**Title:** Assessing multi-scale effects of natural water retention measures on in-stream fine bed material deposits with a modeling cascade

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## 1. Introduction

Freshwater ecosystems are threatened by multiple pressures including organic and inorganic pollution, geomorphological alterations, land-use change, and climate change (Borgwardt et al., 2019; Grizzetti et al., 2017). Complex interactions between such diverse pressures occur at different spatial scales (Bracken et al., 2015; Stoll et al., 2016), and the resulting physicochemical, morphological, and habitat alterations impact water use, quality, security, and biodiversity (Strayer and Dudgeon, 2010; Vörösmarty et al., 2010), with subsequent losses of ecosystem functions and services (Dodds et al., 2013).

Natural water retention measures (NWRMs) include a broad set of in-stream, off-stream, structural, and management practices that aim to mitigate negative impacts of human activities on freshwater ecosystems (Burek et al., 2012; Collentine and Futter, 2018; European Commission, 2013, 2012). Recently introduced in the European policy, NWRMs are part of the broader concept of "Nature Based Solutions", as they use natural processes and cycles to generate benefits to society and the environment (Keesstra et al., 2018; Nesshöver et al., 2017). NWRMs are targeting the alteration of hydrological, nutrient and sediment cycles, similarly to agricultural and forestry best management practices (Liu et al., 2017). However, with NWRMs the focus is on the restoration and rehabilitation of degraded aquatic ecosystems. Therefore, the planning and implementation of NWRMs can take place at very different spatial scales, ranging from local habitats to catchments. In this regard, at least two challenges exist that hinder the realization of the full potential benefits resulting from the implementation of NWRMs.

The first challenge is related to the development of tools or tool-kits able to propagate the effects of NWRMs at different spatial scales. River systems are shaped by interactions between processes acting at very different spatial scales, ranging from catchment (hydrological response, soil erosion, sediment transport), down to the reach meso- and micro- scale (local hydraulics, nutrient processing in the hyporheic zone, sediment deposition and resuspension; Jähnig et al., 2012; Stoll et al., 2016). Current studies have used large-scale, catchment hydrological models (> $10^2$ $km^2$) to predict biotic and abiotic responses to the implementation of some NWRMs at the reach scale (Einheuser et al., 2012; Sommerlot et al., 2013; Woznicki et al., 2015). However, such developments neglect the important role of meso- and micro-scale (< $10^2$ $m^2$) conditions for local hydraulics, nutrient, and sediment transport (Guse et al., 2015; Kail et al., 2015). Modeling cascades (MCs) integrate models with different spatial scales and purposes (Jähnig et al., 2012; Lian et al., 2007). For example, Baldan et al. (2020) integrated a catchment scale hydrological and sediment generation model with a reach-scale hydraulic model and a mesohabitat-scale sediment deposition model. Thus, MCs offer the potential to propagate the effects of NWRM along different spatial scales.

The second challenge is related to the scale at which the effects of NWRMs are assessed (Zhang and Chui, 2019; Zhang et al., 2017). Hydrological models are used for assessing the effectiveness of NWRMs at the catchment scale (Liu et al., 2017; Rao et al., 2009). This approach supports the planning process by providing information on the cumulative effectiveness, compliance with regulatory frameworks (Lam et al., 2011), and tradeoffs with other environmental or societal sectors (e.g. agricultural production; Haas et al., 2017). It also allows the exploration of different targeting, allocation, and optimization strategies (Teshager et al., 2017; Yang and Best, 2015; Zhang and Chui, 2018). However, this approach does not evaluate effects of NWRMs at the local scale (e.g. river reach, or mesohabitat scale; Liu et al., 2017; Xie et al., 2015), that are relevant for river management. The effectiveness of a single NWRM can be assessed locally with experimental studies, monitoring, numerical modeling or remote sensing, but upscaling possibilities are limited (Golden and Hoghooghi, 2018; Zhang and Chui, 2019). Therefore, tools that allow for the diagnosis of their effects at multiple spatial scales can greatly support the planning of NWRMs.

Fine bed material can be distinguished from the natural substrate because much of it is commonly transported and deposited selectively (Lisle and Hilton, 1999). Excessive fine bed material deposition has become an increasing concern for river systems around the world for its negative effects on the morphological, physicochemical, and biological status of water bodies (Doretto et al., 2018; Naden et al., 2016; Wohl et al., 2015). Fine bed material deposits (FBMDs) alter the river morphology by reducing the heterogeneity of mesohabitat structures (pool-riffle, plain bed) (Hauer, 2015). At smaller spatial scales, FBMD can clog the hyporheic zone matrix, leading to accumulation of toxic substances and to dissolved oxygen depletion (Geist and Auerswald, 2007; Mueller et al., 2013; Scheder et al., 2015). Thus, FBMD also can potentially impair fish spawning (Sternecker and Geist, 2010), and reduce macroinvertebrate abundances (Larsen et al., 2009; Leitner et al., 2015). Generally, fine sediment inputs and land use change are recognized to be a dominant factor shaping biological communities of fish (Mueller et al., 2020) and macroinvertebrate (Doretto et al., 2018). Finally, FBMD constituted of sand and fine gravel (modal diameter 1 – 10 mm, the focus of this study) are particularly affecting the stream bed stability. Such size classes require the lowest critical water flow velocity to initiate their motion compared to coarser and finer size classes (Hjülström, 1935; Strayer, 2008). Unstable streambeds are also recognized to be detrimental for the biotic compartment (Fuller and Death, 2018; Leitner et al., 2015). FBMDs are governed by a variety of catchment-, reach-, and micro-scale factors (Bracken et al., 2015; Czuba et al., 2017; Hancock et al., 2017; Wohl et al., 2015). Despite clear evidence of the importance of soil erosion and fine sediment inputs for stream health (Mueller et al., 2020), a clear link between FBMD and their controlling processes is still lacking (Anlauf and Moffitt, 2010; Sutherland et al., 2010).

FBMDs can be reduced with different NWRM strategies. For example, hydromorphological improvements (HMI) are structural modifications of the river morphology for increasing habitat heterogeneity (Bisson et al., 1992). Such modifications include removal of bank fixation, addition of wooden debris, installation of flow deflectors, or cross sectional modifications such as re-meandering or floodplain reconstruction (Haase et al., 2013). Cross-sectional modifications can support the achievement of stable conditions for net sediment transport and prevent sediment deposition without targeting sediment fluxes (Flödl and Hauer, 2019). Alternatively, vegetated filter strips are buffers of dense vegetation located at the edge of agricultural fields and along river banks with the purpose of trapping sediments (Daniels and Gilliam, 1996; Magette et al., 1989). Moreover, sedimentation ponds can be implemented in agricultural and forested areas to store excess runoff water and to trap sediments (Mekonnen et al., 2015; Verstraeten and Poesen, 2002).

Given the challenges and the knowledge gaps highlighted above, the objectives of this paper were to: a) apply a MC for identifying the controls for FBMDs; b) propagate the effects of selected NWRMs on the catchment hydrological response and sediment generation, and on reach hydraulics; and c) diagnose the effectiveness of NWRMs to mitigate FBDMs at different spatial scales (catchment and reach scales). The assessment was undertaken for the Aist catchment (Austria).

## 2. Methods

### 2.1. The Aist catchment

The Aist catchment (650 $km^2$) is located in the northeastern part of the state of Upper Austria (Fig. 1a). The climate is temperate with an average annual temperature of 7.1 ºC and an average annual precipitation of 835 mm (HDLO, 2017). The Aist River forms after the confluence of two main tributaries, the Feldaist and the Waldaist, draining respectively an agriculture/pasture dominated landscape (47 %, main crops: winter wheat, winter barley, corn silage) and a pasture/forest dominated landscape (49 %), with a small share of land use by settlements (4 %). The catchment is part of the Bohemian Massif (Ofenböck et al., 2004), with granite and gneiss bedrock (GBA, 2019), and sandy loam to silty loam haplic cambisols (Hengl et al., 2017). Rivers in the Aist catchment have a mainly "plane bed" morphology with cobbles and sands as dominating substrates (Leitner et al., 2015), while steep sections are classified as "cascade type" with a boulder substrate (Hauer, 2015). Both Feldaist and Waldaist are affected by FBMDs in the form of sand to fine gravel (modal diameters: 1 – 10 mm; Fig. 1b).

