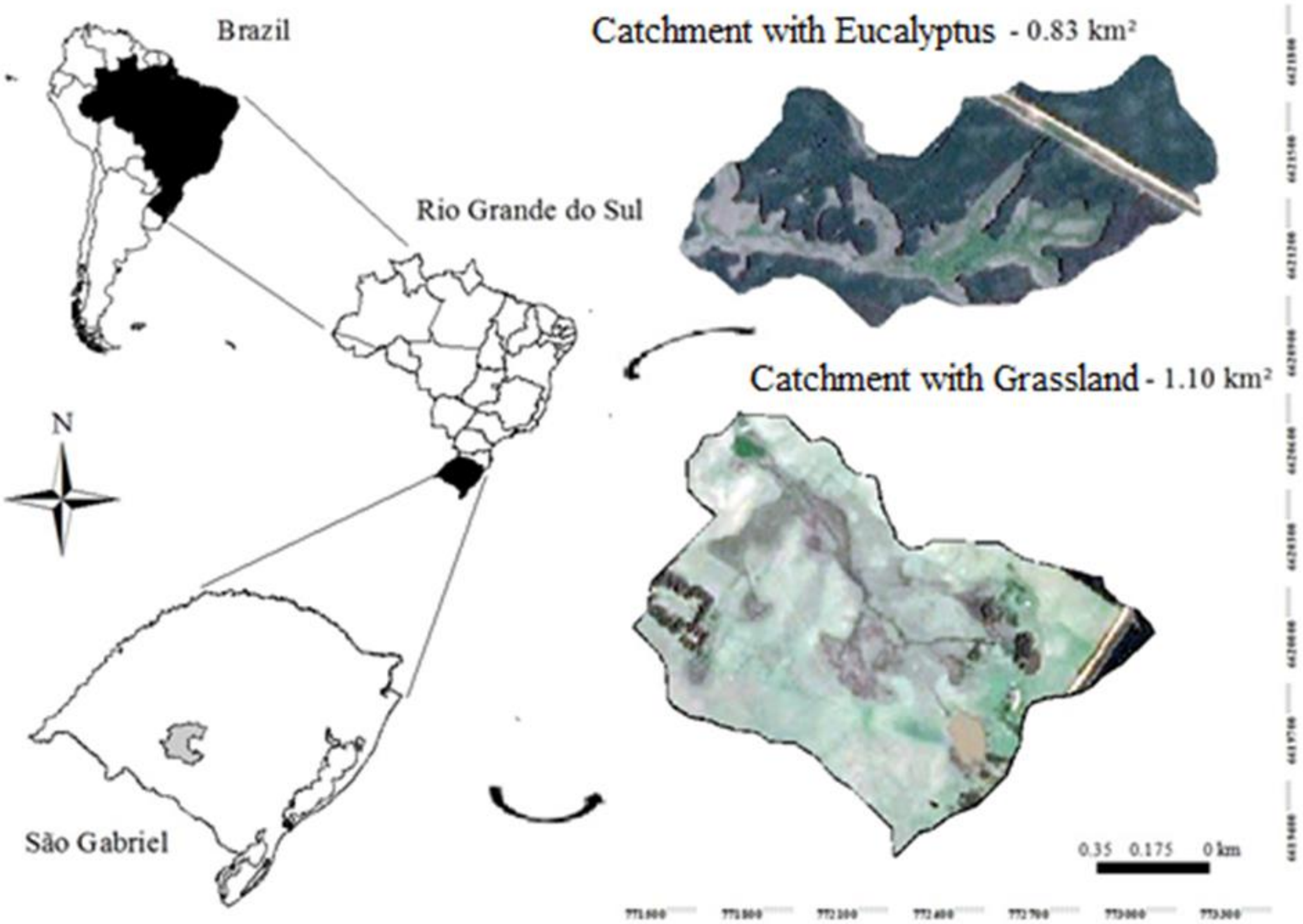

Fig. 1. Location of the municipality of São Gabriel-RS, Brazil, and delineation of the study catchments covered with eucalyptus and grassland.

