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Sediment trap efficiency of paddy fields at the watershed scale in a mountainous catchment in Northwest Vietnam

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Abstract

Composite agricultural systems with permanent maize cultivation in the uplands and irrigated rice in the valleys are very common in mountainous Southeast Asia. The soil loss and fertility decline of the upland fields is well documented, but little is known about reallocation of these sediments within the landscape. In this study, a turbidity-based linear mixed model was used to quantify sediment inputs, from surface reservoir irrigation water and from direct overland flow, into a paddy area of 13 hectares. Simultaneously, the sediment load exported from the rice fields was determined. Mid-infrared spectroscopy was applied to analyze sediment particle size. Our results showed that per year, 64 Mg ha^{-1} of sediments were imported into paddy fields, of which around 75 % were delivered by irrigation water and the remainder by direct overland flow during rainfall events. Overland flow contributed one third of the received sandy fraction, while irrigated sediments were predominantly silty. Overall, rice fields were a net sink for sediments, trapping 28 Mg ha^{-1} a^{-1} or almost half of total sediment inputs. As paddy outflow consisted almost exclusively of silt- and clay-sized material, 24 Mg ha^{-1} a^{-1} of the trapped amount of sediment was estimated to be sandy. Under continued intensive upland maize cultivation, such a sustained input of coarse material could jeopardize paddy soil fertility, puddling capacity and ultimately also food security of the inhabitants of these mountainous areas. Preventing direct overland flow from entering the paddy fields, however, could reduce sand inputs by up to 34 %.

1 Introduction

Paddy cultivation is one of the most long-term sustainable cropping systems, as irrigated rice is the only major crop cultivated in monoculture for centuries without severe soil degradation (Bray, 1986; Uexkuell and Beaton, 1992). Two mechanisms facilitate this continuing productivity: first, flooding applies suspended particles and soluble nutrients to the fields that contribute to the indigenous nutrient supply (Dobermann, 1998;

Schmitter et al., 2011). Second, puddling creates an environment of high input and low breakdown of organic matter (Cao et al., 2006; Huang et al., 2015). As nutrient content of sediments is closely related to sediment particle size, and puddling is favored by high clay content (De Datta, 1981), the potential for long-term sustainable rice production is related to the soil texture in paddy fields.

Irrigated paddy fields, however, are not isolated elements in a landscape, as they are connected to surrounding upland areas. They receive sediments from those upland areas, both directly through overland flow, and indirectly from irrigation water released through surface reservoirs (Schmitter et al., 2012). These processes bring sediments into the rice fields, which can alter paddy soil texture (Schmitter et al., 2011). The vast majority of paddy fields in Vietnam are subject to these processes: 97 % of Vietnamese rice is irrigated, and the main water source for irrigated rice in Southeast Asia is water from surface reservoirs (FAO Aquastat, 2014). Therefore, most paddy areas receive sediment-conveying irrigation water.

The amount and nature of sediments in irrigation water depends on their source, i.e. the upland fields surrounding both the paddy fields and the surface reservoirs. Traditionally, in the mountainous regions of Northern Vietnam, Thailand and Laos as well as Southern China, paddy systems have been located in the valleys, surrounded by shifting cultivation on the hills. In Northern Vietnam, 60 % of paddy cultivation is located in such hilly areas, on terraces that form cascades (Rutten et al., 2014).

Traditional shifting cultivation systems are very extensive in space and time, generating very limited runoff and erosion at the watershed scale (Ziegler et al., 2009). In recent years, under the influence of market mechanisms and population pressure, the traditional shifting cultivation systems on the slopes have been replaced by permanent upland cultivation (Ziegler et al., 2009). Implications of these land use changes have been studied in detail on the upland fields, and the increased erosion due to these changes are well documented. In our study area, maize and maize-cassava intercropping on steep slopes resulted in erosion of up to 174 $Mg\,ha^{-1}\,a^{-1}$ (Tuan et al., 2014), coupled with a loss of soil organic matter reaching 1 $Mg\,ha^{-1}\,a^{-1}$ (Häring et al., 2014).

Additionally, changes in texture occurred as fertile silt and clay fractions were exported from the upper and middle slope positions whereas sandy material was deposited at foot slope positions (Clemens et al., 2010). Differences in amount and texture of eroded material from upland fields could therefore entail a shift in matter exchange between upland cultivation and valley paddy rice.

Increased erosion may therefore not only jeopardize the continued production of cash crop maize on upland fields, but also adversely affect the long-term sustainability of the food crop production in the paddies. Schmitter et al. (2010) showed that soil fertility in paddy cascades varies with distance to the irrigation channel, and thus established a link between sedimentation processes and soil properties. Rüth and Lennartz (2008) and Schmitter et al. (2011) found that variability of paddy soil texture and yield were a function of position along the catena, related to differential settling of sediments in irrigation water. If soil properties and yield are closely linked to sedimentation processes, then changes in amount and texture of the sediment inputs have a potential effect on long-term soil fertility and crop production, and hence on food security in the area, as rice is the main staple food crop.

In order to assess these risks, there is a need for reliable data not only on the amount and texture of sediments entering the paddy fields, but also on the quantity and quality of the material exported from the paddies. Because of their terraced structure, paddies can function as a sediment filter in the landscape (Maglinao et al., 2003). But few studies have assessed both inputs and exports. Dung et al. (2009) monitored a watershed in Northern Vietnam with shifting cultivation in the upper area of the catchment and paddy rice in the valley. Annually, for an experimental plot of 0.3 ha, between 11 and 29 Mg of sediments entered the paddies, and from this amount, 27 to 63 % was trapped within the field and the remainder was exported with the runoff. The proportion that remained behind was mostly sandy, and hence altered the soil texture in the experimental paddy plots.

While these results indicate that paddy fields act mainly as a net sediment trap, their function might differ when up-scaled to a larger area as sediment deposition changes over cascade length (Schmitter et al., 2010). Thus, at the watershed level, it is not clear whether paddy fields act as sediment sources or sinks. For example, Mai et al. (2013) found that paddies acted as a green filter, reducing runoff peaks, when their water storage capacity was not yet fully used by irrigation at the onset of the runoff event. But if the maximum storage capacity was already reached, runoff increased, as full paddies are not able to retain any water and so all overland flow was propelled through them, causing high runoff peaks at the catchment outlet.

Therefore, there is a need for a more detailed understanding of sediment fluxes and budgets in paddies at watershed-scale. Our specific aims were to (i) quantify the contribution of overland flow and irrigation water to the sediment inputs of a paddy rice area, (ii) determine if paddy fields are a net sediment source or a sink, (iii) assess the particle size distribution for the sediment input and export from paddy fields, and (iv) evaluate the potential effects of within-watershed sediment reallocation on long-term soil fertility in Chieng Khoi watershed, Northwest Vietnam.

