Assessing the Return On Investment in Watershed Conservation

Best Practices Approach and Case Study for the Rio Camboriú PWS Program,

Santa Catarina, Brazil

The Nature Conservancy


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Best Practices Approach and Case Study for the Rio Camboriú PWS Program,

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Contents


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Executive Summary

Preserving and restoring water quality is a major concern for cities around the world. In most cities, urban population growth, coupled with degradation of municipal source watersheds, has increased drinking water treatment costs. One recent estimate suggests that in one-third of large cities, costs per unit of treated water have increased on average by roughly 50 percent over the last century because of land conversion and development in source watersheds.

Restoring source watersheds can reverse this trend, and may be a cost-effective approach for cities to reduce drinking water treatment costs while enhancing supply resiliency and protecting biodiversity among other co-benefits. Nevertheless, the potential to cost-effectively deliver key hydrologic services through watershed investments far exceeds the current extent of watershed conservation programs. Mobilizing the investments needed to realize this potential hinges in part on the business case for water users — that is, the competitiveness of watershed conservation programs with conventional engineering solutions.

Yet credible economic assessments of watershed conservation or restoration are almost entirely absent from the literature, leaving the business case for watershed conservation an important yet largely unanswered question. Worse still, those interested in evaluating the business case in their own geography lack the examples and tools to do so in a robust manner.

ROI ANALYTICAL FRAMEWORK

Remedying this shortcoming requires a rigorous analytical framework that combines ecosystem service production functions, benefit production functions, economic valuation and the comparison of scenarios with and without the interventions to allow the measurement of welfare changes caused by watershed interventions. We synthesize and apply such a framework (Figure ES-1) to a recently created payment for watershed ecosystem services (PWS) program in the Camboriú River watershed in Santa Catarina State, Brazil.

The Camboriú watershed — situated in Brazil's threatened and biodiversity-rich Atlantic Forest biome that has been reduced to 12 percent of its historic extent— is experiencing fine-scale land cover changes and high sediment loading. The main objective of the PWS program for its principal funder, the municipal water supply company EMASA, is to reduce concentrations of total suspended solids (TSS) at the municipal drinking water intake and associated water treatment costs and water losses.

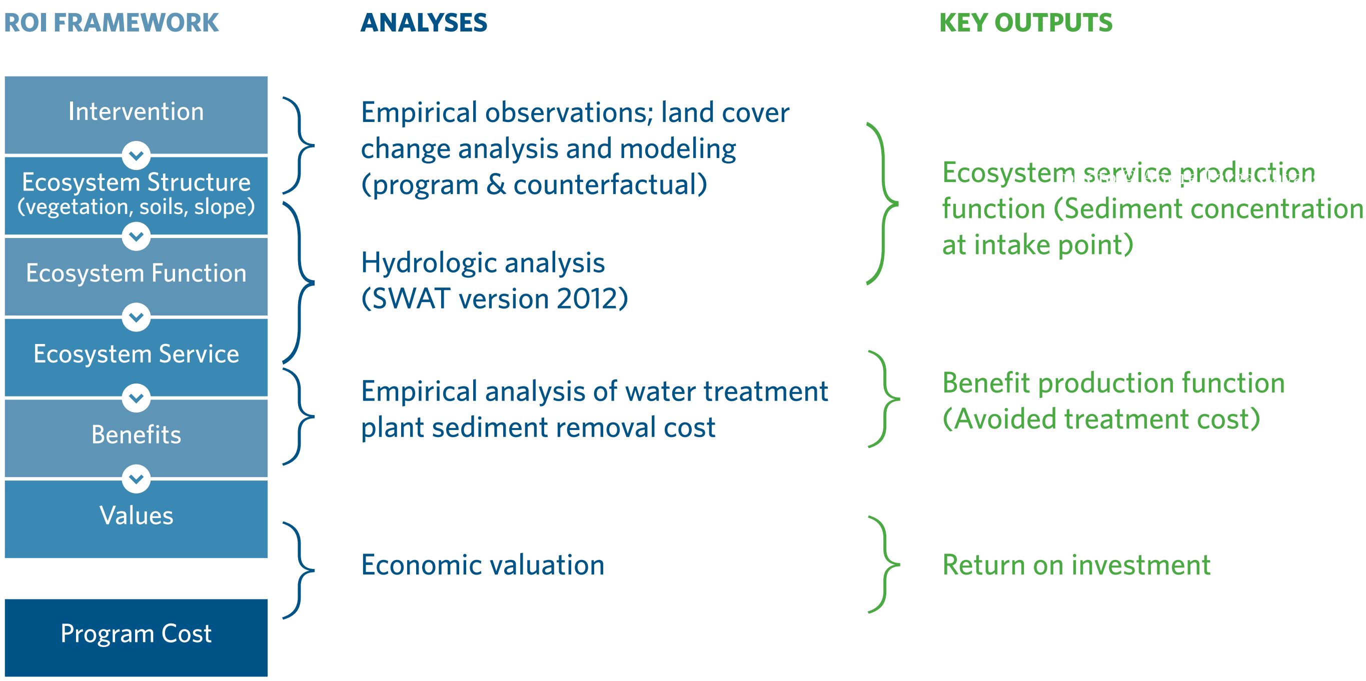

Figure ES-1: Analytical framework and associated analyses used to assess the return on investment of the Camboriú watershed conservation program for water treatment plant sediment management

ANALYSIS STEPS

Using high-resolution remote sensing imagery of recent land use and land cover (LULC) change in the watershed, we model future LULC change without the PWS program. We then calibrate a hydrologic model to the watershed using climate, flow and turbidity monitoring data in order to identify intervention areas that would achieve the highest reduction in sediment yields above the baseline, *without*-PWS program case. We target these areas for restoration and conservation interventions subject to program implementation capacity, site costs and size of sediment reduction, generating a future LULC scenario *with* PWS program.

We run the calibrated hydrologic model with both predicted LULC maps (i.e., *with* and *without* the PWS program) to estimate by how much program interventions would reduce TSS concentrations at the municipal drinking water treatment plant intake. Using treatment plant data on sediment removal costs and discounted actual and projected PWS program costs, we estimate the ROI of the program —the ratio of the present value benefits and costs associated with the reduction in TSS in municipal drinking water. Because only a portion of the program cost is borne by the municipal water company, EMASA, we also estimate the ROI of the program for the water company itself.

CAMBORIÚ FINDINGS

We find that reductions in sediment treatment cost and water losses offset 80 percent of the public water company's investment in the watershed conservation program and 60 percent of the program's total costs over a 30-year (2015 to 2045) time horizon. The Camboriú PWS program's ROI is therefore <1 for just reducing TSS concentrations, although its ROI for the public water company exceeds 1 for time horizons of 43 years or more. The program thus would appear not to be justified on strictly financial grounds as only a control measure for TSS absent any cost-sharing with beneficiaries of the co-benefits it provides, including reduced risk of flooding and water supply shortages during the tourist high season. Importantly, both of these co-benefits are of high concern to the two municipalities in the watershed.

Cost-sharing can be achieved by incorporating watershed conservation costs into water user fees or levying a conservation fee on visitors during the high-season when water resources are strained. Specifically, a surcharge of only USD 0.005 (BRL 0.02 at the average 2015 exchange rate) per cubic meter water consumed — equivalent to less than 0.4 percent of the current average rate paid by municipal water customers or USD 1.25 (BRL 4) per household per year — would lift the ROI of the program above 1. Cost sharing of that amount would be justified as long as the combined value of the reduced risk of flooding and water supply shortages presently is at least USD 88,000 (BRL 275,000) per year — a condition that may well be satisfied given the importance of the booming tourism and real estate sectors to the local economy and the high economic cost associated with even localized flooding or relatively brief water supply disruptions.

For context — and acknowledging fully that the value of watershed conservation is highly context-specific — studies in Brazil and elsewhere of the value of improved water supply security or flood control report an average household willingness to pay several orders of magnitude higher than what would be required to lift the ROI of the Camboriú program above 1. Recognizing these additional benefits provided by the program, the Balneario Camboriú municipality is concluding a review of a new water tariff structure that incorporates watershed conservation and would cover the program's full operational costs.