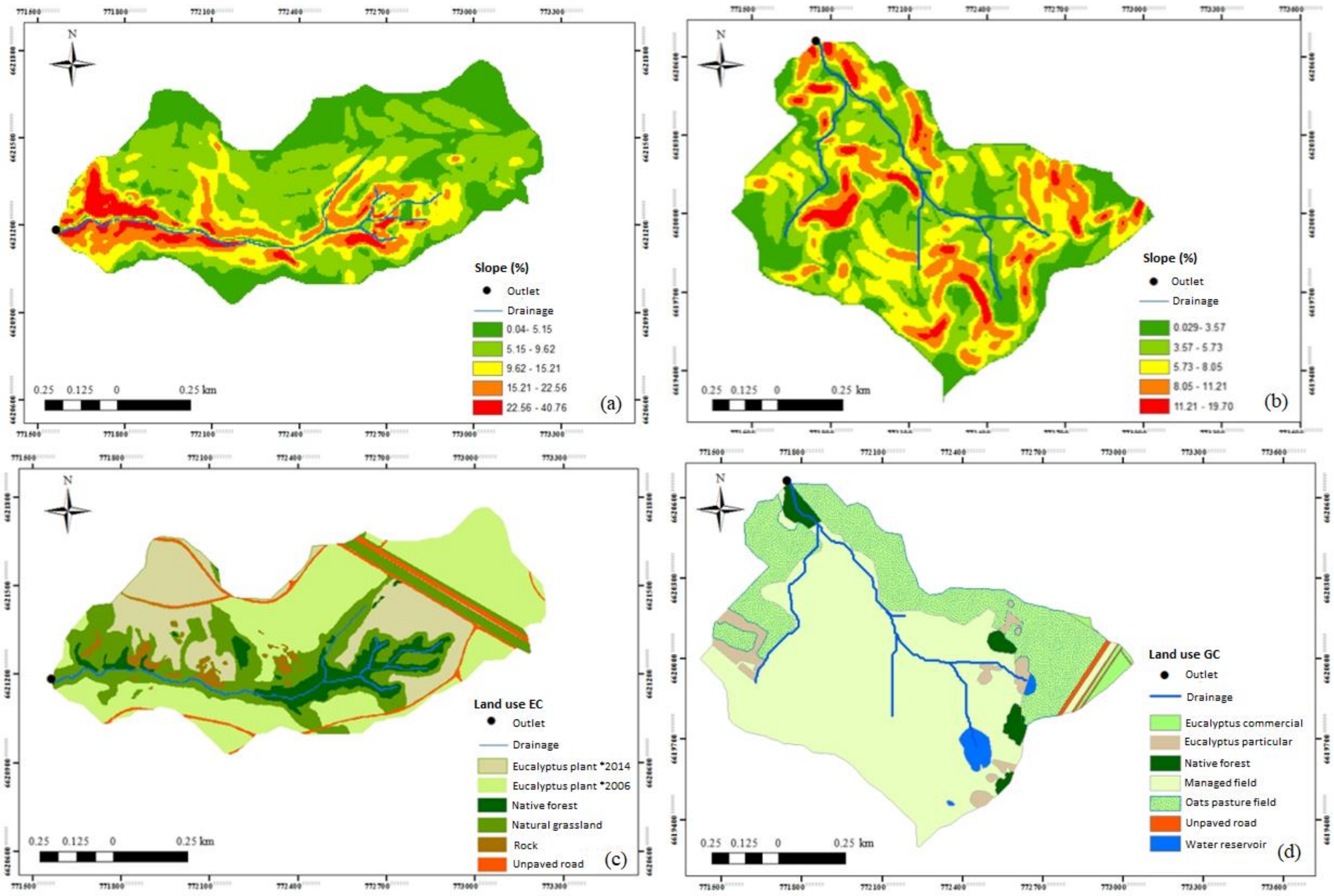

Fig. 2. Maps of slope for eucalyptus (a) and grassland catchment (b), and land use for eucalyptus (c) and grassland catchment (d).