2 Material and methods

2.1 Study site

The study was conducted in a small agricultural watershed, located in Chieng Khoi commune, Yen Chau district, Son La province, North-West Vietnam (21°7′60″ N, 105°40′0″ E, 350 m a.s.l., Fig. S1 in the Supplement). The catchment is 200 ha in size, and sediment reallocation in a sub-catchment of 50 ha which consists of 13 ha of paddy rice and 27 ha of upland fields was monitored in greater detail. In the area, the dominant soil types are Alisols and Luvisols (Clemens et al., 2010) and the climate is monsoonal, with a rainy season from April till October and average annual rainfall of around 1200 mm. Land use in the watershed is characterized by maize and maize-cassava intercropping on the slopes, and irrigated rice in the valleys. The source of irrigation water is a surface reservoir that feeds a concrete irrigation channel, ensuring two rice crops per year: a spring crop from February till June, followed by a summer crop planted in July and harvested in October. The reservoir was formed by the damming of a river that originates in the karst mountains of the area. It has a capacity of 10^6 m^3 and a contributing area consisting of 490 ha of intensively cultivated upland fields and forest. The channel splits in two, just below the reservoir, and feeds two paddy rice areas (6.5 ha each), on the banks of a river that intersects the paddy fields. The irrigation water flows from the channel into the paddy fields, which drain into the river (Fig. 1).

2.2 Hydrological monitoring

Discharge and sediment concentration were monitored at five different locations in the catchment (Figs. 1 and S1). As the irrigation management in the catchment disturbed the relationship between discharge and sediment concentration, a turbidity-based method was used to monitor the sediment concentration. Self-cleaning turbidity sensors (NEP395, McVan, Australia) were installed, with the optical eye down, in a vertically suspended pipe that could float with water level fluctuations, ensuring that the sensor remained approximately at the center point of flow.

Discharge was monitored using pressure sensors (Ecotech, Germany) and the stage-discharge relationship was established using the salt dilution method for the channel and the area-velocity method for the river (Herschy, 1995). Rainfall was measured with a tipping-bucket rain gauge (0.1 mm accuracy, Campbell Scientific, USA) in the upper part of the catchment. The water level of the lake was recorded on a daily basis.

2.3 Sediment concentration predictions

Water samples were collected manually with a storm-chasing approach, where more samples were taken when water level and turbidity were rapidly changing. A typical sampled rainfall event thus consisted of ten to twenty water samples, depending on the duration of the event. Additionally, base-flow samples were collected every two weeks. Each sample consisted of two 500 mL bottles. Sediment concentration in the samples was determined gravimetrically (ASTM, 2013) as recommended for samples with very high Suspended Sediment Concentration (SSC), by letting the sediment settle overnight in cold storage ($< 4\,^{\circ}$C) and then siphoning off the supernatant followed by oven-drying of the sediment at $35\,^{\circ}$C.

Continuous predictions of sediment concentration were then obtained from a linear mixed model (Slaets et al., 2014) with SSC as response variable and turbidity, discharge and cumulative rainfall as predictor variables. To account for temporal correlation in the observations, an error with a first-order autoregressive covariance structure was fitted to the data. The response variable was log-transformed to stabilize the variance, as were the predictor variables discharge and turbidity. Model fit was evaluated with five-fold cross validation using a SAS macro described in Slaets et al. (2014).

2.4 Separating sediment sources

The monitoring locations in the concrete irrigation channel were chosen in order to separate the contributions of irrigation water from the surface reservoir, and Hortonian overland flow, to the sediment inputs into the paddy fields. The station situated furthest upstream in the channel (Location 1 in Fig. 1) was placed directly below the reservoir outlet, and thus monitored the discharge and water quality of the surface reservoir. An additional station (Location 2 in Fig. 1) was installed directly below the split of the concrete channel, and monitored only discharge, as the water quality here was the same as at Location 1. This second location quantified how much of the irrigation was flowing to the left arm of the irrigation channel after the split, and how much was going to the right arm. As the water in the left channel was fully irrigated to the paddy fields in this watershed, no further measurements were conducted in this branch of the channel. But the right channel leaves the watershed, exporting part of the irrigation water from the catchment. Therefore, a measurement station was installed downstream in the channel, at the point where the irrigation channel crosses into a neighboring watershed (Location 3 in Fig. 1).

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In the absence of rainfall, Location 3 received water with the same sediment concentration as the reservoir outflow (Location 1). As there were no other water sources entering the concrete-lined waterway, the hydrological balance when it is not raining can be described by

$$Q_{in} = Q_{irr} + Q_{out}, \quad (1)$$

where Q_{in} is the discharge measured at Location 2, consisting of the irrigation water originating from the reservoir, Q_{irr} the irrigated discharge to the paddies, and Q_{out} the discharge measured at Location 3, as not all irrigation water in the channel was used up fully in this catchment, but a part was transported further to irrigate rice in a watershed downstream. Since Q_{in} is the discharge measured at Location 2 and Q_{out} is the discharge measured at Location 3, Q_{irr} can be calculated as the difference in discharge between those two sites.

During rainfall events, Hortonian overland flow entered the channel directly from the upland fields (Fig. 1), changing the water balance to

$$Q_{in} + Q_{pp} + Q_{of} = Q_{irr} + Q_{out}, \quad (2)$$

where Q_{pp} is the direct rainfall into the channel and Q_{of} the overland flow that enters the channel from the upland area between the upstream and downstream locations. During rainfall, Q_{pp} could be calculated directly from the rainfall intensity and the surface area of the channel. Assuming that the irrigated discharge to the paddy fields prior to the onset of the rainfall remained constant during rainfall, Q_{of} can be calculated using Eq. (2). Flow component separation was performed with the statistical software R. Details of the procedure can be found in Schmitter et al. (2012).

The calculation of sediment loads for these sources requires that not only discharge, but also sediment concentration of each component is known. Rainfall does not contain sediment, so Q_{pp} makes no contribution to the sediment load. The sediment concentration c_{in} of Q_{in} was monitored at Location 1, and c_{out} of Q_{out} at Location 3. The irrigated discharge to the paddy fields, Q_{irr}, had the same sediment concentration as the discharge exported from the watershed at Location 3, assuming full mixing. The sediment load from overland flow can then be calculated from

$$L_{of} = \left[(Q_{irr} \times c_{out}) + (Q_{out} \times c_{out}) - (Q_{in} \times c_{in})\right]. \tag{3}$$

In the river, the water sources are paddy outflow and reservoir overflow. The measurement stations were installed in a similar manner as they were in the irrigation channel, with one station upstream and one downstream of the paddy fields (Locations A and B in Fig. 1). The only sediment input between these two locations was drainage from paddy fields and fish ponds in the paddy area. The river receives outflow from both banks of paddy fields, and we only monitored the overland flow entering the right bank. Therefore, in order to quantify the net sediment balance for the paddy fields, the assumption is made that the upland fields on the left bank of the river generated the same amount of erosion as those on the right bank, as the areas are very similar in land use, slope and size (17 and 20 hectares of contributing area).