### 2.2. Modeling cascade

The MC used in this study is composed of a sequence of models structured in a way that the outputs from the coarser spatial scale model can be used as inputs to finer scale models (Kiesel et al., 2013). A complete description of the MC, the models setups and performances is provided in Baldan et al. (2020). Here a summary of the main components of the MC is presented (illustration in Fig. S1):

(i) the ecohydrological Soil and Water Assessment Tool (SWAT 2012 v670, Arnold et al., 2012) simulates discharge and sediment generation and transport at a daily time step and at the catchment scale;

(ii) the hydrodynamic numerical 1D-model hydraulic Engineering Centre – River Analysis System (HEC-RAS v5.0.5; Brunner, 2002) simulates hydraulics at the river reach scale;

(iii) an ensemble of Random Forests (RFs, R package 'caret'; Kuhn, 2008), using SWAT and HEC-RAS outputs to classify the FBMD risk at the mesohabitat scale ($10^2$ $m^2$).

The semi-distributed, process-based model SWAT discretizes the landscape in subcatchments and further in hydrological response units (HRUs, unique combinations of land use, soil type, and slope, see Table S1 in the supplementary material for the input data used). The model was first calibrated at five gauging stations for daily discharge. Then a second calibration for monthly discharge and sediment loads was performed. As continuously recorded sediment data for the Aist catchment are not available, sediment loads for calibration were obtained using bi-weekly total suspended solids data and flow-constituents regressions for the five gauging stations (Baldan et al., 2020). The model was calibrated for the years 2002 – 2010 and validated for the years 2011 – 2016. Calibration and validation were performed at the daily time step for flow and at the monthly time step for sediment. As measured by the Kling-Gupta efficiency (KGE, Knoben et al., 2019), the model performed well for both calibration (KGE = 0.70 ± 0.12 for flow and 0.67 ± 0.21 for sediments, standard deviation represents variability across flow and sediment gauges, respectively), and validation (KGE = 0.78 ± 0.10 for flow and 0.63 ± 0.17 for sediment). The calibrated SWAT model was used to simulate daily discharge and monthly sediment loads at the outlet of 103 subcatchments (mean area 6 $km^2$, mean reach length 5.5 km; Fig. 1d). High, medium, and low flow discharges (90th, 50th and 10th percentiles) were calculated from daily hydrographs modeled in SWAT and used to perform static flow profiles in the HEC-RAS model.

Hydraulics calculations were carried out for the Feldaist and tributaries (n = 10 reaches, length between 3.4 km and 32.6 km), the Waldaist and tributaries (n = 7; length between 4.5 km and 61.0 km), and for the Aist and tributaries (n = 3; length between 7.3 km and 21.8 km) for a total of 20 reaches and 11000 cross sections (Fig 1d). Water surface elevation for model calibration was measured in the Feldaist system at three representative reaches, ten cross sections each (250 m sections), during low-flow conditions. The difference between the measured and modeled water surface elevation was always smaller than 3 cm (considered acceptable according to Bolla Pittaluga et al., 2014; Miori et al., 2006). Field measurements of water surface elevation were not possible during high flow conditions. Here, the calibrated roughness values from a flood hazard analysis study for the Aist system were applied (Hauer et al., 2015).Outputs from both SWAT and HEC-RAS were processed and used as inputs (hereafter "predictors") to the RFs ensemble. Starting from the SWAT-modeled daily hydrographs, Indicators of Hydrological Alteration (IHAs) describing flow magnitude, frequency, duration and rate of change were calculated for each subcatchment (R package 'EflowStats', Olden and Poff, 2003) and used as predictors (Tables 1, S3). Predictors describing the average and peak (50th and 90th percentiles) lateral and upstream sediment input into streams (units: t $km^{-1}$ $y^{-1}$) were calculated from the SWAT model sediment outputs (Tables 1, S3). Predictors were generated also out of HEC-RAS cross-section outputs, including flow velocity (m $s^{-1}$), flow depth (m), and shear stress (Pa, Tables 1, S3). All predictors were rasterized along the channel centerlines and resampled at a resolution of 50 m. The SWAT simulations to generate the predictors covered the years 2002 – 2013.

The RFs ensemble (Breiman, 2001) uses the predictors listed in Table 1 to estimate the FBMD risk class at the mesohabitat scale (length scale = 50 m). The data used to train the RF models composition (collected between December 2013 and July 2014) consist of the classification of the river bed into five risk classes, ranging from 0 = natural status (low risk) to 3+ = FBMD unstable even during low flows (high risk; Hauer, 2015; Table 2, Fig. 1b, S2, S3). The model compositions in the ensemble were calibrated on different random splits of the input dataset (70 % for training, 30 % for testing), for a total number of 1000 models (Rokach, 2010). The use of ensembles ensures the stability of both the classification and the variable importance measure (Calle and Urrea, 2011; Millard and Richardson, 2015). The performance of the model compositions was always acceptable (Accuracy 0.72 ± 0.02, Kappa = 0.61 ± 0.02; standard deviation represents the variability across individual models; Allouche et al., 2006). For each model composition, the class-specific predictors importance was measured as the drop in accuracy when a predictor is randomly permuted (Strobl et al., 2009). Partial dependence plots (Friedman, 2001) were used to relate the changes in the likelihood of each sand accumulation risk class when a single predictor is varied. Empirical partial dependence plots were fitted with two different functions (monotonous log-logicstic and optimum Brain-Cousen, R package 'drc', Ritz et al., 2015). The best fitting function, selected based on Akaike's Information Criterion, provides information on the type of the response: monotonous (positive/negative slope) or optimum (global maximum/minimum).

### 2.3. Scenario development and SWAT sediment control measures

Five different sets of sediment control measures were implemented in SWAT (Table 3), including three individual measures set: (i) hydromorphological improvements (HMI), (ii) vegetated filter strips (VFS), and (iii) sediment retention ponds (PND) with two different volumes design. The last measures set was a combination of HMI and VFS.

For each set of measures, two strategies for implementation were used: i) a catchment-wide, uniform distribution of the measures in each HRU/subbasin/reach (henceafter: “catchment strategy”); and ii) a targeted implementation of each measure in specific HRUs/subbasins/reaches (henceafter “local strategy”). Criteria used for the local implementation were based on the current stream hydromorphological status and on the geographic position of hotspots of sediment generation modeled with SWAT.

Scenarios were defined as a SWAT model setup with a unique combination of a measure type and implementation strategy (Table 3). Scenarios were implemented by executing the following steps: (i) modifying SWAT input files according to a scenario definition; (ii) running SWAT model for the time interval 2002 – 2013; (iii) passing the resulting modeled outputs along into the next part of the modeling cascade. The results of the ten scenarios were then compared to the baseline scenario. The baseline scenario was implemented by running the models for 2002 – 2013 without any implemented measure.

### 2.3.1. Hydromorphological improvements

For the catchment strategy, HMI were implemented for each SWAT reach that was modeled with HEC-RAS. For the local strategy, only the reaches that are classified as having a moderate to very bad status in the hydromorphological quality elements of the Water Framework Directive were selected for the improvement (Fig. 1d, data source: Government of Upper Austria).

The implementation of HMI was done at the subcatchment level in SWAT by increasing the Manning's channel roughness up to a maximum of +0.05 (parameter CH_N2; Table 4, Aldridge and Garrett, 1973; Arcement Jr. and Schneider, 1984; Cowan, 1956). A reduction factor corresponding to the fraction of the reach where HMI are needed (Fig. 1d) was applied to the maximum increase.

#### 2.3.2. Vegetated filter strips

For the catchment strategy, VFS were implemented in every HRU, including forest-dominated HRUs. For the local implementation strategy, all HRUs from the 17 subcatchments responsible for generating 88 % of the total catchment sediment yield were selected for VFS (Fig. 1d). Such an high number of subcatchments was considered to be acceptable, as VFS are relatively low-cost, and are employed to target diffused pollution (Liu et al., 2019).

The VFS impact on sediment transport was assessed with the semi-empirical routine implemented in SWAT that calculates sediment loads reduction as a function of the size of the VFS and flow channelization (White and Arnold, 2009). The parameters FILTER_CON, FILTER_CH, and FILTER_RATIO were adjusted to account for the impact of landscape heterogeneities on sediment trapping (Table 4; Verstraeten et al., 2006). The selected parametrization yields sediment trapping efficiencies of approximately 30 % at the HRU scale (Hösl et al., 2012; Zessner et al., 2019).