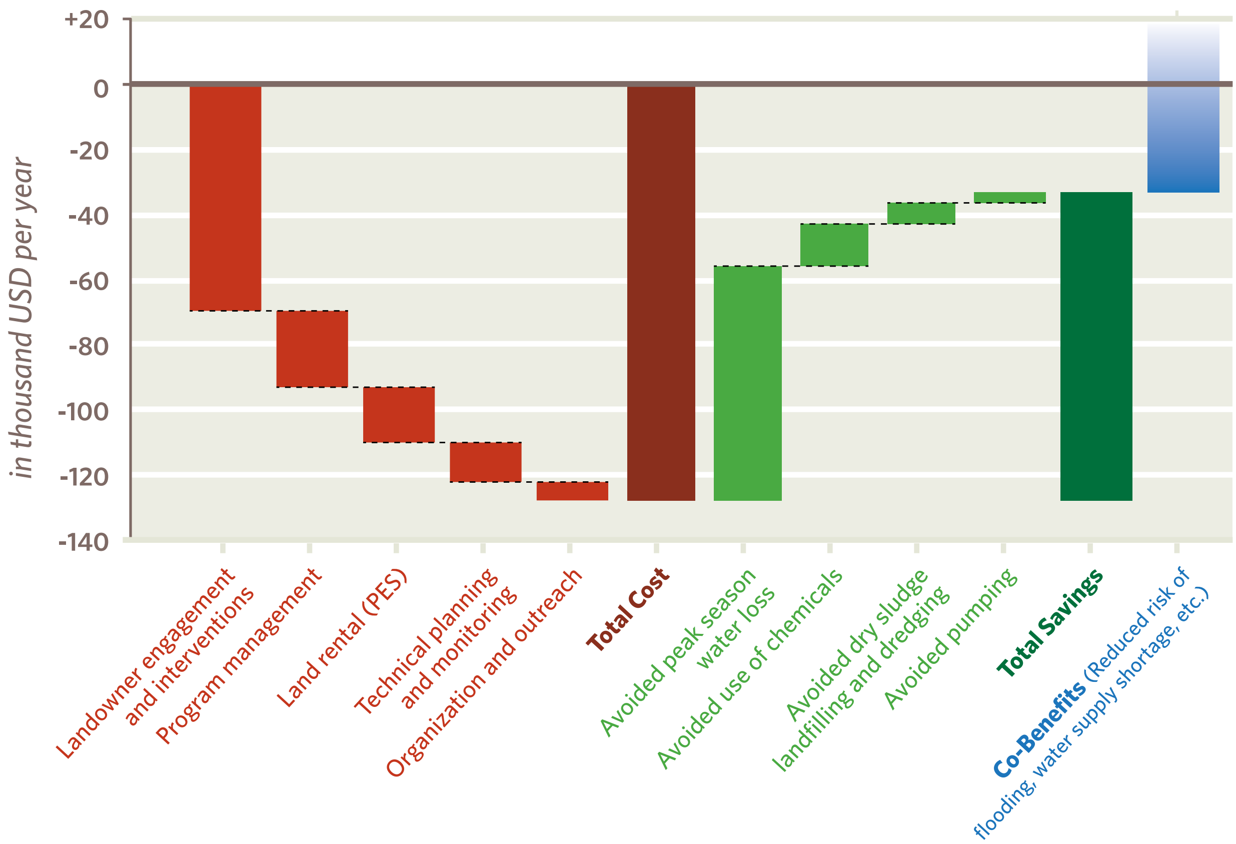

Figure ES-2: Comparison and composition of annual costs and benefits of the Camboriú PWS program, amortized over 30 years using Brazil's 3.85% social discount rate. Note: The value of co-benefits was not quantified in this analysis

CONCLUSION

Our findings are driven in part by program costs per hectare that substantially exceed those reported for other PWS programs in the region. Our higher cost estimates result from our full accounting for significant cost components often acknowledged but rarely quantified — specifically, the transaction and management costs associated with establishing, operating and assessing the impact of a PWS program designed to achieve high additionality and to be sustainable in the long term. Because transaction costs account for such a large share of total program costs, and because many of them are independent of the size of the intervention area, the ROI of the program could be improved by expanding interventions to the remaining high-sediment loading areas.

Our study provides a template for how to assess the business case of other PWS programs. It also highlights the fact that the business case for a given watershed conservation program for a particular investor can depend on the broader social-economic case, and on the ability to forge institutional arrangements that allow internalizing the value of the multiple benefits watershed conservation produces for diverse stakeholders.

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ESTIMATING RETURN ON INVESTMENT IN WATERSHED CONSERVATION:

Analytical Framework and Principles

INTRODUCTION

Preserving and restoring water quality is a major concern for cities around the world. In most cities, urban population growth, coupled with degradation of municipal source watersheds, has increased drinking water treatment costs. One recent estimate suggests that in one-third of large cities, costs per unit of treated water have increased on average by roughly 50 percent over the last century because of land conversion and development in source watersheds (McDonald et al., 2016). Restoring source watersheds can reverse this trend, and may be a cost-effective approach for cities to reduce drinking water treatment costs while enhancing supply resiliency and protecting biodiversity among other co-benefits (Alcott et al., 2013; Furniss et al., 2010).

The idea of deploying "natural infrastructure" to complement or substitute conventional engineering-based solutions to environmental problems has been receiving widespread interest (Das and Vincent, 2009; Ferrario et al., 2014; Kroeger et al., 2014; Kroeger and Guannel, 2014; Temmerman et al., 2013). This is certainly true for freshwater, where the impact of watershed conservation, restoration and management on improved water quality, flow regulation and flood control has drawn much attention (Alcott et al., 2013; Furniss et al., 2010; Opperman et al., 2009; McDonald and Shemie, 2014).

Three economic rationales are commonly advanced for investing in natural infrastructure solutions: cost-effectiveness, co-benefits and the precautionary principle.

- Natural infrastructure is a cost-effective alternative to manufactured "grey" infrastructure if it is at least cost-competitive with conventional engineering-based solutions in producing a specific target service or bundle of services (Kroeger et al., 2014; Ferrario et al., 2014).
- It also generates co-benefits that result from the additional ecosystem services any natural infrastructure provides beyond the specific target service(s) (Bennett et al., 2009), and that competing grey infrastructure generally does not provide (Kroeger and Guannel, 2014; Spalding et al., 2013).
- Finally, the precautionary principle supports preserving the option value associated with more intact watersheds and their higher resiliency to climate change and higher hydrologic service flows (Furniss et al., 2010) in the face of uncertainty about the size (Furniss et al., 2010) and value (Sterner and Persson, 2008) of reductions in future service flows due to ecosystem degradation coupled with the potential irreversibility of that degradation (Randall, 1988; Gollier and Treich, 2003). The precautionary principle can also justify conservation or restoration of natural systems based on the recognition that such systems have worked well so far (Wunder, 2013).

With the exception of the precautionary principle, assessing the economic rationale for investing in natural infrastructure requires sufficiently reliable quantitative information about both the benefits or "returns" a particular natural infrastructure solution delivers in a given place for a given level of investment, as well as its actual total implementation costs. While watershed conservation and restoration may offer substantial and widespread potential to cost-effectively deliver hydrologic services (McDonald and Shemie, 2014), in non-industrialized countries, few such credible return on investment (ROI) analyses exist for watershed conservation or restoration projects (Ferraro et al., 2012).

ROI ANALYTICAL FRAMEWORK: GENERATING CREDIBLE ROI ESTIMATES OF WATERSHED CONSERVATION PROGRAMS

Reliable ROI assessment of any natural infrastructure project requires application of an analytical framework that quantitatively links the biophysical and economic spheres and allows relating a specific natural infrastructure intervention to resulting changes in human well-being by quantifying the relationships along the *Intervention* → *Ecosystem Structure* → *Ecosystem Functions* → *Ecosystem Services* → *Benefits* → *Values* chain (Figure 1).

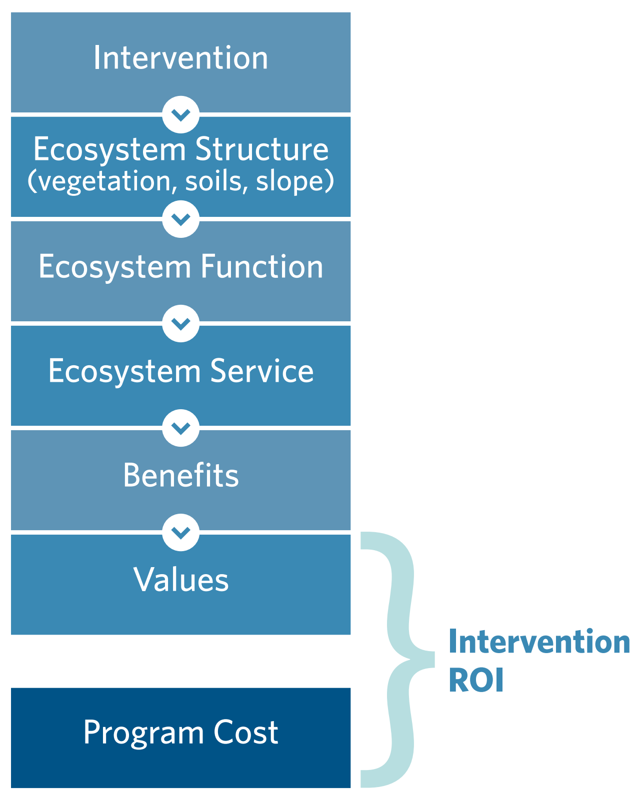

Figure 1: ROI analytical framework

IMPLEMENTATION

Implementation of this framework requires that an analysis follow seven key principles (see Figure 2):