There was one additional measurement location in the river further downstream (overall outlet, Fig. S1), at the outlet of a larger watershed of 2 km^2 in which the monitored paddy area was nested, in order to assess scaling effects on paddy watershed sediment losses.

2.5 Sediment load estimates

Instantaneous sediment loads at a time i ($i = 1$ to t) are generally estimated from the continuous discharge data and the continuous sediment concentration predictions according to

$$\hat{L}_i = \hat{Q}_i \times \hat{C}_i, \tag{4}$$

where $\hat{L}_i$ is the estimated instantaneous load at time i in $g\,s^{-1}$, $\hat{Q}_i$ is the estimated discharge at time i in $m^3\,s^{-1}$ and $\hat{C}_i$ is the estimated concentration at time i in $g\,m^{-3}$. The

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time series consisted of two minute intervals. As such, the estimated monthly or annual sediment load $\hat{L}_{1\ \text{to}\ t}$ in grams can be computed by summing up the instantaneous loads, across t measurement intervals of turbidity and discharge:

$$\hat{L}_{1\,\text{to}\,t} = \sum_{i=1}^{t} (\hat{L}_i \times 120). \tag{5}$$

The full sediment load balance for the irrigation channel then equals

$$L_{\text{in}} + L_{\text{of}} = L_{\text{irr}} + L_{\text{out}}, \tag{6}$$

where L_{in} is the sediment load at Location 2, L_{of} is the sediment load brought into the channel by direct runoff during rainfall events, L_{irr} is the load irrigated to the paddies, and L_{out} is the sediment load exported from the channel at Location 3, with each load in the sediment balance in Eq. (6) computed using Eq. (5).

In order to calculate 95 % confidence intervals on the sediment load, a bootstrap method was used that accounts for uncertainty in the discharge and sediment concentration predictions (Slaets et al., 2015; Appendix A). The number of bootstrap replicates was 2000. As the direct sediment load estimation described in Eq. (4) is typically biased downwards when concentration and discharge are predicted on the log-transformed scale (Ferguson, 1986), taking the medians of the bootstrap replicates is a simple approach to bias-correct the estimates (Efron and Tibshiriani, 1993). Therefore, the estimated sediment loads reported in this study are the medians of the bootstrap empirical distribution, rather than the direct estimates from Eq. (4; Slaets et al., 2015; Appendix A).

2.6 Sediment texture with mid-infrared spectroscopy

Diffuse reflectance Fourier transform mid-infrared spectroscopy (MIRS) is a practical alternative to conventional methods for determining particle size distribution on sediment samples, as only 25 mg is needed for analysis and the measurement is not destructive

(Schmitter et al., 2010). A total of 152 samples were analyzed for texture, covering the full range of locations, seasons and flow regimes. A Bruker Tensor-27 mid-infrared spectroscope (Bruker Optik, Germany) was used and three analytical replicates were measured per sample. Baseline correction and atmospheric compensation were performed on each spectrum before averaging the analytical replicates. As the MIRS method requires a subset of the samples to be analyzed with conventional wet analytical methods for calibration and validation, laser diffraction with a Coulter LS 200 (Beckman Coulter, Germany) was performed on 50 samples. Organic matter and carbonates were destroyed prior to laser diffraction analysis and samples were shaken overnight with a dispersing agent (5 mL 2 % sodium metahexaphosphate for 5 g soil). Three analytical replicates were done per sample.

Sand, silt and clay were predicted from the spectral data using Partial Least Squares Regression (PLSR; Wold, 1966). All spectral manipulation and model selection was performed using QUANT2 package within software OPUS 7.0 (Bruker Optik, Germany). Models were evaluated with leave-one-out cross validation. OPUS offers several spectral processing techniques to enhance spectral information and reduce noise. The selection of the most suitable method can be automatized using the OPTIMIZATION function, which selects the method resulting in the highest r^2 of observed vs. predicted values after cross-validation. For sand, the pre-processing method was the calculation of the second derivative of the spectra, which can help to emphasize pronounced but small features over a broad background. After validation, an r^2 of 0.81 was obtained. For silt, multiplicative scattering correction was applied, which performs a linear transformation of each spectrum for it to best match the mean spectrum of the whole set, and the model resulted in an r^2 of 0.83. For clay, no satisfactory model could be obtained, and so the clay percentage was calculated as the remaining amount of sediment after subtracting the sand and silt fractions.

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3 Results

3.1 Hydrological processes driving sediment flows

Model fit for the discharge rating curves varied between locations, with the coefficient of determination ranging from 0.96 to 0.99 (Table 1). As expected, accuracy of the sediment rating curves was lower than that of the discharge rating curves, and explained between 52 and 72 % of variability in the data after cross-validation.

In 2010, a total of 920 mm of rainfall was measured with the onset of the rainy season in April, whereas in 2011, 961 mm fell but rains were delayed, resulting in less rainfall in April–May and a precipitation peak in July. These differences in rainfall pattern led to differences in irrigation patterns between the two years (Fig. 2). Although the total amount of water irrigated to the 13 ha of paddy fields was similar, i.e. 3978×10^3 m^3 in 2010 and 4021×10^3 m^3 in 2011, the seasonal distribution of the irrigated amounts varied between the study years. As the rainy season started late in 2011, there was more water irrigated during the first rice season (February–June) in 2011 (913×10^3 m^3) than in 2010 (700×10^3 m^3). The opposite was true for the summer crop (July–October), during which 1308×10^3 m^3 was irrigated in 2011 compared with 1448×10^3 m^3 in 2010. As the rains came late in 2011, the reservoir was not filled up yet in July at the start of the summer crop, and so there was less irrigation water available.

Variation in rainfall throughout the year was also reflected in the sediment concentration of the irrigation water. In the irrigation channel, the median sediment concentration during base-flow regime was 240 mg L^{-1}. The predicted base-flow sediment concentration fluctuated seasonally, peaking in April and May 2010 and in April, May and June 2011 (Fig. 3b), and resulting in a higher median in those months, between 350 and 430 mg L^{-1}. As for sediment texture, the sand content of the sediments in the irrigation channel during base-flow regime ($n = 18$) varied between 0 and 50 % with an average of 34 % over the whole study period (Table 2). The silt content ranged from 14 to 58 % with an average of 34 %. For clay, the minimum measured content was 0 %, the maximum was 86 % and the average clay content of the sediments was 32 %.