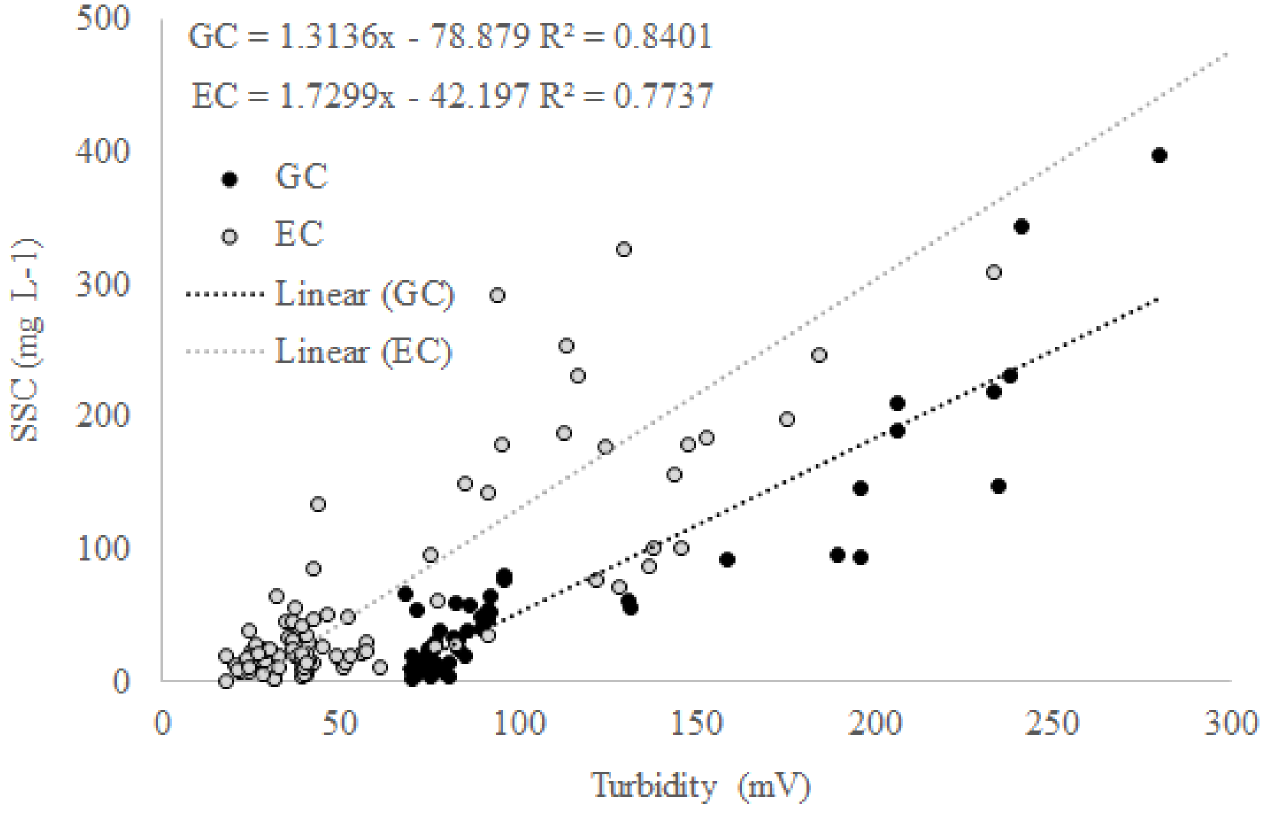

Fig. 3. Relationship between suspended sediment concentrations (SSC) and turbidity in the study catchments covered with eucalyptus and grassland.