	PRINCIPLE	EXAMPLE
1	Focus on ecosystem services and clearly distinguish among ecosystem functions, ecosystem services, benefits and values (Boyd and Banzhaf, 2007; Brown et al., 2007; Tallis and Polasky, 2009; Kroeger, 2013)	Reduced concentration of total suspended solids (TSS) in municipal intake water *(service)* vs reduced erosion *(function)*, reduced treatment needs *(benefit)* or lower treatment cost *(value)*
2	Focus on final ecosystem services, that is, "components of nature that are directly enjoyed, consumed, or used to yield human well-being" (Boyd and Banzhaf, 2007:619)	Municipal intake water with lower TSS concentrations vs improved soil retention in watershed
3	Define services in benefit-specific terms (Boyd and Banzhaf, 2007; Landers and Nahlik, 2013; Keeler et al., 2012), using metrics that reflect the service characteristics crucial to benefit generation (physical unit, spatial and temporal incidence)	Reduced average hourly TSS concentrations (ml/m^3) at the municipal water treatment plant intake point(s) vs reduced tons of total sediment load per year somewhere in the river
4	Use appropriate (locally calibrated and validated) ecosystem service production functions (National Research Council, 2005) that incorporate spatial attenuation effects between intervention and service provision sites, if any	Well-calibrated hydrologic model that simulates all relevant sediment sources and sinks in the source watershed and their effects on TSS concentrations at the municipal water treatment plant intake
5	Construct a true baseline of service flows via development of "counterfactual" service flow estimates, to control for the effect of other factors on service flows (Blackman, 2013; Ferraro, 2009; Ferraro and Pattanayak, 2006; Pattanayak, et al. 2010)	Modeled future change in land use/land cover (LULC) in watershed without the watershed conservation program, and resulting TSS concentrations at municipal water treatment plant intake point(s)
6	Use empirically-based benefit functions for key service beneficiaries that show the quantitative relationship between particular service flows and specific, actual benefits	Treatment plant application of chemical x reduced proportionally with TSS concentrations; chemical y reduced by $0.05 ml/m^3$ up to TSS concentration a; water lost in treatment sludge reduced proportionally to TSS concentrations
7	Use appropriate valuation approaches to quantify the changes in human wellbeing associated with those benefits (Brown et al., 2007; Griffiths et al., 2012; Wilson and Carpenter, 1999)	Avoided cost of chemicals used for TSS removal by municipal water treatment plant; increased revenue for municipal water plant from reduced loss of marketable water in sludge

Figure 2: Key principles for generating reliable ROI analyses

Importantly, this framework yields natural infrastructure ROI estimates expressed in the same performance metrics routinely used to evaluate relevant engineering alternatives. This facilitates identification of instances where natural infrastructure solutions are competitive with, or outperform, engineered alternatives — something that is critical to attracting increased investment in natural infrastructure solutions.

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Camboriú River Payment for Watershed Services Project

INTRODUCTION

Balneário Camboriú in Santa Catarina state is a famous beach destination in southeastern Brazil that attracts increasing numbers of both domestic and international visitors (Ferreira et al., 2009; Lohmann et al., 2011). As a result of the booming tourism and civil construction sectors that now dominate the local economy, the combined population of Balneário Camboriú and neighboring Camboriú city, approximately 200,000 year-round (Instituto Brasileiro de Geografia e Estatística, 2016a, 2016b), now swells to more than 800,000 during the summer high season (December through early March).

Both municipalities count on the Camboriú River as a reliable low-cost source of drinking water supply. However, the increasing demand, especially during the summer tourist high season, severely stresses this supply. This is due primarily to a lack of any large-scale water storage infrastructure in the watershed, such as reservoirs that could buffer the impact of extended low streamflow events. The topography of the watershed, with its broad alluvial plain and relatively small surrounding sloped areas, is not naturally well-suited to reservoirs. However, high sediment loads at the water plant's intake point also are limiting supply as they lead to large water losses in the treatment of stream water for municipal water supply.

To avoid future supply shortfalls, the Balneário Camboriú Water Company, EMASA (which supplies both municipalities), evaluated options for securing sufficient future water supply. These include storing water in the watershed through flooding of native forest and agricultural lands; transferring water from a neighboring watershed characterized by substantially lower water quality that would necessitate advanced treatment; and conserving and restoring natural lands in the Camboriú watershed to maintain the high-quality water supply historically provided by the Camboriú River.

Because of the high projected costs of the first two options and the promising results of initial feasibility assessments of the third, EMASA decided to first invest in protecting remaining natural forests and restoring degraded areas with high sediment loading to reduce treatment water losses and costs. Should this prove insufficient to achieve a reliable water supply, investments in the other two grey infrastructure options will need to be considered.

To implement the watershed conservation strategy, in 2013 EMASA created the Camboriú Payments for Watershed Service (PWS) project in partnership with The Nature Conservancy, the municipalities of Balneário Camboriú and Camboriú, the Camboriú Watershed Committee, the State Sanitation Regulatory Agency (Agesan), the National Water Agency (ANA), Santa Catarina State's Environmental Information and Hydrometeorology Center (EPAGRI-CIRAM) and the Camboriú city council.

PROJECT AREA

The Camboriú River watershed is located in Santa Catarina state, southern Brazil, and has a humid subtropical climate. The drainage area is 199.8 square kilometers large, with EMASA's water intake located in the lower portion of the watershed, just before the beginning of the urbanized area. The area upstream from the water intake — the area of interest for this study — comprises approximately 13,000 ha and receives the flows of the Braço, Macacos, Canoas and Lajeado subwatersheds.

The land use pattern in the Camboriú watershed resembles that of many other coastal watersheds in Brazil's Atlantic Forest. The urban area is heavily concentrated along the coast and characterized by a thin strip of very high-density, high-rise development at the ocean front surrounded by an established high- to medium-density mixed use area. This, in turn, is followed by a zone of residential sprawl that is fast expanding into the alluvial floodplain. The alluvial plain beyond the urban area is dominated by pasture and row-crops (primarily irrigated rice), while the slopes are primarily in native forest, but also feature pastures and, increasingly, timber plantations, primarily eucalyptus.

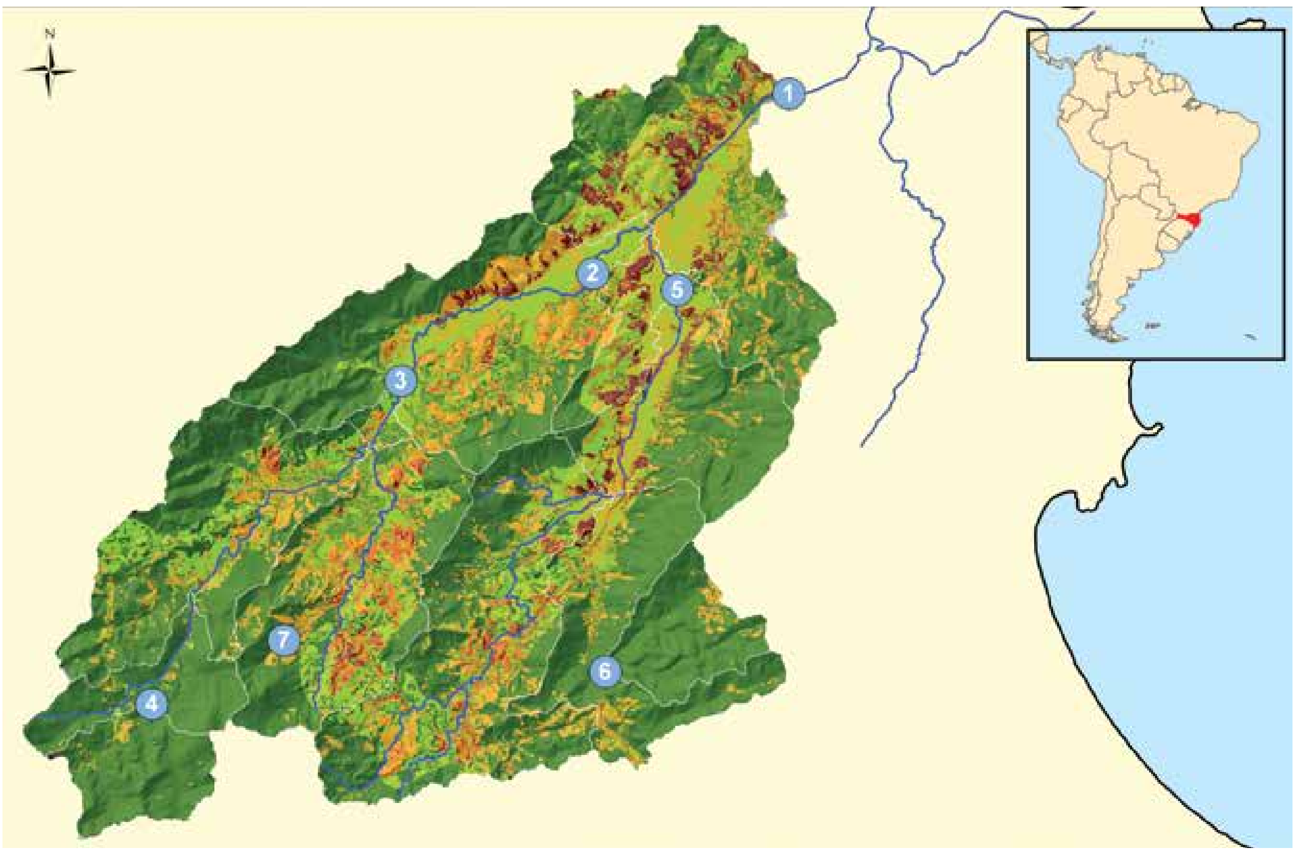

Figure 3: Camboriú River watershed upstream of municipal drinking water treatment plant intake, monitoring stations and modeled sediment yield rates in 2014