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The median sediment concentration in the irrigation channel during rainfall events was 1200 mg L^{-1}, and the concentration reached a maximum of 70 000 mg L^{-1} (Fig. 3a) during the rainfall event on 12 July 2011, during which 70 mm of precipitation fell in just over one hour. The water samples taken during rainfall events in the channel ($n = 109$) showed a different particle size distribution than those taken during base-flow, with higher proportions of coarser particles: on average, 50 % of sand, 30 % of silt and 20 % of clay were measured during the full duration of rainfall event sampling (Table 2). When only looking at the peak sediment concentration of each event (thus excluding rising and falling limb samples), sand concentrations were higher and varied from 29 to 94 % with an average of 72 % for the 14 measured events.

In the river, the median of the suspended sediment concentration predictions was 300 mg L^{-1} during periods of no rainfall (data not shown). There were no differences in base-flow concentrations between Locations A and B. The river sediment concentrations were very little affected by overland flow as the paddy fields buffered inputs from Hortonian overland flow, and so the maximum concentrations in the river only reached up to 5000 mg L^{-1}. Water samples of Location A in the river, upstream of the paddy fields, had on average 61 % sand, 22 % silt and 17 % clay ($n = 12$, Table 2). After paddy discharge, the river sediment texture on average had 47 % sand, 33 % silt and 20 % clay ($n = 13$, Table 2).

In the river at the overall outlet of the larger catchment, the median base-flow concentration was 190 mg L^{-1} (data not shown). Between Location B and the overall outlet, an additional 47 ha of paddy rice drain into the river, adding filtered irrigation water with lower sediment content to the river, resulting in a lower sediment concentration during base-flow at the overall outlet compared with Location B. During rainfall events, concentration increased at the overall outlet, with a maximum peak of 22 000 mg L^{-1} on 5 June 2010 when 46 mm of rain fell in 160 min. These peak concentrations during rainfall events were higher than those measured at the same time at Location B. As there are point sources of overland flow that reach the stream directly at the overall outlet, the river is not completely isolated from overland flow as it is in Location B where the paddy fields buffered the input of runoff from upland fields, explaining the difference in peak concentrations between these two stations.

3.2 Seasonal sediment load trends in the irrigation system

Monthly sediment loads from irrigation water (Fig. 4) reflected changes in the suspended sediment concentration (Fig. 3b), related to fluctuations in the level of the surface reservoir (Fig. 4) as well as changes in amount of water irrigated to the paddy fields. The first rice crop (from February till June) received about half the water volume of the second crop (Fig. 2), as a smaller area of the paddy fields was cultivated during the spring season, resulting in a lower sediment input from irrigation during the spring season (200 Mg in 2010, 263 Mg in 2011) compared with the summer season (445 Mg in 2010, 346 Mg in 2011). The difference in load between the spring crop and the summer crop was smaller in 2011, as the rains came late that year. Consequently, the reservoir was depleted during the first rice crop and the first rains fell on a much smaller volume of water, increasing the sediment concentration in the reservoir, thus causing the higher sediment load compared with 2010. In the summer season of 2011, the irrigated amount of water was 10 % less than in 2010 (Fig. 2), as the rains came late and the irrigation manager wanted to preserve water. Overall, the largest sediment inputs from irrigation occurred in August in both years of the study (Fig. 4), with 137 Mg of sediments in 2010 and 114 Mg in 2011.

Even though the sediment concentration in the overland flow was orders of magnitude higher than the concentration in the irrigation water (Fig. 3), over a full year, the contribution of irrigation water was about three times larger than the contribution of overland flow (Table 3). As the rainy season starts in April, paddy water inputs from overland flow play a more important role during the second rice crop. The contribution of overland flow was almost negligible during the first rice crop, particularly in 2011 when the onset of the rains was late and the volume of overland flow was much smaller during the first crop (Fig. 4). During that spring cropping season of 2011, the contribution of overland flow to the sediment input of the paddy fields was negligible,

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reaching only 46 Mg compared to 263 Mg from irrigation water. But during July 2011, the month in the study which had the highest rainfall (247 mm), direct overland flow provided almost as much sediments to the paddy fields as irrigation water from the reservoir (62 Mg vs. 71 Mg).

3.3 Sediment budget for paddy fields

Irrigation water from the surface reservoir removed 806 Mg of sediment from the reservoir in 2010 (Table 3). Of this amount, 646 Mg entered paddy fields through irrigation and 160 Mg were exported from the sub-watershed, as the irrigation channel crosses the watershed border into a neighboring catchment. In 2011, the sediment load from the irrigation water was similar with 762 Mg, of which 612 Mg entered the rice fields, and 150 Mg were exported to the next catchment. Using the average textural class percentages of the surface reservoir outflow, irrigation water can be estimated to have contributed 208 Mg of sand, 208 Mg of silt and 196 Mg of clay to the paddy rice fields in the watershed in 2011 (Table 4). As there were not enough samples analyzed to obtain continuous predictions of the different particle size classes using a regression model, simple averages were used for the texture loads. In this sense, all reported sand, silt and clay loads are more a semi-quantitative estimate that provides an order of magnitude, rather than an exact figure.

For the upland area bordering both irrigation channels (37 ha), overland flow generated a sediment load of 249 Mg in 2010 and 278 Mg in 2011. Of this total amount, 193 Mg of overland flow sediments actually entered the paddy fields in 2010 and 219 Mg in 2011. The remainder of the sediments was exported from the watershed through the irrigation channel (Table 3). Again assuming average texture values, the input of overland flow to the paddy fields in 2011 hence consisted of 109 Mg of sand, 66 Mg of silt and 44 Mg of clay (Table 4). Thus the combined addition to the paddy fields from reservoir outflow and overland runoff amounted to 318 Mg of sand, 274 Mg of silt and 240 Mg of clay (Table 4).

The sediment load exported from the paddy fields on both banks of the river, calculated as the difference between Location A and Location B, was 469 Mg in 2011 (Table 3), of which 60 % was exported during the spring cropping season, and 40 % during the summer crop. As the monitoring station in Location B was only installed in 2011, data for 2010 are not available. Combining all of these loads, the difference between inputs and export from the paddy resulted in a sediment yield of 363 Mg in total, or 28 Mg ha^{-1} that remained in the paddy fields in 2011. Since the load exported and the net paddy load are differences between positive numbers (loads measured at Location A minus B for the export, and inputs minus export for the net load), the lower limit of the confidence interval for these two estimates can become negative (Table 3). Negative load estimates can be interpreted as net sediment trapping of the paddy area. Looking at the texture-specific loads (Table 4), the sediments exported from the paddy fields consisted mostly of finer material. Thus, in 2011 approximately 326 Mg of silt and 141 Mg of clay were exported from the rice paddies. Combining inputs and losses, 315 Mg of sand and 99 Mg of clay remained behind in the paddy fields over the whole year, while a net amount of 52 Mg of silt was lost from the 13 ha paddy area (Table 4).