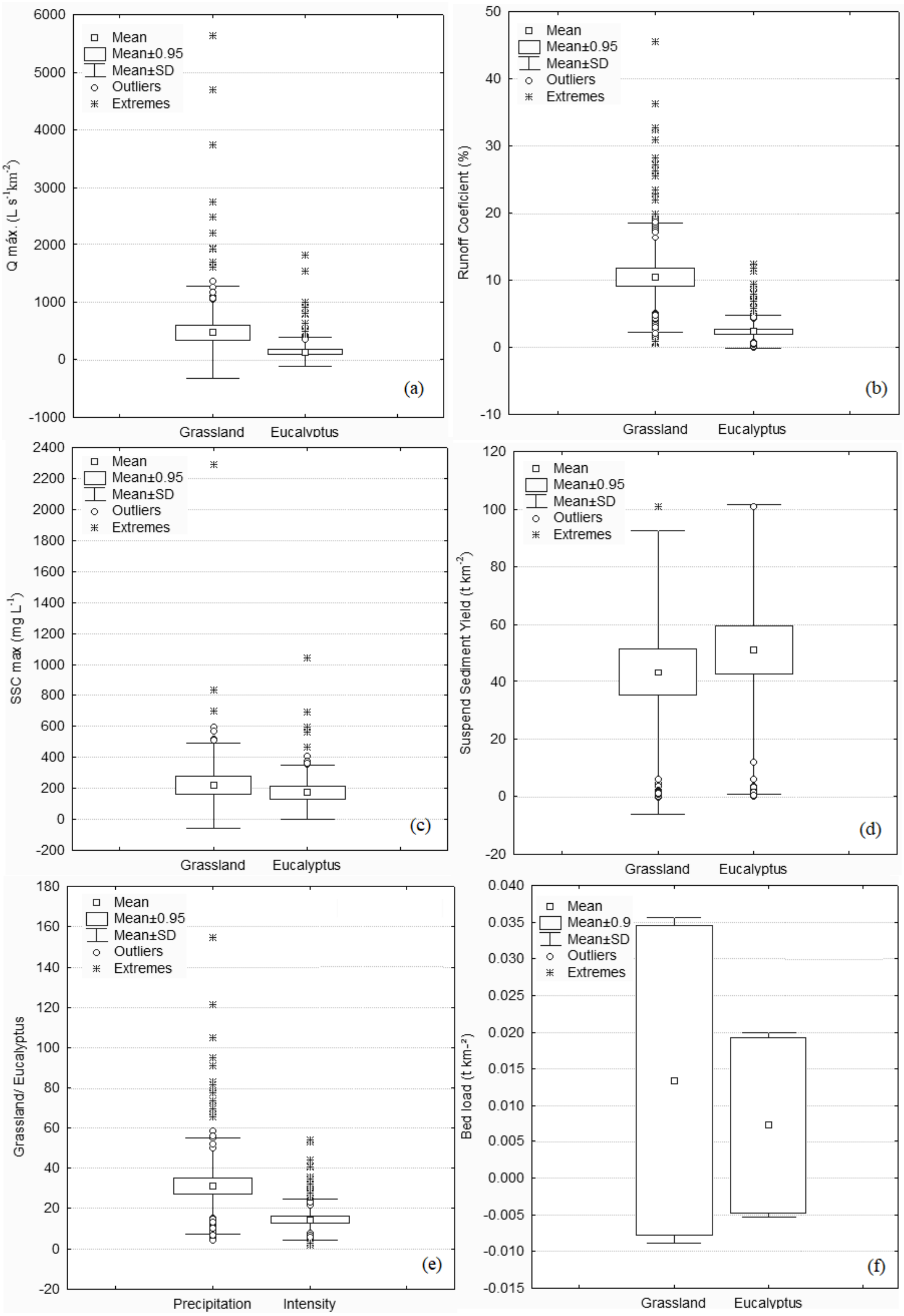

Fig. 4. Box-plot of hydrosedimentologic variables: maximum streamflow (a), runoff or streamflow (b), maximum suspended sediment concentration (c), suspended sediment yield (d), and bed load (e) of the events monitored in the grassland and eucalyptus catchments. As shown in Table 2, medians of the different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