Average annual Sediment Yield form HRU 2014 - ton/daily

- 0.0—0.1
- 0.2—1.0
- 1.1—5.0
- 5.1—10.0
- 10.1—500.3

Hydrologic Monitoring Station

1. Water Intake - EMASA
2. Braço outflow
3. Braço middle basin
4. Braço headwaters
5. Canoas/Macacos outflow
6. Macacos headwaters
7. Louro climatic gauge

The high rates of both forest loss and regrowth experienced in recent decades in the Camboriú watershed have the potential to make the use of counterfactual analysis particularly important because the latter may be able to identify where deforestation and regrowth, respectively, are likely to occur, thus allowing a targeting of interventions that can increase additionality of impacts and program ROI.

The PWS currently implements three interventions:

1. restoration of degraded riparian and headwater areas, through fencing for cattle exclusion and planting of native tree seedlings or enrichment, depending on the state of degradation (highest priority);

2. conservation of relatively intact riparian areas featuring regenerating forest, through riparian fencing for cattle exclusion (second-highest priority); and

3. restoration of degraded upland forest on steep slopes through fencing for cattle exclusion and either planting of native tree seedlings or enrichment, depending on the state of degradation (lowest priority).

Landowners receive payments as compensation for the maintenance of interventions on their property and the recurrent annual opportunity cost associated with the forgone use of those sites, with payments contingent on good maintenance of interventions. Payments by the Camboriú PWS program are based on the average opportunity cost of forgoing grazing in the watershed (BRL223 [USD 70]/ha/yr) and the size, priority ranking and level of degradation of the enrolled area. Restoration and conservation of riparian areas and restoration of high sediment contribution areas earn 1.5 times this rate, while conservation areas not classified as riparian earn 0.5 times.

The program also intends to promote the implementation of dirt road mitigation measures for sediment control by the two municipalities, especially on slopes. While improved dirt road management practices have the potential to substantially reduce sediment loading into streams, their effect is highly site-dependent, and current data limitations prevent modeling watershed-scale impacts.

Analyses Needed to Estimate ROI

We apply the ROI analytical framework (Figure 1) to estimate the predicted ROI of the Camboriú PWS program as a sediment control measure. The necessary analyses are described below and summarized in Figure 4.

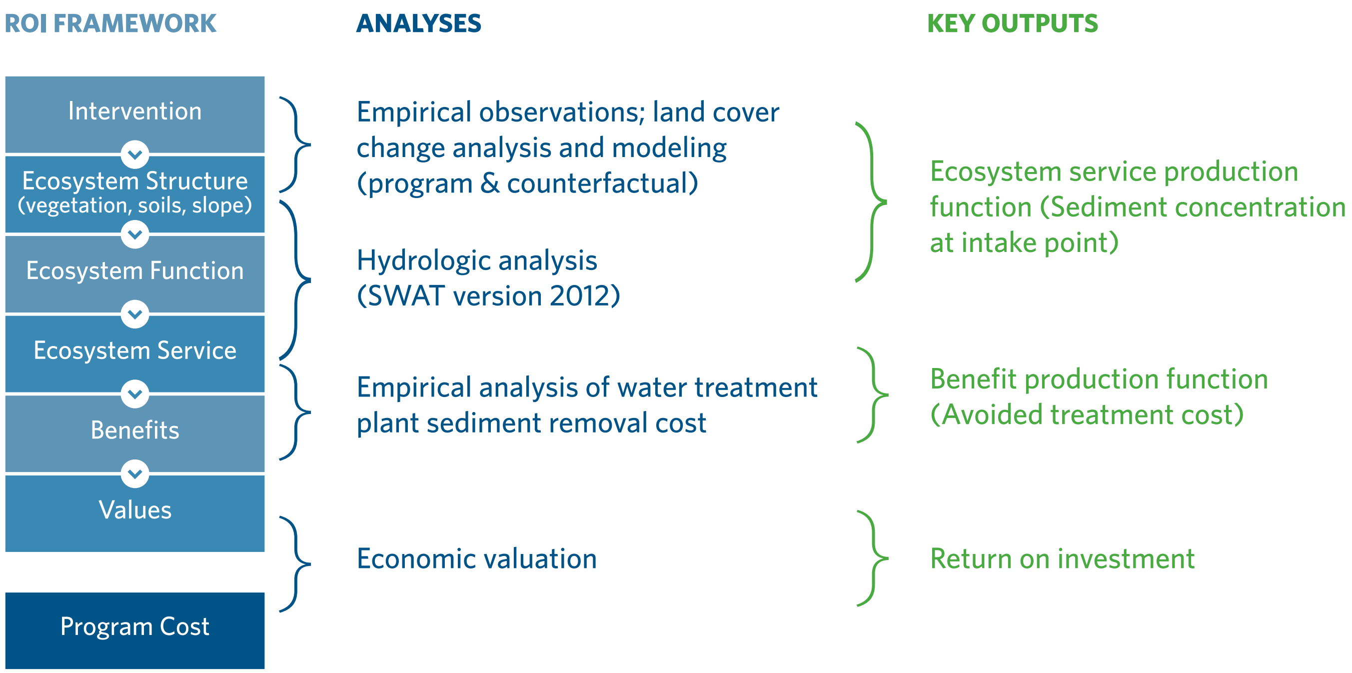

Figure 4: Analytical framework and corresponding analyses used to assess the return on investment of the Camboriú watershed conservation program for water treatment plant sediment management

1. EMPIRICAL OBSERVATION, LAND COVER CHANGE ANALYSIS AND MODELING

Using high-resolution (1 meter) land use/land cover (LULC) data from a recent time period (2004 to 2012) expected to be representative of the near- to medium-term future, we estimate a LULC change model for the watershed. We use this model to generate counterfactual (i.e., without the PWS program) LULC for the year 2025, by which time the program is expected to have enrolled the lands considered most crucial for sediment control whose owners are interested in participating in the program, and many of the interventions are expected to have attained much of their eventual full functionality.

2. HYDROLOGIC ANALYSIS (SWAT VERSION 2012)

We then use this counterfactual LULC layer and the interventions carried out by the PWS program to predict the conservation LULC resulting with the program. We calibrate the Soil and Water Assessment Tool (SWAT version 2012_Rev._637) to the watershed using climate data, flow and turbidity data from existing and newly added hydrologic monitoring infrastructure, and current (2012) LULC (Fisher et al., 2017). The model then was run on both the counterfactual and the conservation LULC scenarios for the year 2025 to estimate the PWS program-attributable reduction in sediment levels at EMASA's water intake.

3. EMPIRICAL ANALYSIS OF WATER TREATMENT PLANT SEDIMENT REMOVAL COSTS

The concentration of sediment in intake water affects various aspects of the treatment process (Appendix A1). The main operational processes in EMASA's treatment plant that are impacted by sediment in intake water are the application of chemicals for coagulation and flocculation of raw water; sludge discharge and disposal; water pumping to and within the treatment plant; and back flushing of the final sediment gravity filters with already-treated water.

Since the heavier sediment fraction settles in the intake channel prior to reaching the pumping station, the fraction reaching the treatment plant is composed almost exclusively of suspended solids (TSS). TSS thus is the target ecosystem service metric of concern. Accordingly, the hydrologic modeling was set up to yield the estimated impacts of the PWS interventions on TSS concentrations at the treatment plant intake. Likewise, our benefits analysis is based on changes in intake water TSS concentrations.

4. ECONOMIC VALUATION

We use empirically derived sediment cost functions to estimate the value of the sediment reductions the program produces at the water treatment plant and compare this value to program costs. Finally, we calculate three ROI metrics useful for describing the economic performance of natural infrastructure (NI) projects.

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Detailed Steps to Estimate ROI

1. Historic Land Cover Change Analysis
2. Future Land Cover Change Modeling
3. Hydrologic Modeling
4. Targeting of Interventions Based on SWAT and LCM Results
5. Benefits Estimation
6. PWS Program Costs
7. ROI Calculation

Historic Land Cover Change Analysis

1

The choice of spatial resolution of land cover data was based on observations of the spatial characteristics of individual instances of recent land cover changes in the watershed, many of which are incremental and fine-scale.

Similarly, the interventions carried out by the PWS program result in fine-scale land cover changes often on the order of less than 20 m. Because change at that scale often is undetectable even with medium-resolution imagery such as 30 m (Landsat), we acquired readily available commercial sub 1m resolution imagery for the land cover change analysis.