#### 2.3.3. Sediment retention ponds

For the catchment strategy, sediment retention ponds were implemented in every subcatchment. For the local implementation strategy, eight subcatchments responsible for 58 % of the sediment yield in the catchment were selected for ponds implementation (Fig. 1d). As sediment retention ponds are more expensive to build than VFS and are designed to retain sediments with a higher efficiency (Ockenden et al., 2012; Rickson, 2014), they were located in the few subcatchments with high sediment yields.

Sediment retention ponds in SWAT are located off-stream, receiving a fraction of the subcatchment runoff and releasing it with defined outflow rules (Jalowska and Yuan, 2019). Pond properties are lumped at the subcatchment scale; thus an "equivalent pond" needs to be defined for each subcatchment (Neitsch et al., 2011). The physical dimensions of the pond impact the water and sediment routing through ponds.

Two approaches were used to define the required "equivalent" volumes. The first approach stores the surface runoff resulting from a 10 year return period rainfall (Huffman et al., 2011). The SWAT-derived, subcatchment-specific, rainfall-runoff coefficients were analyzed together with precipitation data to assess the volume of the pond to store the calculated water yield, which resulted in a volume per unit subcatchment area of 330 ± 80 $m^3$ $ha^{-1}$ (mean ± standard deviation across subcatchments; hence after "PND300" measures set). The second approach balances the sediment trapping efficiency (STE) and pond volume requirements by considering the mean annual 90$^{th}$ precipitation percentile as a design parameter (Papa et al., 1999; Urbonas, 2000), resulting in pond volumes in the range of 45 ± 5 $m^3$ $ha^{-1}$ (mean ± standard deviation across subcatchments; hence after "PND50" measures set). In both setups, 1984 – 2016 precipitation data were used. The two sets of pond volumes were assigned to the PND_EVOL parameter in the SWAT model. The other parameters defining pond volumes and area were calculated accordingly (Table 4).

As the SWAT routine for pond outflow maximizes water storage for agricultural use (Jalowska and Yuan, 2019), new outflow rules were defined to allow for water storage during high flows and a gradual release during low flows. The new criteria were implemented by modifying the outflow rules in the SWAT 2012 v670 source code as follows:

$$IF\ V \leq PVOL,\ THEN \quad OUT = 0 \tag{1.1}$$

$$IF\ PVOL < V \leq EVOL,\ THEN \quad OUT = (V - PVOL)/ndtarg \tag{1.2}$$

$$IF\ V > EVOL,\ THEN \quad OUT = V - EVOL \tag{1.3}$$

Where V is the volume of the pond, *ndtarg* is the number of days needed to empty the pond, PVOL is the principal volume of the pond, and EVOL the volume at full capacity. Therefore, when the pond is filled above the full capacity volume, no water retention occurs (i.e. failure). Water storage and slow release occur only when the volume is below the full capacity volume. The *ndtarg* parameter was set to 5 to simulate a slow release of the stored water. The permeability of the bottom of the pond (PND_K) was set to 1 mm $h^{-1}$ (Habets et al., 2018).

Sediments are routed in SWAT ponds with a simplified settling model based on equilibrium sediment concentration (Jalowska and Yuan, 2019). A one-at-a-time sensitivity analysis was performed to check the sensitivity of each parameter with the new code (Fig. S4 – S7). The sensitivity analysis revealed the STE to have a low sensitivity to changes in pond volume (Fig. S5) Thus, sediment parameters (PND_NSED and PND_D50, Table 4) were adjusted to obtain an STE value of approximately 70 % (i.e. within the ranges reported by Mekonnen et al., 2015; Verstraeten and Poesen, 2002, 2001).

## 2.4 Assessment of the effectiveness in reducing sediment accumulations at different spatial scales

The impact of NWRMs on the SWAT and HEC-RAS predictors (Table 1) was diagnosed at the catchment scale. The relative percent change of predictors (RCp) compared to the baseline was calculated for each predictor. The modal value of the RCp indices for each scenario (namely: mRCp) was used as an assessment of the effectiveness of measures.

The impact of measures on the simulated FBMDs risk was diagnosed at two different spatial extents: (i) for the whole catchment and (ii) for selected reaches (Fig. 1c; Tables 3, 5), corresponding to sites of high ecological value (reaches A, B, C; Baldan et al., 2020) or to sites where the FBMDs issue is severe (reaches D and E). First, the spatial coverage of each FBMD risk class was computed in the selected spatial extent. Second, the relative percent change and the absolute change in the spatial coverage of each FBMD class (RCe and ACe) was computed for each model in the RF ensemble. The modal values of RCe and ACe (namely mRCe and mACe) calculated across all ensemble members for each scenario and diagnostic extent were also used to assess the effectiveness.

The uncertainty in mRCp, mRCe, and mACe estimations was assessed using the direction of the change index (DC):

$$DC = 2\,|F_{>0} - 0.5| \qquad (2)$$

where $F_{>0}$ is the fraction of RCp, RCe, or ACe that is positive. The DC ranges between 0 and 1, with DC = 1 when all the RCp, RCe, or ACe indices are positive or negative (clear direction of the change) and DC= 0 when the distribution of RCp, RCe, or ACe is centered on 0 (unclear direction of the change)

# 3. Results

## 3.1 Important predictors controlling fine bed material deposits

The assessment of the RFs ensemble's predictors importance shows that different predictors control the allocation of FBDMs to different risk classes (Fig. 2a). High flow shear stresses dominate for risk classes 0, 1, and 3+. Low flow velocity and flow depth are relevant for class 3+. For risk classes 0 and 1, peak upstream sediment inputs are also important. Risk classes 2 and 3 do not show a dominant predictor.

The fitting of empirical partial dependence plots shows that the relationship between the risk class likelihood of occurrence and predictors is different depending on the risk classes (Fig. 2b). Risk class 0 (natural conditions) is more likely to occur for sites where shear stresses during high flow are high and when the upstream peak sediment load is low. Class 3 is more likely to occur when the high flow depth is high and the high flow shear stresses and low flow velocity are low, while the dependency from the high flow shear stresses and high flow frequency has an optimum. Class 3+ is more likely to occur when high flow shear stresses and flow depth are low.

## 3.2 Effectiveness of measures on hydrology, hydraulics, sediment loads at the catchment scale

The NWRMs implemented in the MC lead to changes in hydrology, hydraulics, and sediment predictors (measured by mRCp, Fig 3, S10-S12). Reductions in sediment loads are similar for both sediment retention ponds and VFS (- 40.4 % and -28.2 % respectively for the catchment strategy and – 8.9 % and – 9.3 % respectively for the local strategy). In addition, ponds affect the hydrological regime of the catchment, with P300 being the most effective at reducing magnitude (-11.5 % for catchment strategy, -1.6 % for local strategy), duration (-12.2 % for catchment strategy, -1.4 % for local strategy), and frequency (-20.4 % for catchment strategy, -2.3 % for local strategy) of high flows. PND50 are more effective in increasing magnitude of low flows for the catchment strategy (+15.7 %). HMI provide the greatest increase in low flow magnitude for the local strategy (+2.8 %). PND300 are more effective in reducing high flow shear stresses (-3.2 % for catchment strategy, -0.6 % for local strategy) and flow depth (-2.7 % for catchment strategy, -0.5 % for local strategy).

### 3.3 Effectiveness of measures on fine bed material deposits at the reach scale

The baseline scenario status of diagnostic reaches is diverse (Table 6, Fig. S8), with reach A having a high aerial share of sites free from FBMDs (risk class 0, 36 %), reaches B and C being occupied by intermediate risk classes 1 and 2, reach D being dominated by risk class 3+ (37.4 %), and E being dominated by risk class 3 (67 %).