3.4 Watershed sediment yield

The total sediment yield of the sub-watershed, ending at Location B, was 2234 Mg in 2011. This amount was exported via two pathways. First, the irrigation canal distributed 150 Mg from the reservoir and 59 Mg from the upland area through overland flow into the neighboring catchment (Table 3). Second, the river exported 2026 Mg from the sub-watershed at Location B. Of these 2026 Mg, a total of 469 Mg consisted of runoff from the paddy fields. The remaining 1556 Mg that was lost through the river, originated from the surface reservoir as water released via the reservoirs spill-over, which allows excess water to flow into the river whenever the reservoirs maximum capacity is reached. For the larger watershed of 200 ha, which contains the aforementioned sub-catchment, the annual sediment yield was 6262 Mg in 2010 and 5543 Mg in 2011.

4 Discussion

4.1 Upland sediment contribution to the irrigation system

The largest peak of suspended sediment concentration found in this study was two to five times higher compared to the highest values found in other SE Asian studies (Ziegler et al., 2014; Valentin et al., 2008) and the corresponding event contributed 23 % of the total annual sediment load transported by overland flow to the irrigation channel in 2011. The difference in sediment concentration with other studies is most likely due to the more gentle slopes (8 to 15 %) present in the watershed study of Valentin et al. (2008), whereas steep slopes up to 65 % are found in our watershed. Both other studies, however, which contain the highest values found for Southeast Asia in literature, also used a storm-based sampling strategy, underscoring the importance of capturing the highest events in order to reliably assess the erosivity of mountainous catchments. Horowitz et al. (2014) reported that calendar-based sampling typically underestimates constituent transport, while event-based sampling does not. Capturing the highest peaks is crucial, as the importance of single, high-intensity storms for sediment yield in tropical areas is increasing due to climate change. In the monsoon climates of Southeast Asia, a rise in extreme, high intensity rainfall events is expected (IPCC, 2013) and as single large storms already have such a substantial effect on the annual sediment load, in the future they can be expected to dominate annual sediment loads.

Our estimated upland sediment load of 278 Mg a^{-1} in 2011 translates into an annual soil loss of 7.5 Mg ha^{-1}, but this result should be interpreted as an average yield at the watershed level, not as a representative erosion rate at the plot level. This estimate is well within the order of magnitude reported by watershed-scale measurements. For instance, Valentin et al. (2008) monitored sediment yield from 27 catchments in mountainous Southeast Asia and found an average total annual sediment yield of 3.4 Mg ha^{-1}. Plot scale studies, however, frequently report larger erosion rates than the 7.5 Mg ha^{-1} found in our study. Also in the Chieng Khoi commune, Tuan et

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al. (2014) recorded an erosion rate averaging 44 Mg ha^{-1} a^{-1} for sediment fences in unbounded plots for maize-cassava intercropping systems. This discrepancy is typical when upscaling erosion rates (de Vente and Poesen, 2005), as processes are not linear. Erosion can be concentrated at certain hotspots and rill erosion, and internal deposition and filtering processes (e.g. hedges) leave part of the eroded sediments behind within the watershed (Verstraeten and Poesen, 2001). Indeed, in our watershed, the mix of homesteads, maize and maize-cassava cropping and trees on the hills affect both sediment delivery pathways and re-deposition opportunities. The plot-level soil loss on upland fields can thus be expected to exceed the value of 7.5 Mg ha^{-1} that enters the irrigation channel, as a proportion of eroded sediments will be deposited before ever reaching the channel. Nevertheless, even using the conservative estimate of 7.5 Mg and assuming a bulk density of around 1.2 g cm^{-3}, this result entails a loss of around 0.6 mm of soil per year, a value that is well above the soil loss of 2.5 Mg ha^{-1} a^{-1} that is generally considered tolerable (Schertz, 1983).

4.2 Sediment trap efficiency of paddy fields

Surface reservoir water was the largest contributing source to suspended sediment inputs for the paddy fields, with only one quarter of sediment inputs to the paddy fields coming from overland flow in both years. When looking at the sediment quality rather than sediment loads, however, the importance of overland flow increased for sand, with 34 % of the total paddy inputs originating from erosion in 2011. Therefore, while irrigation was the main driver behind water and sediment fluxes in this irrigated catchment, overland flow plays an important role in transfers that could affect plant production and long-term soil fertility.

Paddy runoff amounted to a total of 469 Mg for the 13 ha area in 2011, or 36 Mg ha^{-1} a^{-1} of sediments leaving the rice fields. The majority of paddy sediment export (60 %) took place during the spring season, and can thus be related to overland runoff flowing through the paddies early in the year, when upland fields were bare as the maize crop was not yet established. Hence, intensive land preparation for maize planting and lack of soil cover in spring resulted in a large supply of readily erodible material on the hills. Short-duration, high-intensity spring storms combined with this sediment supply, led to rapid and large inputs of sediment which passed through the paddies. As a result, sediments had little time to settle, thus reducing filter effectiveness of the rice fields and culminating in less trapping and more sediment export from the paddies during the first crop.

Comparing inputs to paddy field exports suggests that the rice area trapped 44 % of the combined re-allocated sediments from reservoir irrigation water and direct runoff from the upland areas. Similarly, Mingzhou et al. (2007) found that the sediment load in the irrigation water resulted in a net deposition, rather than erosion from paddy fields, which led to an additional 4 cm of top soil through irrigation deposits after fifty years of irrigation. While the paddies in our study were a net overall sediment sink, results also showed that the sand fraction was preferentially deposited and was in fact almost entirely captured in the paddies, forming a net deposition of 23 $Mg\,ha^{-1}\,a^{-1}$. About half of the imported clay remained behind in the fields, or a total of 8 $Mg\,ha^{-1}\,a^{-1}$. For silt, the overall balance was negative, with 5 $Mg\,ha^{-1}$ of silt exported on an annual basis. This preferential deposition is likely to have consequences, as long-term fertility of paddy fields is contingent upon the particle size distribution of the soils for physical soil properties, e.g. clay content exceeding 20 % is favorable for puddling (De Datta, 1981). In our study area, top soil in the paddy fields is predominantly silty, with an average of 19 % sand, 68 % silt and 13 % clay (Schmitter et al., 2010). With an estimated deposition of 23 $Mg\,ha^{-1}\,a^{-1}$ of sand and 8 $Mg\,ha^{-1}\,a^{-1}$ of clay in the paddies, and a removal of 4 $Mg\,ha^{-1}\,a^{-1}$ of silt, textural changes can be expected to take place over time. While the clay fraction is expected to add sediment-associated nutrients to the paddies, and thus increase the indigenous nutrient supply for rice, the sand deposits are much larger (76 % of all inputs) and will thus drive the long-term fertility changes in paddy topsoil. Assuming a puddling depth of roughly 25 cm and a bulk density of 1.2 $g\,cm^{-3}$, the sand fraction would dominate after approximately fifty years of these continued inputs. But

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not all fields would have the same longevity, as sediment inputs do not affect the fields equally. Previous research has shown that sedimentation in rice cascades shows spatial variability, and that fields closest to the water source receive most of the coarse material, and the yield declining with decreasing distance to the water source (Schmitter et al., 2010). Thus for certain fields closer to the water source, sand content would increase more rapidly, which is indeed already visible in the study area: paddies higher up on the cascades were often seen to display poor water holding capacity.