We chose 2003 to 2012 as our land use/land cover change (LULCC) reference period for historic LULCC, which were the earliest and latest years, respectively, for which 1 m resolution data were available for the study area. This period is representative of the current phenomenon of urban sprawl into the hinterland following the maximum densification of the coastal zone around the year 2000 (Ferreira et al., 2009). We processed land cover images using Feature Analyst 5.1.21 (Overwatch Systems Ltd.) for ArcGIS and created feature class polygons for each of the land cover classes that had been identified based on SWAT modeling needs. We then used ground reference points (539) collected during a field visit to accurately classify and calibrate the high-resolution imagery.

To increase the accuracy of the land cover products and ensure that they reflect actual land use, we used our knowledge of the watershed to develop rules that reclassified specific transitions involving plantations (see Supplementary Information), eliminating highly unlikely transitions by changing the respective land cover of any pixels affected by these transitions (Fisher et al., 2017).

Results

OBSERVED LAND COVER CHANGE

Net land cover change observed between 2003 and 2012 was 562 ha, or approximately 4 percent of the study area (13,668 ha). The single largest change was a reduction in pasture; this was balanced by increases in plantations, bare, impervious and forest. While forest extent showed a small net increase, forest cover was removed from an estimated over 230 ha during this nine-year period.

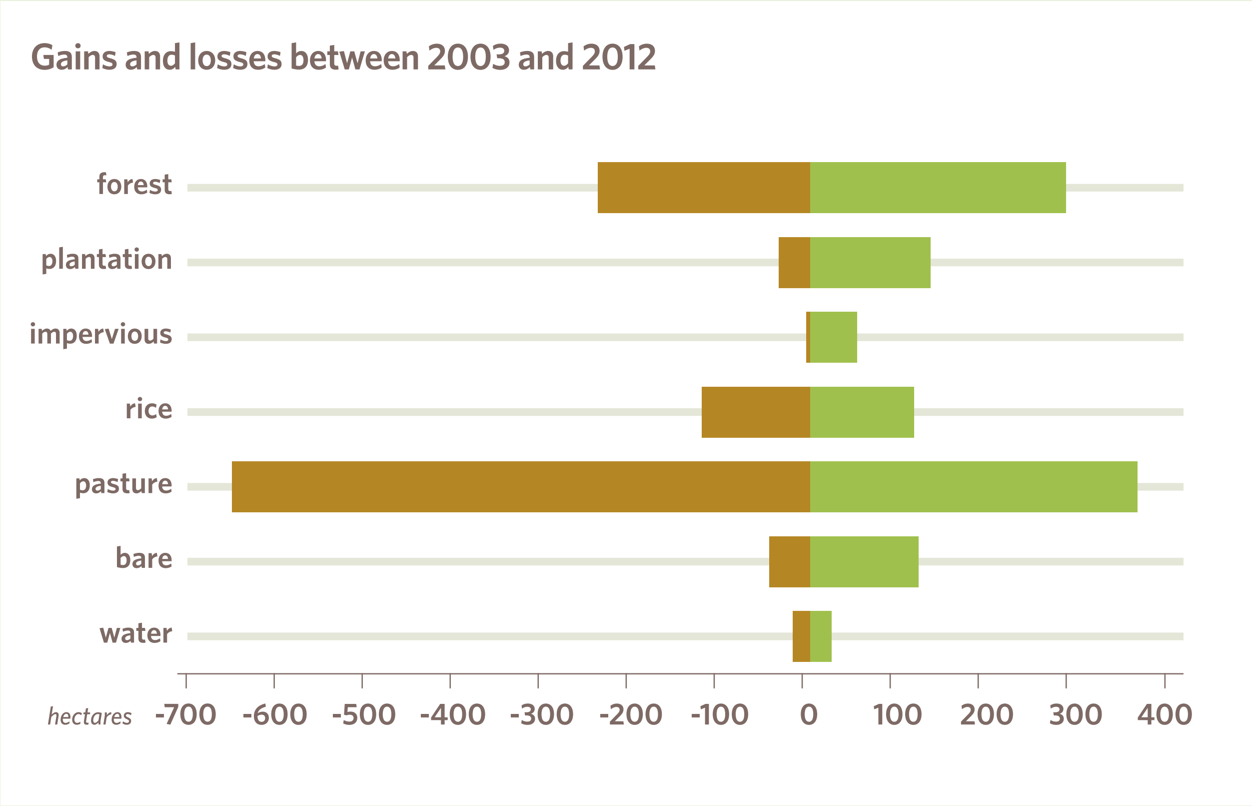

Figure 5: Land cover change in the study area during 2003–2012 (gross)

Modeling future land cover change

2

Land use change is a complex process that is not strictly biophysical-deterministic but rather is influenced also by dynamic and non-linear interactions between nature, humans and larger economic processes (Pérez-Vega et al., 2012). Historic land cover change reflects the past results of those interactions and thus, barring major changes in land use drivers, can be a useful indicator of potential future change. We used Land Change Modeler (LCM) for ArcGIS 2.0 to identify spatially explicit land cover change between the rules-adjusted 2003 and 2012 land cover layers. The model is calibrated on land cover change observed during a historic period, a process during which it assesses the importance of user-provided explanatory variables in discriminating between areas of change and no change.

Out of the large set of potential drivers of LULCC (Blackman, 2013; Ferretti-Gallon and Busch, 2014; Soares-Filho et al., 2004), we selected eight for inclusion in our estimation of the LCM model for the watershed that have been found to be significant drivers of LULCC in other coastal Atlantic Forest regions experiencing similar land use changes as those observed in Camboriú.

By basing our LULCC prediction for 2012 to 2025 on LULCC observed during 2003 to 2012, we make the assumption that the composition and relative strength of the ultimate causes of land use change during the prediction period will be the same as during the earlier period. Principally, this assumes that the economics of the various agricultural activities and of development remain largely unchanged.

In addition, agricultural input and output prices, as well as real estate development-related policies (e.g., zoning regulations), have major impacts on the economics of land use in the watershed. Given the relatively short time period covered in our projection (13 years), the assumption of no substantial change in the economics of land use in the watershed appears reasonable.

Results

PREDICTED LAND COVER CHANGE BY 2025

Total predicted net change (582 ha) during 2012 to 2025 is slightly larger than during 2003 to 2012. A reduction in pasture (-279 ha) is the single largest predicted LULCC, accompanied by a much smaller contraction in the area in irrigated rice (-12 ha). These reductions are balanced primarily by increases in the area in plantations (+176 ha) and, to a lesser extent, in impervious (+48 ha), bare (+43 ha), and forest (+24 ha) covers.

An analysis of the individual transitions reveals that while forests show a net increase fueled by a decline in pasture area, by 2025 more than 310 ha of forest are predicted to be converted to pasture, much of it in the middle watershed (orange areas in Figure 6). These predictions are consistent with the empirically observed continuing replacement of mature native Atlantic Forest with regrowing forest patches (Joly et al., 2014). An analysis of the spatial patterns of change reveals that while pastures are being replaced by plantations and forest throughout the watershed, the effect is most pronounced in the headwater areas, matching Teixeira et al.'s (2009) observation that Atlantic Forest regrowth was highest at higher elevations, farther from urban areas, and farther from roads (green areas in the lower portion of Figure 6).