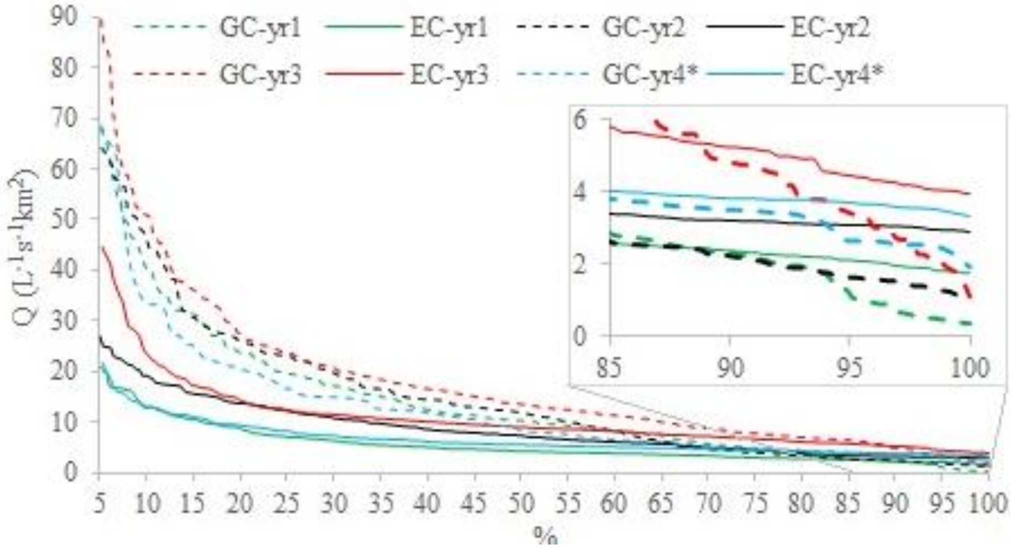

Fig. 5. Exceedance probability curves for streamflow in the grassland (GC) and eucalyptus catchments (EC), during the study period (Sep/2013–Mar/2017).