The effectiveness of the measures implemented changes based on the diagnostic reaches (Fig. 5). In reach A, PND300 reduce the extent of risk class 0 and 3+ (mACe = -2 and +2.2 km, respectively) for the catchment strategy, while the local strategy shows no clear preference ranking for measure types and an overall low effectiveness (mACe always < 0.2 km m). Reach B has a low sensitivity to the implementation of measures (mACe < 0.2 km), with HMI and ponds being able to reduce the extent of risk class 2 and increase the extent of risk class 1 both for the local and for the catchment strategy. In reach C, both ponds and VFS can reduce the extent of risk class 3 (mACe = -0.5 km) and increase the extent of risk class 1 (mACe up to +2.1 km), with local and catchment strategies showing similar effectiveness. In reach D, ponds are effective in reducing the extent of classes 3+ and 3 (mACe = -1 km) and increasing the extent of risk classes 0 and 1 (mACe = +0.7 and +1.5 km for PND50, respectively). Low effectiveness is observed for the local strategy (mACe always < 0.2 km) when compared to the catchment strategy (mACe up to 1.5 km). In reach E, sediment ponds are more effective in reducing the extent of risk classes 3 and 3+ (mACe = -0.4 km and around -2.5 km respectively), while in the local strategy only PND50 are highly effective in reducing risk class 3 (mACe = -2 km).

**3.4 Effectiveness of measures on fine bed material deposits at the catchment scale**

For the baseline scenario, the RF models predict sites with risk classes 0 and 3+ to occupy a small share of the reach at the catchment scale (respectively 11.6 % and 5.9 %; Table 6, Fig. S9).

The effects of the measures implemented in the MC on FBMDs at the catchment scale are diverse and depend on the implementation strategy. Measures that are most effective (mRCe, Fig. 4) in reducing the spatial extent of class 3+ are HMI (-10.3 %) and ponds (-18.5 %). However, sedimentation ponds are not effective for class 3+ when located strategically on sediment generation hotspots (DC = 0.32 for PND300, while PND50 do not reduce the extent). Both PND300 and PND50 are effective at reducing the extent of risk class 3 for the catchment strategy (-28.9 % and -18.1 %, respectively) and the local strategy (-5.8 % and -8.3 %, respectively). PND300 and PND50 are also effective at reducing the extent of risk class 2 (-59.1 % and -56.4 %, respectively) and increasing the extent of risk class 1 (+97.4 % and +78.6 % respectively) for the catchment strategy, but the changes are highly uncertain when the local strategy is implemented (DC <0.34 for class 2 and < 0.47 for class 0). When implemented locally, alone or in combination with HMI, VFS are more effective in reducing the extent of risk class 2 (- 13.5 % and -17 %, respectively) and increasing the extent of risk class 1 (+15.5 % and + 13.5 % respectively). For the catchment strategy, PND300 is the only measure that increases the extent of risk class 0, while for the local strategy, VFS alone or in combination with HMI are effective (+4.7 % and + 3 % respectively).

# 4. Discussion

## 4.1 Important predictors controlling fine bed material deposits

The ensemble of RF models allowed to rank the importance of the predictors in defining the FBMDs risk classification. Consistent with the literature, the local hydraulics were identified as the most important predictor controlling the in-stream sediment depositions (Van Rijn, 1993). Shear stresses that occur during high flow (the most important predictor) were computed for the $10^{th}$ percentile of discharge, which is often used as a proxy for bank full discharge that causes the maximum transport of sediments to take place (Doyle et al., 2007). Thus, sites with low shear stresses are more likely to be prone to sediment accumulations. The negative-sloped dependence of risk class 3+ from the flow depth might be an indication that the local morphology of sites with low shear stress (risk class 3+) have been already altered (Hauer, 2015). In sites belonging to the high-risk class, the natural substrate is filled up with sediments. The channel dynamics prevail with FBMDs not being linked to the upstream channel supply of sediments in the catchment (Czuba et al., 2017). On the contrary, the amount of upstream sediment loads is an important predictor for low-risk classes to occur (i.e. low upstream sediment loads cause lower risk classes downstream), suggesting sites with low FBMDs risk are supply-limited (Naden et al., 2016). Thus, achieving a supply-limited state along a reach can be a strategy for increasing the spatial extent that is free from FBMDs. However, this strategy might not be sufficient for sites where fine bed material deposits are stable due to the reduced transport capacity (Sutherland et al., 2010). Understanding the controlling factors for FBMDs is thus beneficial for a “process-based” diagnosis for determining the critical sites (i.e. the sites that are at high risk for fines deposits) and for identifying the management options that can be implemented in a catchment (Collins et al., 2011; Naden et al., 2016; Wohl et al., 2015).

The approach used in this work allows to predict FBMD occurrence risk at the meso-habitat scale, neglecting other factors that are potentially driving spatial heterogeneities at finer scales, such as the macrophytes cover, reach sinuosity, and spatial autocorrelation (Braun et al., 2012). The developed relationships should be updated to include such factors, if the MC was intended to be used at finer spatial scales. Downscaling can benefit greatly from including finer scale riverbed mapping and 2D hydrodynamic models in the MC (e.g. Kiesel et al., 2013).

**4.2 Effectiveness of measures on hydrology, hydraulics, and sediment loads at the catchment scale**

The relative changes in predictors after the implementation of NWRMs were generally higher for hydrology and sediment loads than for the hydraulics. This can be explained by the different responses of reach morphologies and structures (e.g. pools, riffles, fast-runs) to the changes in flow percentiles (Hauer et al., 2013). This variability in the response is lost when changes are averaged over larger spatial extents (as noted in a MC by Guse et al., 2015).

The HMIs lead to an increase in low flow velocity, high flow shear stresses and water depth. This measure was implemented in SWAT with a simplistic approach, with roughness increments, failing to fully capture two important aspects. First, cross-sectional modifications can be designed to target specific flow magnitudes (e.g. flows above bankfull; Flödl and Hauer, 2019). Second, micro-topographical roughness increases local turbulences and consequently might increase the FBMDs transport capacity (Hauer et al., 2019). On the contrary, those sections of the river with reduced flow velocities might have increased sediment deposition. A higher detail level can be implemented by modifying the HEC-RAS geometries. This approach might be labor-intensive for a catchment-scale analysis but is valuable when the analysis is restricted to smaller spatial extents (e.g. following the approach in Hester and Doyle, 2008).

Impacts of ponds on the water fluxes include the reduction of high flow magnitude and the reduction of high flow shear stresses. PND300 have stronger effects than PND50, while effects on sediment flows are comparable. Sedimentation ponds can be filled with sediment in a short time after implementation (Verstraeten and Poesen, 2002). Thus, it can be expected that the effects of sediment ponds on hydrology and hydraulics will decrease quickly over time as the water storage volume decreases. When the hydrological component is neglected, sediment ponds and VFS have similar impacts on sediment trapping and a preference ranking should be established based on other criteria (e.g. land availability or potential losses in water yield, e.g. Strauch et al., 2013).

Assessing the effectiveness of NWRMs on different environmental components through a MC can support decision making as changes in the hydrological, hydraulic, and sediment predictors are made explicit. This allows for a comparison of possible management alternatives based on different environmental criteria (Liu et al., 2017; Rickson, 2014).

### 4.3 Effectiveness of measures on fine bed material deposits at the reach scale

Our analysis revealed that both the effectiveness and the preference ranking of measures change when the results are examined in different diagnostic reaches. In some cases, results are contradictory when compared to the catchment-scale effectiveness. Such variability can be explained by the fact that diagnostics at the catchment scale hinders the diversity in the local effects by averaging the changes over a large spatial extent.

The diversity in the responses found among the reaches can be explained by the local geomorphic conditions. Reaches A and B in the Waldaist are located in a semi-natural state (e.g. generally good hydromorphological status, Table 5), with reduced sediment loads entering the stream. Reach C in the Waldaist is also in a good hydromorphological status but the sediment pressure is higher. On the other hand, reaches D and E located in the Waldaist already have a poor hydromorphological status and a higher sediment pressure.

Sediment retention ponds are among the measures that show the highest variability in the response. This can be explained by the interplay between the selected timing for outflow regulation (five days) and the time of concentration of the subcatchment located upstream of the diagnostic reaches.

The lack of a clear pattern in the local responses to NWRMs implementation highlights the importance of considering different local conditions when simulating the impacts of measures to reduce FBMDs in a river network (Guse et al., 2015). To account for the contrasting results, they could be used the first step of a multi-criteria decision analysis (Kiker et al., 2005) with the aim of defining a “compromise” measure set where criteria (risk class to be improved) and spatial weights (high priority reaches) can be defined. Furthermore, the optimal spatial arrangement and intensity of measures that maximize the reduction of the extent of high-risk classes under the constraint of minimizing the loss of the extent of low-risk classes could be further explored with optimization algorithms (Yang and Best, 2015; Zhang and Chui, 2018).