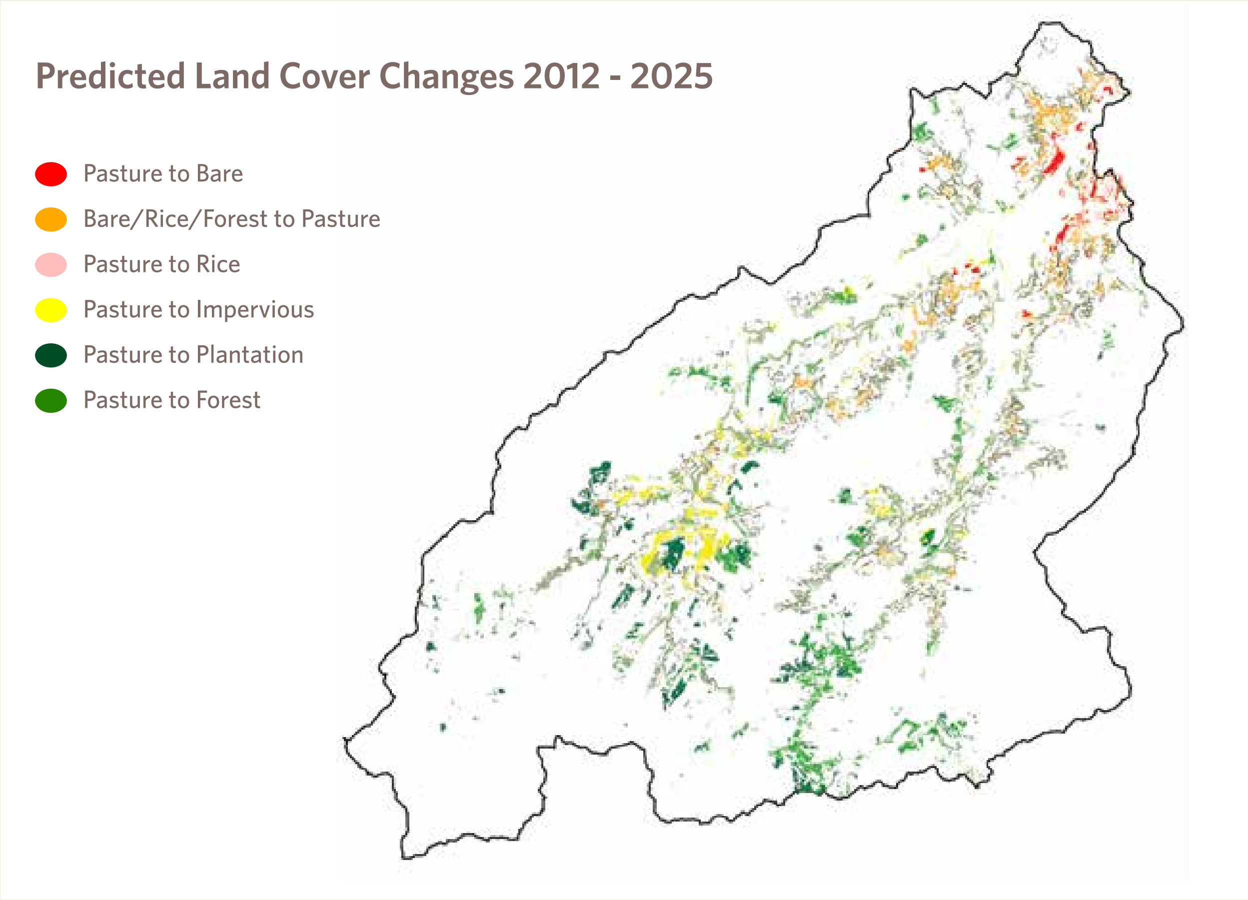

Figure 6: Predicted 2012-2025 land cover change in study area, assuming no PWS program

Hydrologic Modeling

3

We use the Soil and Water Assessment Tool (SWAT 2012 Rev. 637) (Arnold et al., 1998; Gassman et al., 2007) to estimate the change in the watershed's sediment export under the planned PWS program interventions compared to a counterfactual scenario in which those interventions do not occur.

MODEL CONSTRUCTION, DATA AND CALIBRATION

The SWAT model was built with the 2012 land use map developed in the LULC analysis, and calibrated first to daily discharge and then daily sediment concentration monitoring data from 2014 to 2015 using a split-sample calibration method (Fisher et al., 2017; see Appendix A2). A linear rating curve was built for turbidity-to-sediment load based on lab analysis of 35 grab samples at the EMASA intake. We judged that the two-year offset between land use map and discharge was small enough to have a minimal effect. For both discharge and sediment load, model accuracy was assessed based on out-of-sample data from 1/1/2015 to 8/31/2015 using three complementary approaches and was found to be good (Fisher et al., 2017; Appendix A2).

We then ran the SWAT model on the counterfactual (no PWS program interventions) and the intervention scenario 2025 LULC maps in 2025. For the counterfactual scenario, we used the parameter values and settings from the calibrated model. We used the 2014 monitored catchment discharge and EMASA intake volume in all our modeling and calculations. For the intervention scenario, we also left parameter values and settings unchanged from those of the calibrated model.

Hydrologic Monitoring

Historic hydrologic monitoring data are limited to fragmented series of water quality and river level data collected at the treatment plant raw water intake (point 1 in Figure 3). In 2013, as part of a new flood early warning system, river level gauges were installed at the outflows of the two headwatersheds (Braço and Macacos; points 2 and 4, respectively, in Figure 3) along with a climate gauge in the Braço headwatershed (point 6 in Figure 3) and a full meteorological station at the EMASA intake point.

To generate continuous turbidity and flow time series data that would allow improved SWAT calibration, in 2014 automatic turbidity gauges were added to the two headwater outflow monitoring stations. All of these stations are telemetry-linked and collect hourly measurements, except for the turbidity probes, which collect measurements every 15 minutes. Measurements are transmitted to the EPAGRI-CIRAM server and subjected to preliminary quality control using range, step and persistence tests.

Results

PREDICTED SEDIMENT LOADING

Figure 3 shows the SWAT-estimated sediment yield in 2014 of all hydrologic response units (HRUs) in the Camboriú catchment above the EMASA intake point (point 1 in the figure). Given our assumptions of spatial homogeneity of the TSS fraction in sediment loading into streams in the catchment and identical attenuation of TSS between any stream location in the catchment and the EMASA intake, the map indicates the estimated contribution of each point in the catchment to 2014 TSS loads at the municipal water intake. Figure 7 (left panel) shows the predicted sediment yield of catchment HRUs in 2025 absent conservation and restoration interventions.

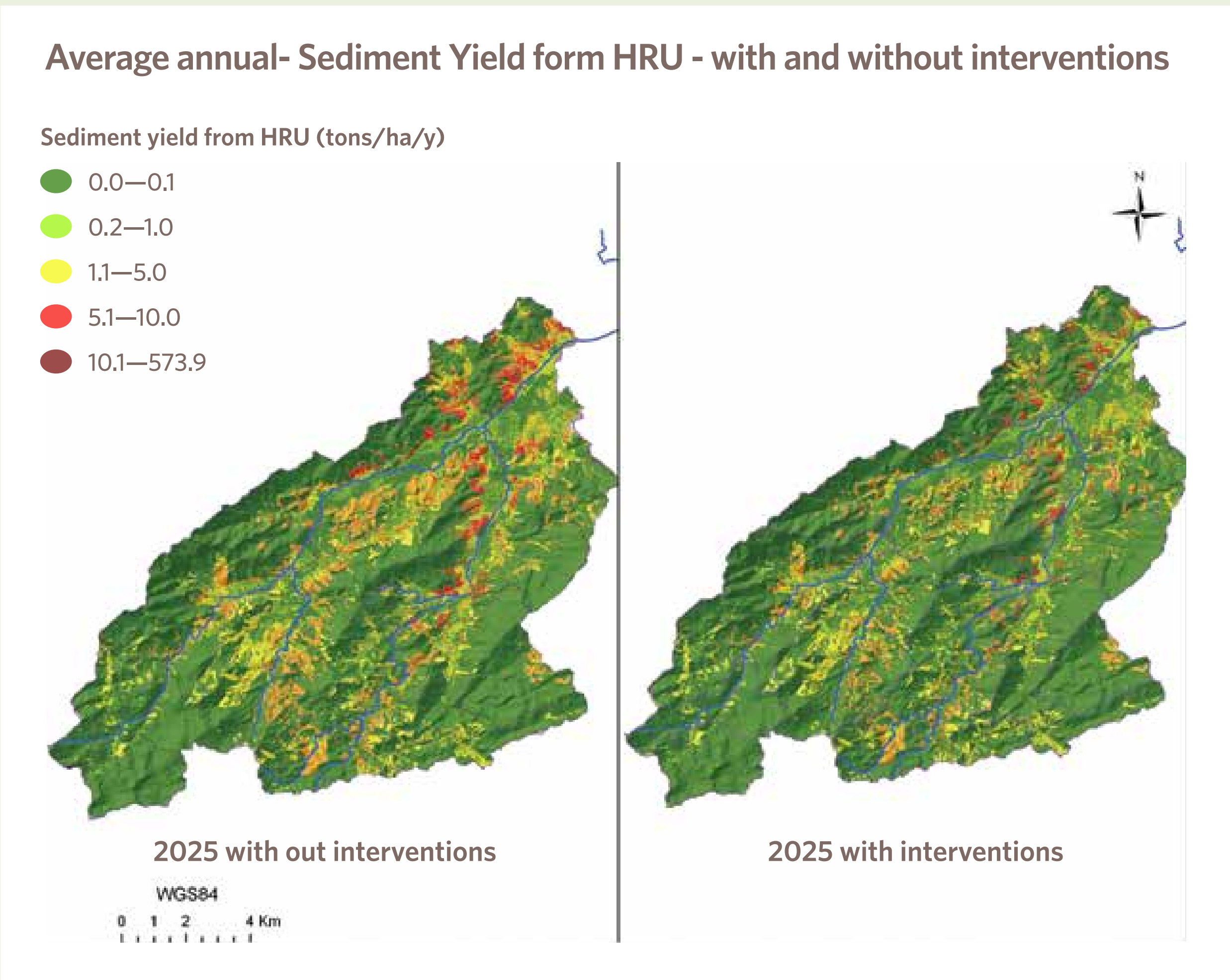

Figure 7: Modeled annual sediment yield in counterfactual (left panel) and intervention (right panel) scenarios

4

Targeting of Interventions based on SWAT and LCM Results

Cost-effective targeting of PWS interventions requires selecting the intervention portfolio (both sites and interventions) based on both costs and benefits (Duke et al., 2014), where benefits are measured as reductions in TSS concentrations at the EMASA intake point. However, modeling TSS yield from individual sites was beyond the scope of our analysis as it would require highly spatially resolved soil information, which does not exist.