Similar composite agricultural systems with permanent upland cultivation on the hills and irrigated rice in the valleys contain 60 % of the total paddy area in Northern Vietnam (Rutten et al., 2014). Consequently, a large agricultural area is potentially affected by such upland-lowland linkages. Eliminating the direct entry of Hortonian overland flow into the irrigation channel, for example by runoff ditches, is one way to prevent up to one third of the total sand inputs from entering the rice fields and thus to protect the food security of the people in the mountainous areas of Northern Vietnam, who depend on rice as their staple food. This solution is not sustainable in the long run from a systems-approach perspective, as the fertility loss of the uplands would affect income when the cash crop income is declining. But with the current high maize prices, it is challenging to identify sustainable hillside land uses that are attractive to local stakeholders (Keil et al., 2009), and deviating direct runoff from entering the paddies would at least be an interim solution. It would, however, also lead to substantial losses of nutrients (Dung et al., 2008) which could not be recycled.

4.3 Buffer capacity of the reservoir

For the sediment yield measured at Location B, the outlet of the sub-watershed, the vast majority of sediments (1557 Mg out of 2064 Mg) stem from the reservoir which spills over into the river when it reaches maximum capacity. In that sense, the bulk of sediments are merely passing through the sub-watershed, having been captured in the reservoir after runoff from the surrounding 490 h of upland fields. Reservoir outflow is thus not only the largest contributor to sediment transport in our paddy area

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within the watershed, but also has a propagating effect beyond the watershed scale: the river water leaving the watershed is either re-used for irrigating paddies in downstream catchments, or will finally end up in the Da river. In either case, the surface reservoir buffers direct sediment inputs that could negatively affect paddy production and river water quality, as average sediment concentrations released from the reservoir were much lower than those measured during rainfall events in the channel (240 mg L^{-1} vs. 1200 mg L^{-1}).

The water in the reservoir also had a lower sand and higher silt and clay content, and sediment profiles in the lake indeed confirmed this preferential settling of coarse material. Weiss (2008) showed that soil profiles taken at the lake bottom had a sediment texture of between 40 and 75 % sand, 20 to 50 % silt and 5 to 14 % clay. The reservoirs filtering effect can be expected to be stronger beyond the watershed, as coarser particles will be trapped preferentially in closer vicinity to the source. While large enough to substantially affect rice production, the amount of sediments trapped by the paddy fields is moderate (12 %) compared to the total amount exported from the watershed by reservoir spillover export. In light of these proportions, effects of climate change and declining soil fertility in upland areas will not remain on-site but can be expected to propagate beyond the watershed, and also affect areas further downstream.

5 Conclusions

The sediment budget for a 13 ha paddy area in a composite agricultural system with permanent maize cultivation on the uplands showed that rice fields at the watershed level are a net sink for sediments, i.e. trapping 46 % of the total sediment inputs. Irrigation water, providing 74 % of the total inputs of 832 Mg, was a larger contributor than direct overland flow from the surrounding upland fields. The irrigation water, however, provided predominantly silty material, while direct runoff sediments had a sandy texture. In the past, extensive swiddening systems with their diverse landscape patterns would have delivered little and mostly fine, fertile sediments to paddy fields via

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direct overland flow. Recent intensification of upland cropping has transformed these previously beneficial inputs into an increased risk for the long-term sustainability of rice production, thus threatening productivity of both upland cropping and paddy yields. The reservoir, however, acts as a buffer by protecting both the rice fields within the watershed, and paddies and water quality further downstream, from unfertile sediment inputs – thus expanding the life time of the paddies.

Our results show the importance of quantifying upland-lowland linkages within and between watersheds, and can be used by scientists, policy makers and extension services to give suitable recommendations to the large group of people in mountainous Southeast Asia who, under influence of population pressure, have gone from practicing composite swidden agriculture to an intensified cropping system with permanent maize cultivation on the hills. Preventing overland flow from reaching the paddy fields, for example, could prevent up to 8 $\mathrm{Mg\,ha^{-1}\,a^{-1}}$ of sand per year, or one third of the total sand deposits, from entering the rice fields. More diversified, sustainable and acceptable approaches, however, benefitting both upland fields as well as downstream paddies, need to be developed at the same time.

Appendix A

Calculating a measure of uncertainty on a sediment load is not trivial. The final value is a sum of instantaneous loads, and those loads are the product of two predicted values, concentration and discharge, which are not independent of each other, as discharge is a predictor variable for concentration. Additionally, the predicted values are on the transformed scale, and there is serial correlation in the sediment concentration data, as samples are taken closely together in time.

In order to calculate 95 % confidence intervals on the sediment loads, a bootstrap method was developed that addresses all of these issues (Slaets et al., 2015). The bootstrap is a Monte Carlo-type method that generates the sampling distribution of a statistic by resampling a large number of times, either from the original observations or from a parametric distribution, to obtain new bootstrap datasets, on each of which the sediment load is calculated. This large number of bootstrap sediment loads provides an empirical distribution, which can be used to estimate the 2.5th and 97.5th percentiles. These percentiles are the limits of the 95 % confidence interval (Efron and Tibshirani, 1993). In our dataset, 2000 bootstrap replicates resulted in smooth histograms and reproducible percentiles. The developed method thus accounts for uncertainty in the parameter estimates of both the discharge and sediment rating curves, and uncertainty due to residual scatter in the sediment concentrations. In this approach, the final bootstrap process consists of three steps:

1. Non-parametric bootstrapping of the (stage, discharge) pairs in order to obtain 2000 bootstrap stage-discharge equations, and thus 2000 time series predictions for bootstrapped discharge.

2. Non-parametric bootstrapping of the sediment concentration dataset, by drawing whole events (to keep the serial correlation intact) and individual base-flow samples, resulting in 2000 bootstrap sediment rating curves, and thus 2000 time series predictions of continuous suspended sediment concentration.

3. Adding a simulated error term to the concentration predictions to account for inherent residual scatter in the data and to facilitate the back-transformation from the log-scale.

Data availability

The source code for the bootstrap analysis with the SAS software that was used for the load estimates and corresponding confidence intervals is freely available at https://www.uni-hohenheim.de/bioinformatik/beratung/index.htm together with necessary input files for testing. The full dataset is available from the authors upon request (hanna.slaets@gmail.com).