### 4.4 Effectiveness of measures on fine bed material deposits at the catchment scale

Existing approaches to study FBMD make use of empirical linkages that do not consider the processes involved in sediment transport and deposition (Anlauf and Moffitt, 2010; Bilotta et al., 2012; Naden et al., 2016; Sutherland et al., 2010). The use of MCs can help filling this gap; the assessment of the response of FBMDs to predictors (subsection 4.1) explains the observed NWRMs effectiveness on FBMDs by considering the intermediate cascading effects on different environmental compartments (subsection 4.2). A prioritization of measures can be obtained based on the target risk class.

For instance, the spatial extent covered by sites in risk classes 0 (natural conditions) and 1 (low risk) can be increased by implementing specific NWRMs targeting sediment fluxes before they enter in the stream (Wharton et al., 2017). On the other hand, the effects of sediment ponds on the hydrology can reduce the in-stream sediment transport capacity by reducing the peak flows and therefore lead to increased sediment deposition downstream.

Moreover, the spatial extent covered by sites in risk class 3+ can be reduced by implementing specific NWRMs able to improve the cross-sectional transport capacity and the in-stream bed quality (Auerswald and Geist, 2018a). The additional implementation of VFS in combination with HMI shows the greatest reduction in the spatial extent, while VFS alone do not provide significant changes. Sediment control measures alone are not sufficient to reduce the extent of the risk class 3+ sites, as the substrate is saturated with sediment and the remobilization of sediment stored in the upstream channel might occur (Naden et al., 2016).Our analysis reveals that a combination of measures targeting different aspects (sediment generation, hillslope sediment transport, in-stream transport capacity) potentially has the highest effectiveness to reduce all the risk classes (Knott et al., 2019). Using synergies between in-stream and catchment sediment transport measures to tackle FBMD have been recommended (Auerswald and Geist, 2018b), but is still rarely considered in measures planning (Auerswald et al., 2019).

The interactions between different NWRMs types can result in nonlinear cumulative effectiveness (e.g. the interaction between HMI and VFS effects is linear for class 3+ and nonlinear for risk class 0). Optimization algorithms can be useful for testing different spatial arrangements and parametrizations (Chen et al., 2015; Yang and Best, 2015; Zhang and Chui, 2018).

### 4.5 Implications for management in the Aist catchment

Our results suggest an effective plan for sediment management in the Aist catchment should involve the use of sets of measures tackling different aspects. Significant improvements for the risk classes 0 (areal increase) and 3+ (areal increase) at the catchment scale can be achieved already when VFS are located on few sediment generation hotspots (17 % of the catchment area), and when HMI are located in reaches with a poor hydromorphological status according to the WFD (approximately 10 % of the modeled river network). Greater improvements for classes 1 (up to 40 % areal increase) and class 2 (up to 25 % areal decrease) can be achieved by increasing the implementation magnitude of HMI and VFS.

Implementing NWRMs in the Aist catchment can also lead to synergistic benefits for species conservation and climate change mitigation. In fact, the reduction of river sections occupied by high FBMD risk classes has beneficial ecological consequences not only for the conservation of the highly endangered Freshwater Pearl Mussel (Baldan et al., 2020; Hauer, 2015), but also in general for the health of benthic communities (Leitner et al., 2015). Furthermore, changes in precipitation/runoff patterns have been recorded in the area (Hauer et al., 2015). Thus, the implementation of structures increasing the mean water residence time in the catchment can be also an effective mitigation strategy for climate change impacts in the area (Hauer et al., 2016, 2013).

## 5. Conclusion

In this paper, we assessed the potential of different NWRMs scenarios in mitigating FBDMs. This was done by propagating the effects of measures from the catchment scale to the meso-habitat scale with a sequence of models in a MC framework. General implications for FBMD management from this study are: (i) the identification of the controlling factors for FBMDs provides a rationale for the selection of the appropriate NWRMs to implement; (ii) FBMDs responses to measures that are implemented can be different when diagnosed at different spatial scales; (iii) combinations of NSWRMs have the potential to address multiple goals, e.g. reducing the spatial extent of sites severely impacted by FBMDs and increasing the spatial extent of sites in natural conditions.

Thus, planning of NWRMs should account for the effects of measures across multiple spatial scales, considering both catchment and local scales diagnostics when assessing the impacts on the reduction of sediment. A multi-scale assessment of the effectiveness of the NWRMs can be beneficial to unravel all the potential benefits and highlight tradeoffs hidden by analyses limited to only one specific spatial scale. A combination of measures has the potential to mitigate tradeoffs and provide sound solutions to FBMDs.

Finally, the presented work underlines the advantage of implementing nature based solutions to increase the multifunctionality of the river network. As discussed for the Aist catchment, NWRMs implemented for mitigating the FBMDs issue can also lead to synergies of soil conservation, species conservations, and climatic goals (Geist and Auerswald, 2019).

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FIGURES CAPTIONS

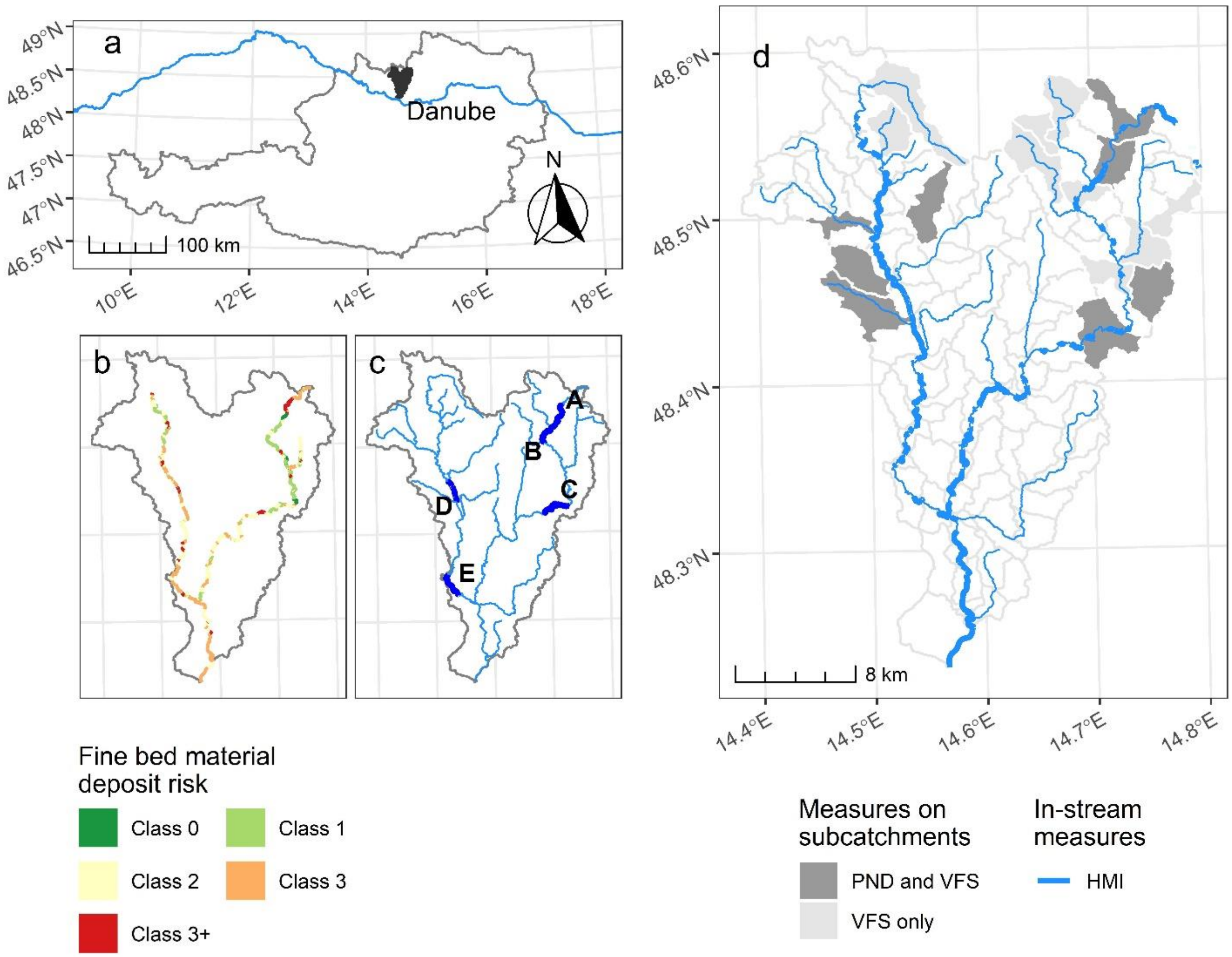



**Figure 1** – a) The position of the Aist catchment in Austria; b) Mapped fine bed material deposit risk class in the main channels, refer to Table 2 for an explanation of the classes; c) position of the diagnostic reaches, marked A - E; d) geographic position of implemented measures for the local strategy: sediment ponds (PND), vegetated filter strips (VFS), and hydromorphological improvements (HMI). Polygons represent SWAT subcatchments, the river network represents the Hec-RAS modeled reaches.