Instead of TSS yield, we use sediment yield of a site as a proxy for the contribution of the site to TSS concentrations at the EMASA intake. This assumption is unlikely to unduly bias our targeting because the small size of the watershed and the absence of reservoirs or significant overbank flow events make it unlikely that there is much TSS attenuation between intervention sites and the water intake, and because any TSS removal that is occurring will occur primarily on floodplains or in rice or fish ponds in the lower watershed, and thus would affect TSS delivery to EMASA from all upstream intervention areas equally.

To target restoration activities, we first identified potential intervention lands as lands currently in pasture or bare covers (excluding roads). Within this subset, we identified lands located in riparian or headwater areas which, following the general rule defined by the Brazilian Forest Code (Soares-Filho et al., 2014), we defined using a buffer of 30 m on both sides of a stream and a radius of 50 m around headwaters. We focused on riparian and headwater lands as these are most important for preventing sediment stream loading. From these lands, we excluded all lands that the LCM analysis predicted to revert to forest by 2025. From this set of potentially restorable riparian or headwater lands not expected to experience forest regrowth absent intervention, we selected for restoration activities the highest sediment-yielding lands in 2012 as indicated by our SWAT model, until reaching the current implementation capacity of approximately 326 ha by 2022.

To target conservation activities, we selected the 313 ha that our LCM model predicted to change from forest in 2012 to non-forest in 2025. We quantified the expected sediment yields of these areas by running the calibrated SWAT model with the LCM-predicted 2025 land cover and selected all of them for conservation interventions because their modeled sediment yields exceeded 10 t/ha/yr.

Results

TARGETING OF INTERVENTIONS

Figure 8 shows the areas selected for restoration (326 ha) and conservation (313 ha) activities, based on current and future sediment yield and predicted forest cover change. A comparison of both scenarios reveals that the interventions will substantially reduce sediment yield from the majority of high sediment yield areas predicted to exist in 2025 in the absence of the interventions. Nevertheless, the presence of high sediment yield areas in the intervention scenario (bright and dark red areas in the right panel in Figure 7) indicates that more opportunities exist to further reduce sediment loading of streams.

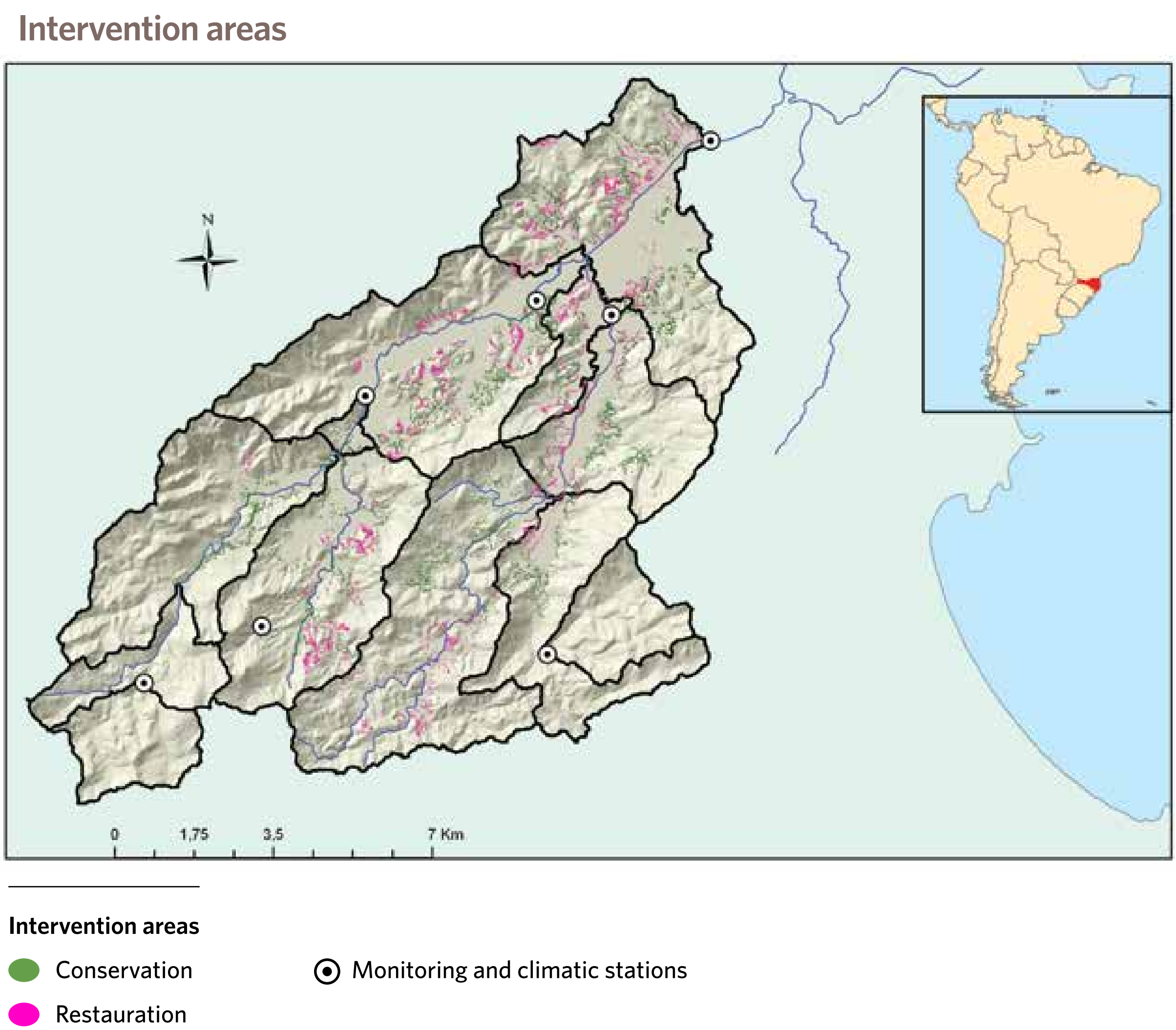

Figure 8: PWS interventions targeted using current land cover and use, predicted counterfactual land cover change and estimated contribution of lands to TSS concentrations at the treatment plant intake

Benefits Estimation

5

To estimate benefits, we developed a future scenario to assess the impact of reduced TSS levels from catchment restoration and conservation on municipal drinking water provision. (See Appendix Table A1 for details on the sediment-related water treatment plant operational cost calculations.)

In estimating the value of reduced sediment concentrations in intake water, we distinguish between the period of peak demand (the December through March tourist high season) and the rest of the year (April through November).

We assume that in off-peak months, there is no demand for any additional water outputs. We assume that the reduced water loss from lower TSS loads and consequently lower sedimentation basin sludge discharge and filter backwashing needs is used to reduce treatment plant intake rather than increase plant output. This reduction in intake leads to energy savings in the form of a reduction in pumping of intake water to and within the treatment plant; reduced coagulate and flocculant application; and reduced sludge disposal.

In contrast, during peak months, when excess supply frequently approaches zero and one or both municipalities face a threat of shortages, we assume that the reduced water loss during sediment removal instead is used to increase drinking water output to permit keeping short-term storage infrastructure at capacity to safeguard against acute supply shortfalls.

Thus, during peak months, the value of reduced sediment concentrations in addition to reduced treatment costs also includes the additional revenues collected from increased water sales. We use the current (August 2015) user type and use volume-weighted marginal price of water and sewage (automatically billed at 80 percent of water use) rate of USD 1.90 (6.08 BRL) per cubic meter of water to estimate the additional revenue for EMASA from the reduced peak season water loss.

TEMPORAL INCIDENCE OF BENEFITS

The SWAT-modeled TSS reductions for the future scenario represent the full impact achieved once interventions have developed full ecological functionality. However, that functionality, and hence the impacts on TSS concentrations at the EMASA intake, do not instantaneously materialize. Rather, they increase over time as a function of the increase in total intervention area from 2015 to 2022 and the time it takes a given intervention to develop its full functionality.

To calculate the TSS reduction produced in a given year, we first estimate the proportion of its full or maximum TSS reductions that is achieved in a given year. Implementation of additional conservation (313 ha) and restoration (326 ha) interventions in 2015 to 2022 is spread evenly over those years.

With conservation interventions targeted to lands expected to experience land cover conversion by 2025 in the absence of the PWS, conservation activities achieve full functionality (i.e., maximum TSS reduction over the counterfactual) in the year of their implementation. Restoration measures, however, only gradually gain functionality over time. We assume that their impact on TSS is zero in the year of installation and then experiences a linear increase through year 10, when full functionality is attained. Because of this delay, the full impact of the total extent of restoration is not achieved until year 19 of the program (2033).