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Table 1. Number of observations (n), coefficient of determination (R^2) and method used for stage-discharge relationship (Q); and number of observations and Pearson's correlation coefficient (r^2) after five-fold cross-validation for suspended sediment concentration predictions (SSC).

	Stage-discharge relationship (Q)			Suspended sediment concentration (SSC)	
	n	R^2	Method	n	r^2
Channel (1)	6	0.99	Salt dilution	Identical to location 3	
Channel (2)	6	0.99	Salt dilution	Identical to location 3	
Channel (3)	6	0.96	Salt dilution	327	0.72
River (A)	9	0.99	Area-velocity	145	0.52
River (B)	8	0.98	Area-velocity	71	0.66
River (main outlet)	15	0.98	Area-velocity	228	0.56

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Table 2. Average sediment particle size distribution measured at the different measurement locations for the different components of the paddy area sediment balance.

Sediment source	% sand			% silt			% clay		
	min	av	max	min	av	max	min	av	max
Reservoir water – Location 1	0	34	50	14	34	58	0	32	86
Overland flow	0	50	100	0	30	61	0	20	61
River – Location A	29	61	89	9	22	40	0	17	80
River – Location B	1	47	74	17	33	47	9	20	53

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Table 3. Sediment inputs from irrigation water and overland flow from the 37 ha upland area in the sub-watershed, and sediment export and trapping by the 13 ha paddy area (Figs. 1 and S1). Loads are estimated as the median of the bootstrap estimates (Med), and 95 % confidence intervals are shown (LL = lower limit, UL = upper limit) in Mg per year.

	Sediment load (Mg a^{-1})					
	2010			2011		
Sediment source	LL	Med	UL	LL	Med	UL
Reservoir water:						
Total to channels	617	806	1123	587	762	1331
Irrigated to paddies	492	646 (77 %)	903	496 (74 %)	612	1085
Exported via channel	124	160	222	117	150	248
Spill-over to river	nd	nd	nd	917	1556	18128
Overland flow:						
Total to channels	121	249	303	129	278	516
Irrigated to paddies	119	193 (23 %)	302	110	219 (26 %)	517
Exported via channel	36	56	88	35	59	135
Total paddy input		839 (100 %)			832 (100 %)	
Paddy outflow	nd	nd	nd	−361	469 (56 %)	2555
Net paddy balance	nd	nd	nd	−1625	363 (44 %)	1586
Paddy balance per ha					28 Mg ha^{-1} a^{-1}	

nd = not determined.

Table 4. Texture -specific sediment inputs from irrigation water and overland flow from the 37 ha upland area in the sub-watershed, and texture-specific sediment export and trapping by the 13 ha paddy area (Figs. 1 and S1).

	Load (Mg a-1)					
	2010			2011		
Sediment source	Sand	Silt	Clay	Sand	Silt	Clay
Reservoir water:						
Total to channels	274	274	258	259	259	244
Irrigated to paddies	220	220	207	208	208	196
	(70 %)	(79 %)	(84 %)	(66 %)	(76 %)	(82 %)
Exported via channel	54	54	51	51	51	48
Spill-over to river	nd	nd	nd	950	343	265
Overland flow:						
Total to channels	124	75	50	139	83	56
Irrigated to paddies	96	58	39	109	66	44
	(30 %)	(21 %)	(16 %)	(34 %)	(24 %)	(18 %)
Exported via channel	28	17	11	30	18	12
Paddy input (100 %)	316	278	246	318	274	240
Paddy outflow	nd	nd	nd	2	326	141
Net paddy balance	nd	nd	nd	+315	−52	+99
				(99 %)	(40 %)	

nd = not determined.

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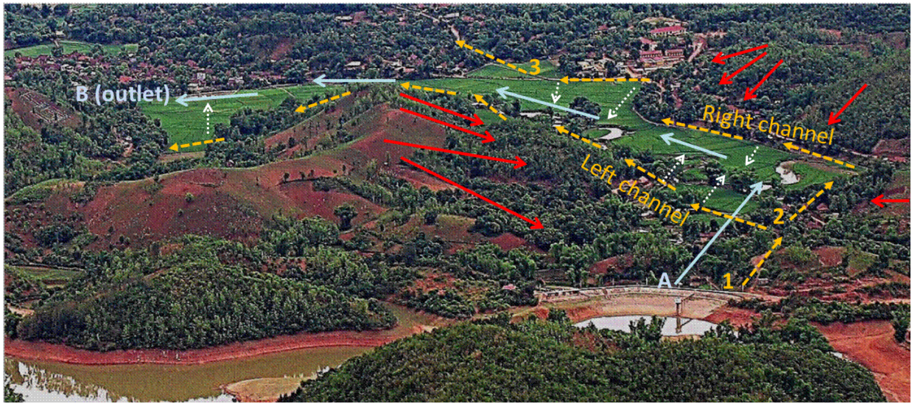

Figure 1. Sediment sources and water flows into and out of paddy rice fields in Chieng Khoi watershed. The dotted yellow arrows show the irrigation channel leaving the reservoir and splitting in two, feeding the two banks of paddy rice. The rice fields subsequently drain into the river, which is indicated by the blue arrows. During rainfall, runoff generated on the uplands flows into the irrigation channel and the paddy fields (red arrows). Measurement locations are indicated with numbers in the channel (1: reservoir outflow, 2: channel split, 3: channel leaving watershed) and with letters in the river (A: river before paddy fields drainage, B: river after paddy fields drainage).

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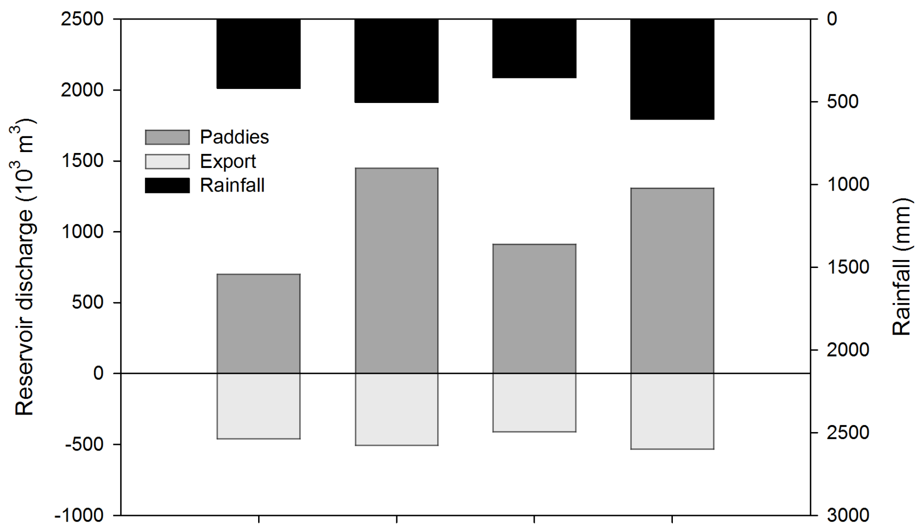

Figure 2. Total discharge from the reservoir irrigated to the 13 ha paddy area draining between Locations A and B in the river, and total discharge exported from the sub-watershed via the irrigation channel at Location 3, per rice crop (spring, summer) per year, and amount of rainfall per rice crop per year.