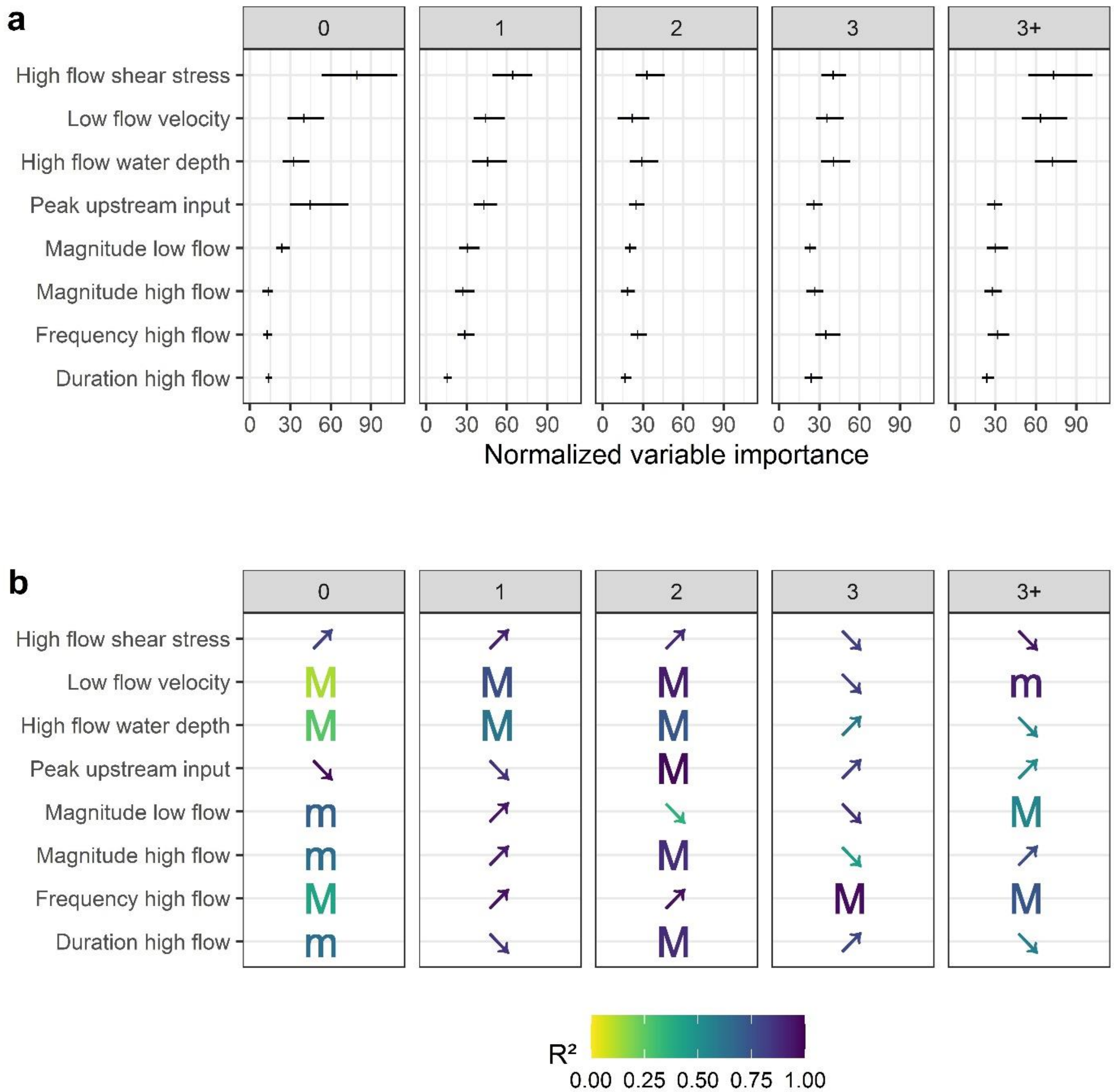
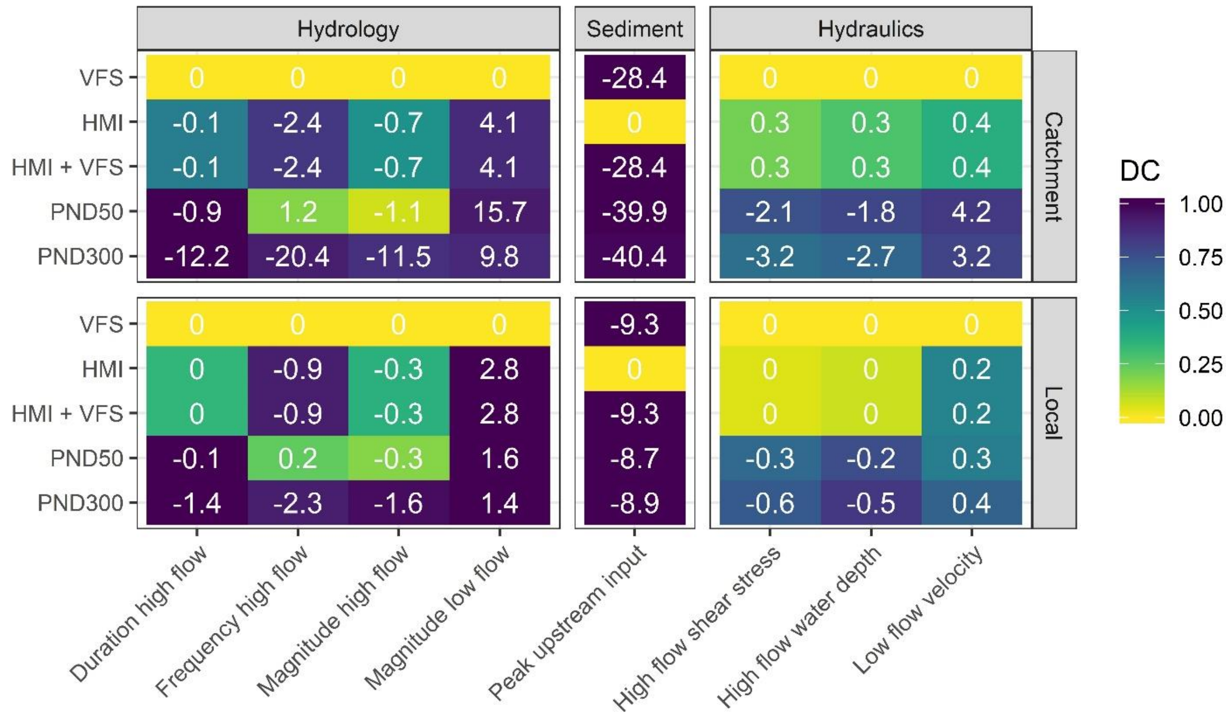



**Figure 2** – a) The importance of predictors from the Random Forest ensemble (horizontal axis) is displayed for every combination of predictors (vertical axis) and predicted fine bed material deposit risk class (horizontal grouping). The vertical ticks represent the mean of the variable importance metric,the horizontal lines represent the variation range; b) Partial dependence of the likelihood of each sand accumulation risk class from the predictors. Symbols syntetically represent the type of the response: M = maximum, m = minimum, upwards arrow (↗) = monotonous response with a positive slope; downwards

arrow (↘) = monotonous response with a negative slope. The color of the annotation shows the goodness of fit, as measured by $R^2$.