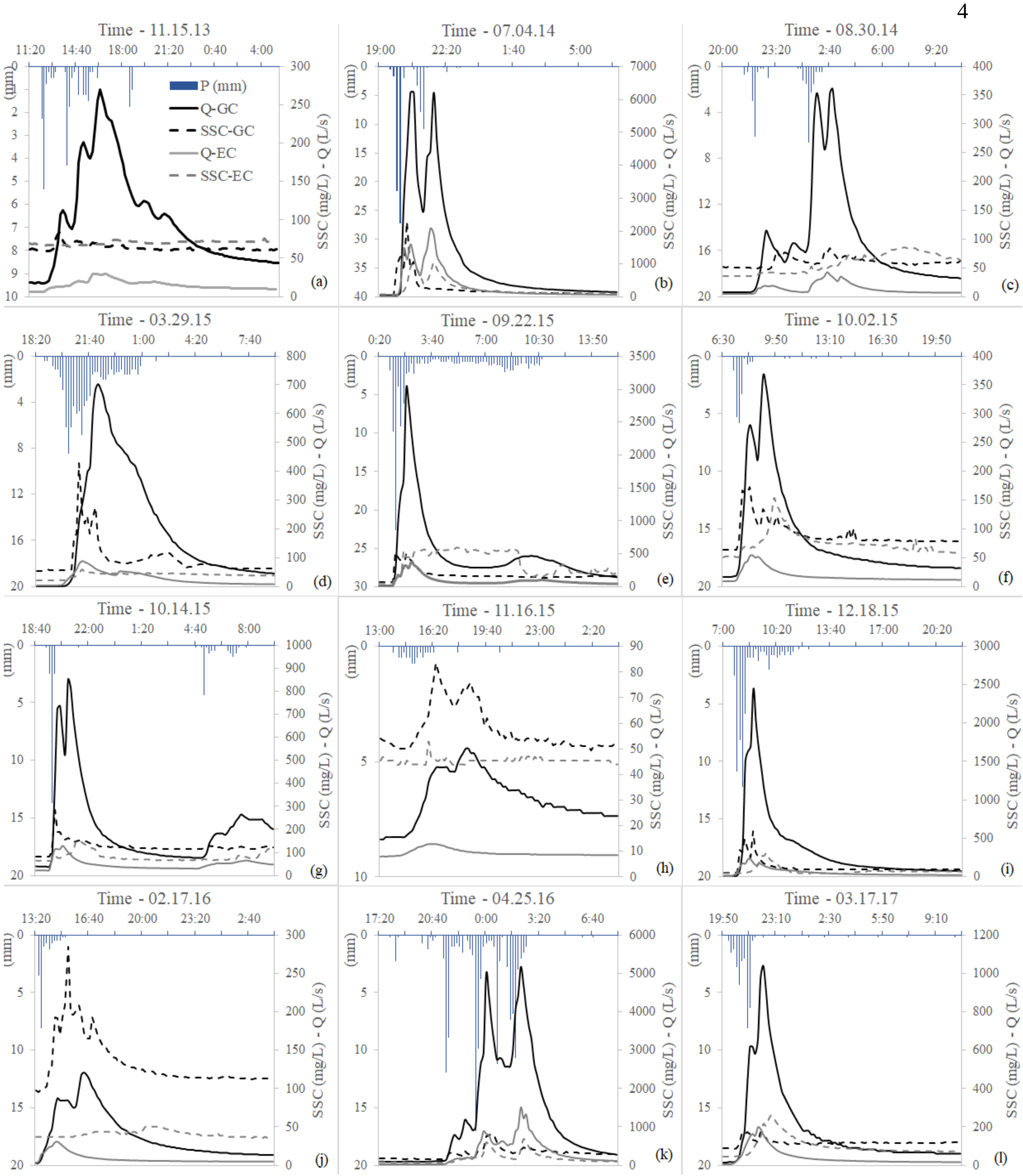

Fig. 6. Time series for events with different water flow (Q), suspended sediment concentration (SSC) and rainfall, selected to represent the typical peak events in the grassland (GC) eucalyptus catchments (EC).