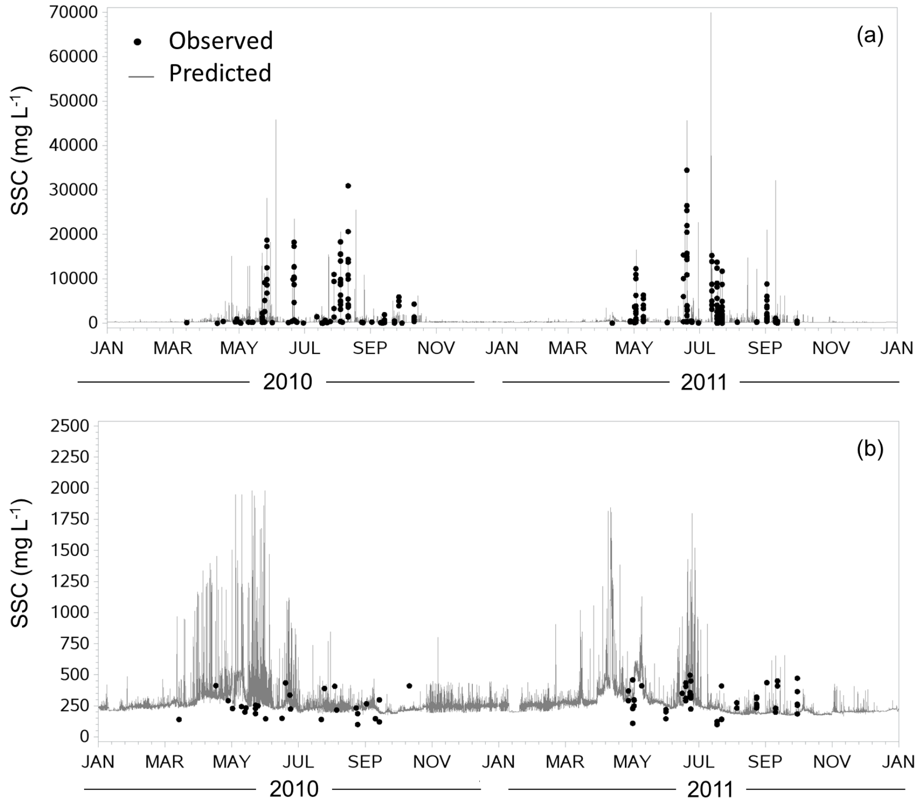

Figure 3. Observed and predicted sediment concentrations (in $mg\,L^{-1}$) for Location 3 in the irrigation channel **(a)**, and zooming in on base-flow, showing only non-event samples and concentration predictions **(b)**.

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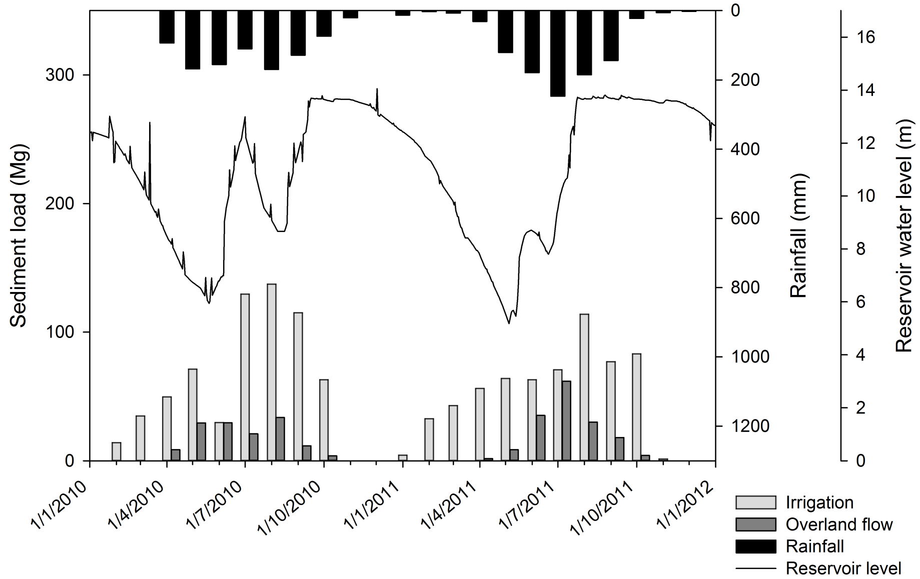

Figure 4. Monthly variations in rainfall, reservoir water level, and sediment load inputs to the paddy fields, both from the surface reservoir and from overland flow.

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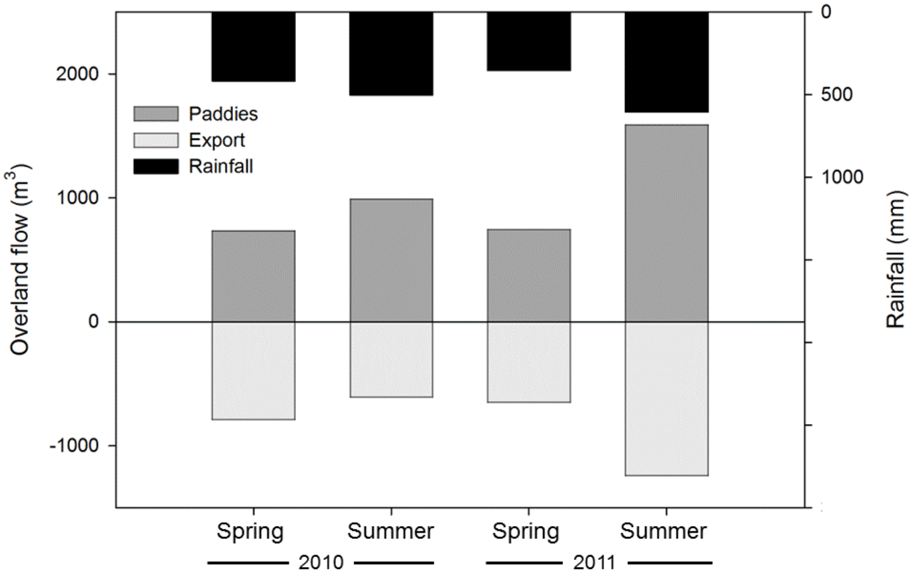

Figure 5. Total amount of water from overland flow during rainfall events, irrigated to the paddy fields and exported out of the sub-watershed via the irrigation channel per rice crop per year, and amount of rainfall per rice crop (note the different units on the *y* axis compared to Fig. 2).

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