**<u>Figure 3</u>** –Effectiveness of measures on hydraulics, hydrology and sediment for both the catchment and local strategies (vertical axis grouping). Numbers represent the relative percent change for each scenario compared to the baseline scenario (mRCp%). The color scale represents the direction of change (DC) index. VFS = vegetated filter strips, HMI = hydromorphological improvements, PND = sediment ponds

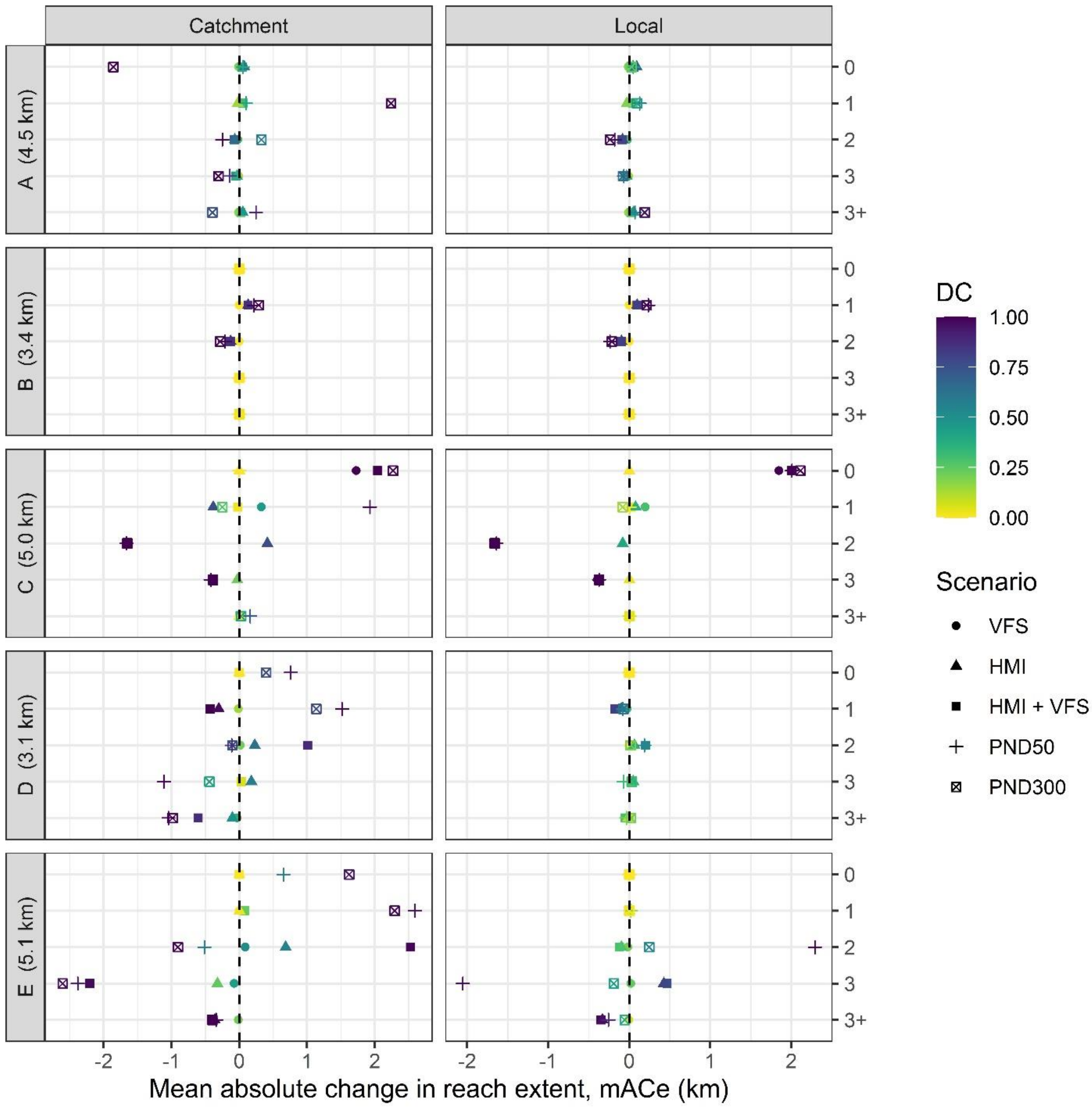



**Figure 4** – Effectiveness of measures for local diagnostic reaches represented as absolute changes in river length (mACe, x axis) occupied by each fine bed material deposit risk class (0 to 3+, y axis on the left), for the diagnosed reach (A-E, vertical grouping factor), for the two implementation strategies ("Catchment" and "Local", horizontal grouping), and for each accumulation risk class (0 to 3+, y axis on the right). Sets of measures are represented with symbols shape. The color scale of the symbols represents the value of the

direction of change (DC) index. VFS = Vegetated Filter strips, HMI = Hydromorphological improvements, PND = Sediment ponds

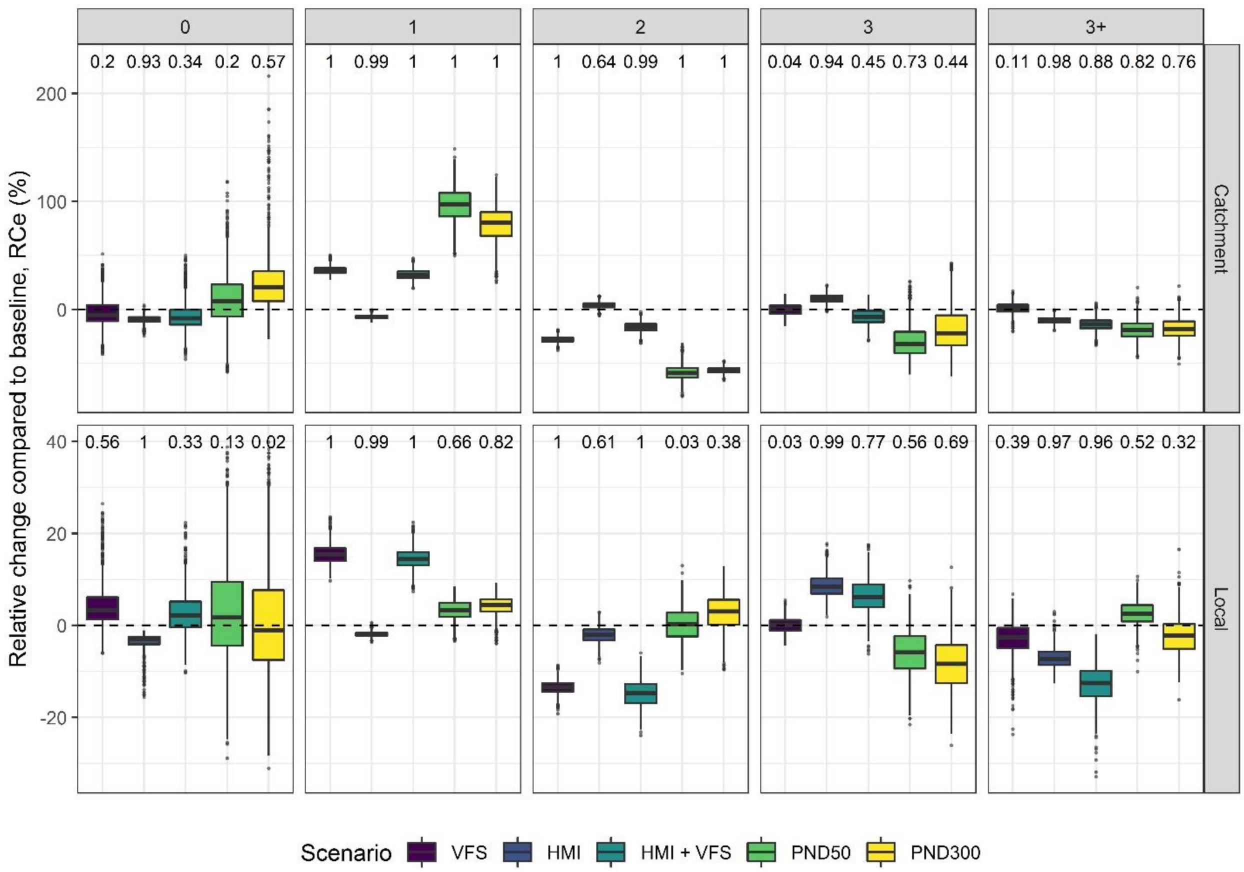



**Figure 5** – Effectiveness of measures for different fine bed material deposit risk class (0 to 3+) for both the catchment and the local implementation strategies. Effectiveness in the y axis is assessed as relative percent change in spatial coverage of each sand class compared to the baseline scenario (RCe). The annotation above each boxplot represents the DC index. Refer to Table 5 for the spatial extent covered by classes in the baseline scenario. VFS = vegetated filter strips, HMI = hydromorphological improvements, PND = sediment ponds