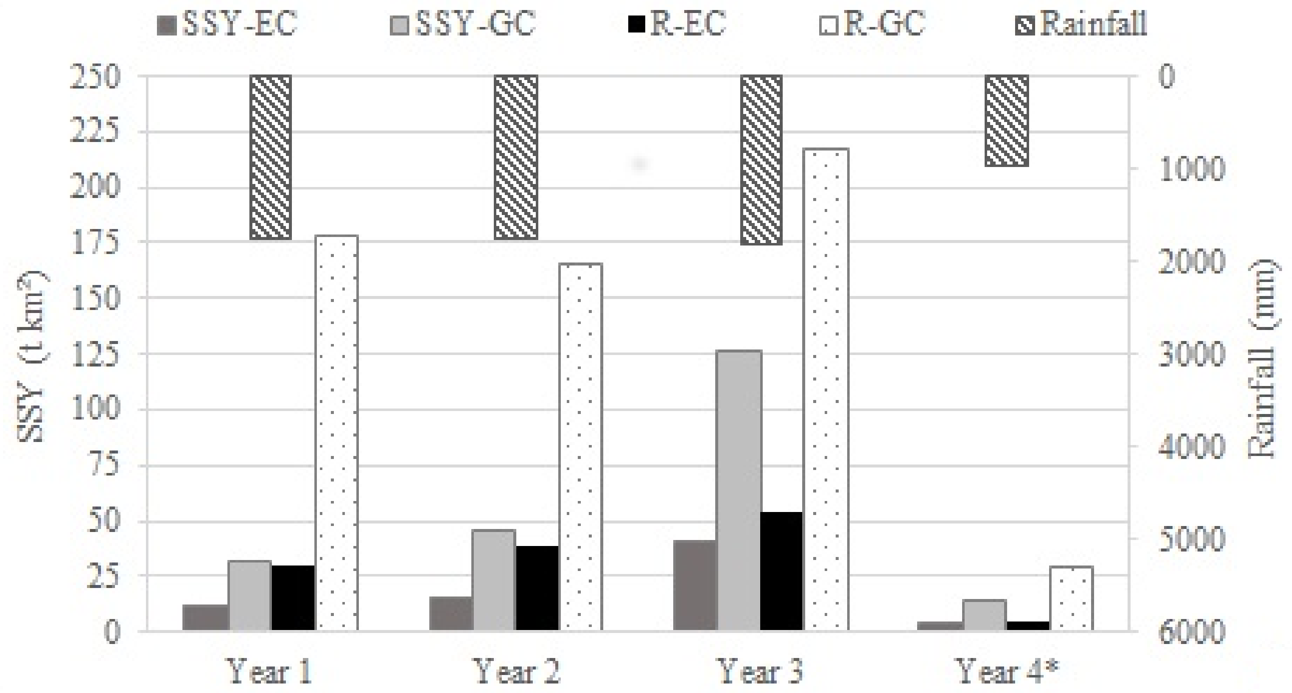

Fig. 7. Annual suspended sediment yield (SSY), streamflow or runoff (R) and rainfall were selected to represent the typical peak events in the eucalyptus and grassland catchments, during the study period (Sep/2013–Mar/2017). *Year 4 corresponds to a half-year, i.e. from Sep/2016 to Mar/2017.

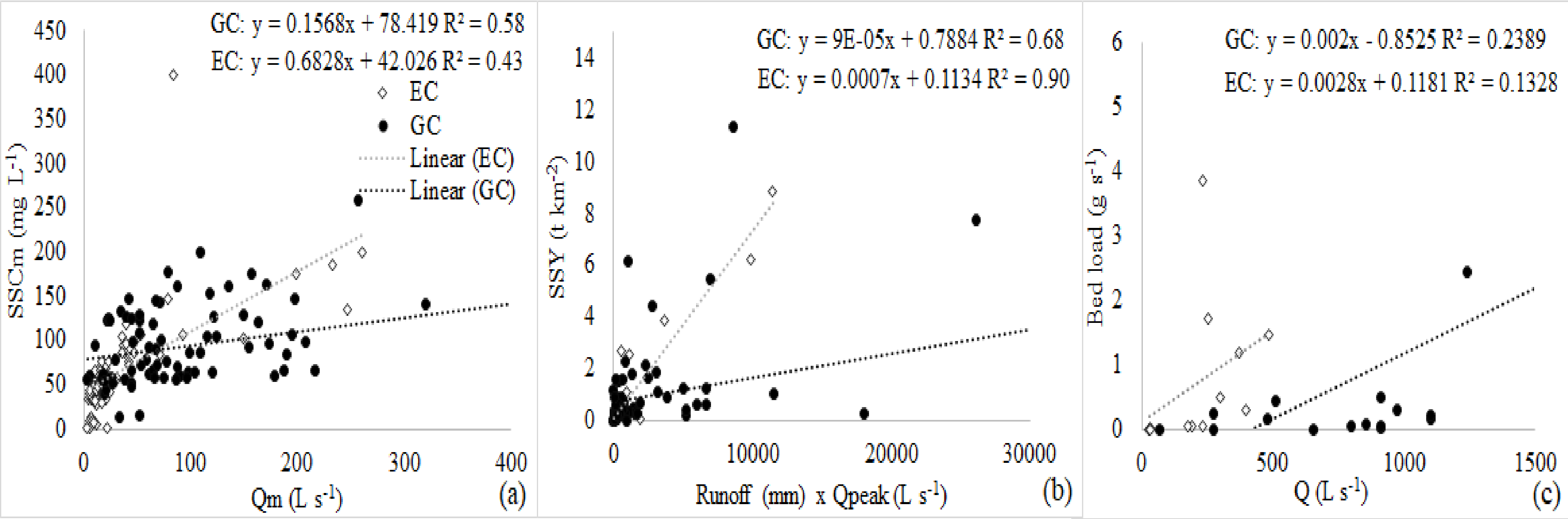

Fig. 8. Relationship between mean of suspended sediment concentration (SSCm) and mean discharge (Qm) (a), suspended sediment yield (SSY) with product of runoff and peak discharge (b), and bed load and discharge (c) in grassland (GC) eucalyptus catchments (EC).