**Table 1** – SWAT and HEC-RAS outputs (predictors) used in the fine bed material deposit model (Random Forest ensemble).

| Category | Output | Short Name | Description | Units | Source |
|---|---|---|---|---|---|
| Sediment | Peak upstream input | LTs_up_90 | $90^{th}$ percentile of the SWAT sediment yield normalized by the drainage density and cumulated for upstream subcatchments. | $t\ km^{-1}\ month^{-1}$ | SWAT |
| Hydrology | Duration high flow | dh3 | Annual maxima of 7-day means of daily discharge | $m^3\ d^{-1}$ | SWAT |
| Hydrology | Frequency high flow | fh5 | Mean yearly number of events where the flow exceeds two times the median discharge | - | SWAT |
| Hydrology | Magnitude high flows | mh16 | Mean of the $10^{th}$ flow percentile from the flow duration curve divided by the median daily flow across all years | | SWAT |
| Hydrology | Magnitude low flows | ml15 | Median of the lowest annual daily flows divided by median annual daily flows averaged across all years | - | SWAT |
| Hydraulics | Low flow velocity | v_LF | Cross sectional average of flow velocity calculated with the $10^{th}$ discharge percentile | $m\ s^{-1}$ | HEC-RAS |
| Hydraulics | High flow shear stress | SS_HF | Cross sectional average of shear stresses, calculated with the $90^{th}$ discharge percentile | Pa | HEC-RAS |
| Hydraulics | Mean flow water depth | d_MF | Cross sectional average of flow depth; for median discharge | m | HEC-RAS |

**Table 2** – Fine bed material deposit risk class definition (adapted from Hauer, 2015).

| Risk class | Description |
|---|---|
| 0 | No alteration of the natural substrate |
| 1 | Little disturbance due to fine bed material deposit |
| 2 | Some habitat changes but main morphological features are kept |
| 3 | Mesohabitat is fully covered by fine bed material deposit |
| 3+ | Mesohabitat is fully covered by fine bed material deposit, that are mobile during low flow conditions |

**Table 3** – Summary of scenarios and diagnostics. A scenario is a unique combination of a measure type and an implementation strategy (catchment or local).

| | | | |
|---|---|---|---|
| Scenario | Measure | VFS | Vegetated filter strips |
| | | HMI | Hydromorphological improvements |
| | | HMI + VFS | Hydromorphological improvements and vegetated filter strips |
| | | PND300 | Sediment retention ponds, big volume |
| | | PND50 | Sediment retention ponds, small volume |
| | Strategy | Catchment | Measure is implemented in every HRU / Subcatchment / Reach |
| | | Local | Measure is implemented in specific HRUs / Subcatchments / Reaches |
| Diagnostic extent | | Catchment | Spatial extent analyzed is the whole catchment |
| | | Reach | Spatial extent analyzed is one out of the 5 diagnostic reaches (A - E) |

**Table 4** – SWAT parameters modified for measures implementation. Modifications are reported for the catchment implementation strategy, refer to the text for the reduction in geographic extent occupied by measures for the local implementation strategy. Refer to Neitsch et al. (2011) for a detailed parameter descriptions.

| Measure | Parameter (SWAT input file) | Parameter modified value | Units | Parameter description |
|---|---|---|---|---|
| Hydromorphological improvements (HMI) | CH_N2 (.rte) | Increased by 0.05 R [a] | s $m^{-0.33}$ | Manning's channel roughness coefficient |
| Vegetated filter strips (VFS) | FILTER_RATIO (.ops) | 40 | - | Ratio of field area to VFS area |
| | FILTER_CON (.ops) | 0.75 | - | Fraction of the HRU whose flow is concentrated |
| | FILTER_CH (.ops) | 0.90 | - | Fraction of the concentrated flow that is not treated |
| Sediment retention ponds (PND) | PND_FR (.pnd) | Calculated assuming the area in a buffer of 200 m buffer from reach is not draining in the ponds | - | Fraction of the subcatchment that drains in the pond |
| | PND_EVOL (.pnd) | Calculated (i) Based on 10 year precipitation return period; (ii) Based on the average 90th yearly precipitation percentile | $10^4$ $m^3$ | Volume of the pond at full capacity |
| | PND_ESA (.pnd) | Calculated based on PND_EVOL assuming a water depth of 1 m | $10^4$ $m^3$ | Area of the pond at full capacity |
| | PND_PVOL (.pnd) | 10% of PND_EVOL | $10^4$ $m^3$ | Minimum volume of the pond |
| | PND_PSA (.pnd) | calculated based on PND_PVOL assuming a water depth of 1 m | $10^4$ $m^3$ | Minimum area of the pond |
| | PND_NDTARG (.pnd) | 5 | days | Number of days needed to empty the pond |
| | PND_K (.pnd) | 1 | mm $h^{-1}$ | Permeability of the pond |
| | PND_NSED (.pnd) | 80 | mg $L^{-1}$ | Sediment equilibrium concentration |
| | PND_D50 (.pnd) | 10 | µm | Modal diameter of pond bottom sediment |

[a] R is a GIS-calculated reduction factor that accounts for the fraction of the length of the reach where the hydromorphological improvements are implemented. For the catchment strategy, R = 1 always.

**Table 5** – Characteristics of diagnosed reaches and fraction of upstream area/river network length that is occupied by measures for the local implementation strategy.

| Reach | Tributary | Upstream catchment area ($km^2$) | Upland slope [ % ] | Upland sediment production (t / ha y) [a] | Reach mean elevation (m) | Reach length (km) | Reach mean Slope (%) | Reach fraction for HYDRO |
|---|---|---|---|---|---|---|---|---|
| A | Waldaist | 20 | 10.7 | 0.17 | 887 | 4.499 | 1.57 | 0.22 |
| B | Waldaist | 45 | 11.1 | 0.14 | 892 | 3.448 | 1.14 | 0.30 |
| C | Waldaist | 139 | 17.6 | 0.32 | 840 | 5.065 | 0.67 | 0.18 |
| D | Feldaist | 186 | 13.9 | 0.37 | 691 | 3.070 | 0.36 | 0.39 |
| E | Feldaist | 250 | 15.1 | 0.36 | 663 | 5.061 | 0.55 | 0.50 |

[a] Upland sediment production calculated from SWAT .sub output for the time period 2002 - 2013

**Table 6** – Mean (standard deviation) fraction of the spatial extent occupied by fine bed material deposit risk classes for the baseline scenario. For the catchment calculation, a total river length of 280 km was considered. Refer to Table 4 for the absolute length of reaches A-E.

| **Fine bed material deposit risk class** | | **0** | **1** | **2** | **3** | **3+** |
|---|---|---|---|---|---|---|
| Catchment diagnostics | | 11.6 (3.5) | 26.6 (1.2) | 32.6 (1.4) | 23.1 (3.3) | 5.9 (0.5) |
| Reach diagnostics | A | 36 (1.6) | 31 (2.7) | 6.7 (1.2) | 6 (1.2) | 20 (2) |
| | B | 0 (0) | 82.3 (2.4) | 17.4 (2.5) | 0 (0) | 0.1 (0.5) |
| | C | 0 (0) | 60.5 (4) | 31.4 (3.9) | 8 (1.5) | 0 (0) |
| | D | 0 (0) | 18.2 (3.5) | 4.3 (1.3) | 39.9 (3.5) | 37.4 (3.9) |
| | E | 0 (0) | 0 (0) | 21.6 (2.8) | 68.7 (2.8) | 9.5 (0.8) |