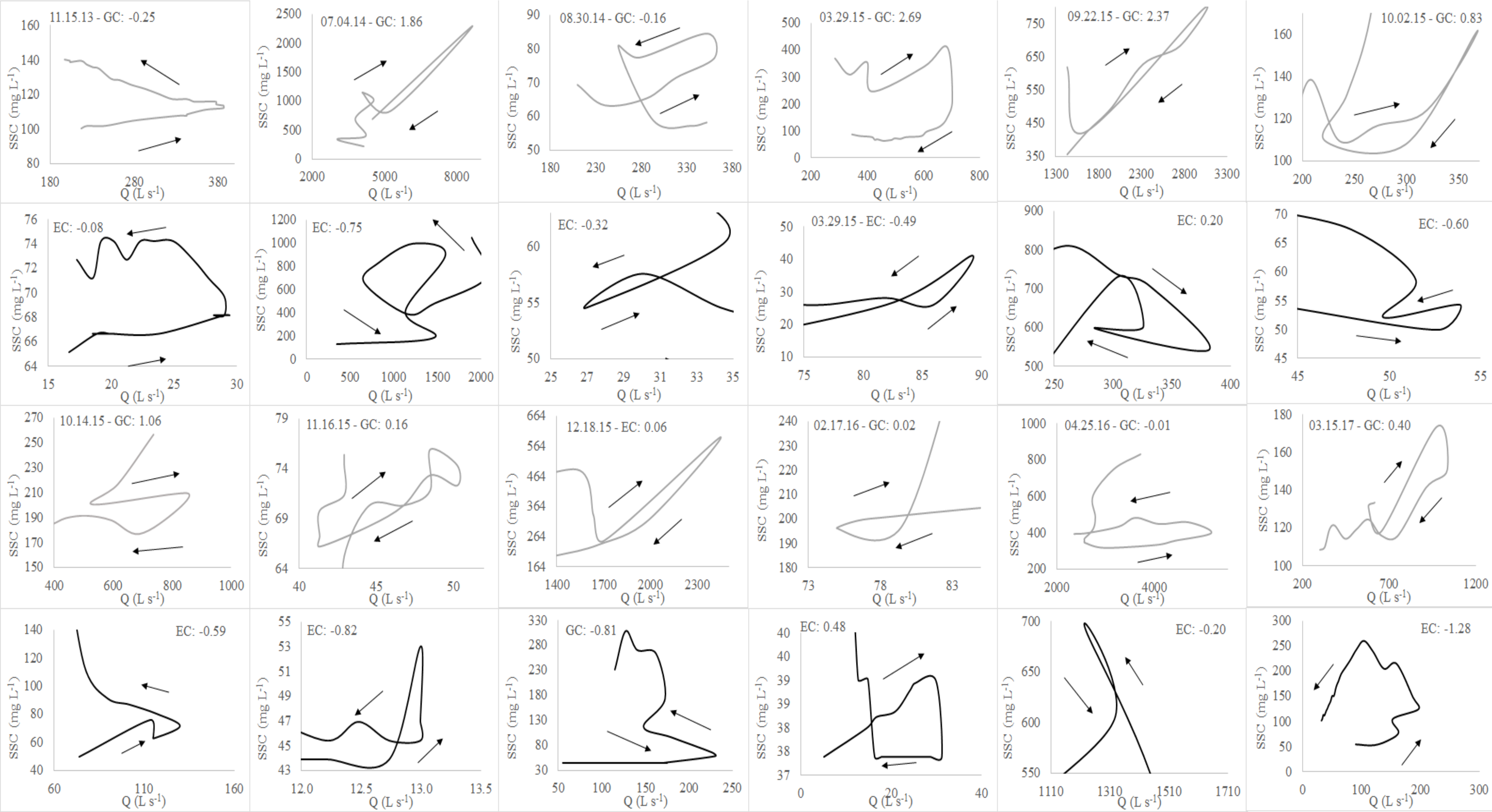

Fig. 9. Hysteresis loops of events in the grassland and eucalyptus catchments.