Results

ESTIMATED BENEFITS

REDUCTION IN TSS CONCENTRATIONS AT MUNICIPAL WATER INTAKE

Once all interventions are implemented and have attained their full functionality, the PWS program will reduce average annual TSS concentrations at the treatment plant intake by an estimated 14 percent compared to the no intervention baseline, from 92 mg/l to 79 mg/l. The average annual volume-weighted reduction during 2015-2045 in the TSS concentration in intake water is an estimated 11 mg/l.

AVOIDED TREATMENT COSTS AND REDUCED WATER LOSSES ATTRIBUTABLE TO THE CAMBORIÚ PWS PROGRAM

We estimate that during 2015 to 2045 the PWS program will produce average annual sediment reduction-related benefits of USD 194,000 (Table 1). Avoided peak season revenue losses from water lost in sediment removal dominate total benefits by value (76 percent), with reduced chemicals usage and sludge disposal also generating substantial cost reductions (14 percent and 6 percent of total treatment plant benefits, respectively).

BENEFIT	AVERAGE ANNUAL IMPACT, 2015-2045	
	Quantity	Value (2014 USD)*
Avoided peak season water loss, ***m³***	77,400	147,000
Avoided use of chemicals, PACl[1], ***kg***	73,400	27,800
Avoided use of chemicals, Polymer, ***kg***:	150	560
Avoided off-peak water pumping (to and within WTP), ***kWh***	77,600	6,100
Avoided dredging, ***m³***	110	1,050
Reduction in dry sludge landfilling, ***t***	640	12,000

* Undiscounted [1] Polyaluminum chloride

Table 1: Estimated average annual impact of Camboriú PWS program on water treatment plant, 2015-2045

6

PWS Program Costs

We compiled information about the full costs of individual program activities to date and projected future annual costs based on the expected time profile of each activity associated with the Camboriú PWS program. These activities comprise hydrologic, political and economic feasibility studies; coordination, communication and program design; program management and administration; landowner engagement and contract development; planning and implementation of interventions (restoration and conservation); payments to landowners; and compliance monitoring.

To calculate our ROI metrics, we distinguish between total program costs and costs incurred by EMASA. Total costs include grants from multilateral institutions and private foundations that supported the development and implementation of the PWS program and costs borne by other institutions partnering in the PWS program, such as EPAGRI-CIRAM. Both ROI measures are informative: EMASA's ROI allows assessing the business case for the company's investments in the program as a sediment control measure, while the broader ROI measure indicates whether the program is economically justified overall on the basis of solely its sediment control benefits.

Results

PROGRAM COSTS

Overall program costs are heavily front-loaded (Figure 9), with 65 percent of total costs through 2045 incurred by 2023. Installation of restoration and conservation interventions and payments to landowners account for less than half (47 percent) of total program costs.

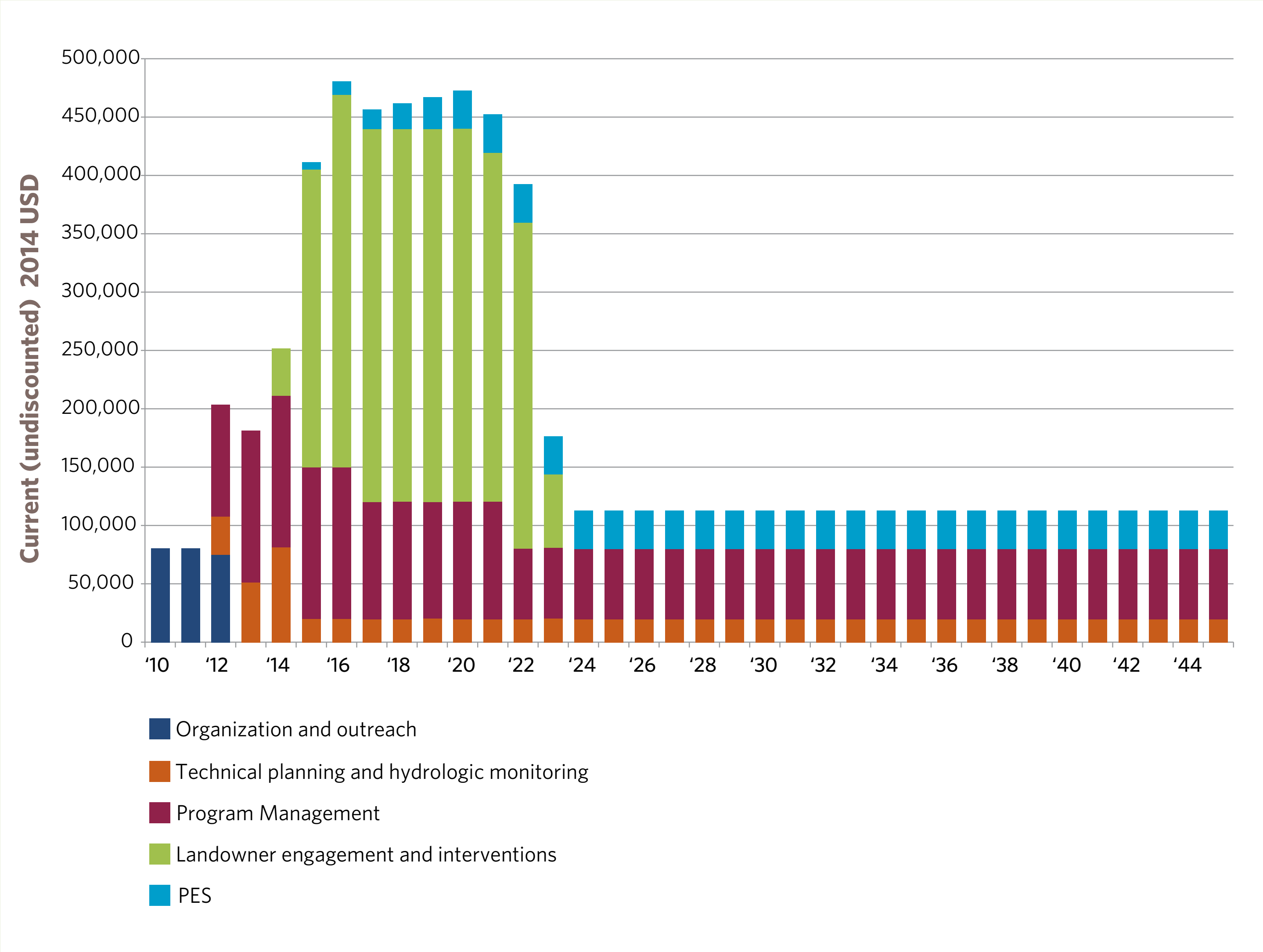

Figure 9: Annual program costs by major component

ROI Calculation

7

We calculate three ROI metrics of the Camboriú PWS program for EMASA and for the program overall: 1) The cost-effectiveness in reducing TSS, expressed as average reduction in mg TSS per liter in intake water per million USD (2014 prices) invested in the PWS program, or as 2) average kg TSS removed from intake water per dollar invested; and 3) the benefit-cost ratio or monetized ROI, calculated by dividing the monetary value of TSS reductions in municipal water treatment plant intake water by the costs of the PWS program.

In addition to reducing TSS, the PWS program also produces co-benefits of high concern to the two municipalities: flood attenuation, and reduction in the risk of municipal water supply shortages during the tourist high season because of low flows in the absence of large-scale water storage infrastructure. These positive externalities justify cost-sharing of the PWS program, which could be achieved by including watershed conservation costs in municipal water user fees or levying conservation fees on high-season tourists. Both would be justified: the former on the grounds that the benefits of supply interruptions and flood risk reduction would be broad-based; the latter because a disproportionate amount of the benefits of reduced risk of supply shortages accrues to tourists, who, during the high season, account for three-quarters of the combined population of the two municipalities.

In calculating the ROI metrics, we discount all costs and benefits of the PWS program through 2045 to their 2014 present value (PV) equivalents using Brazil's estimated social consumption discount rate of 3.85 percent (Fenichel et al., 2016). Social rates, not market discount rates, are generally recognized as the appropriate rates to use in evaluating long-lived publicly financed projects like environmental protection (Arrow et al., 2013).

The 30-year time horizon for our analysis was chosen to ensure broad comparability of our cost-effectiveness estimates for the Camboriú PWS program with that of investments in grey drinking water treatment infrastructure, which have economic lifetimes of 15 to 25 years (mechanical and electrical treatment plant systems and pumping stations) to 60 to 70 years (concrete structures) (U.S. EPA, 2002).

Our ROI estimates are based only on the future (2015 onward) conservation and restoration interventions. While the existing interventions (39 ha restoration; 55 ha conservation) implemented in 2014 were selected using the sediment prioritization and payment determination calculus described above, they did not take into account predicted land use or cover change. Both our hydrologic and cost impacts therefore ignore those early interventions.

Results

CAMBORIÚ PWS PROGRAM ROI

Our analysis indicates that if analyzed purely as a sediment control measure for the municipal water supply, the Camboriú PWS program has an ROI<1 over a 30-year time horizon. This is true both for the program overall (all costs counted) as well as for EMASA in particular (Table 2), and holds true even if the reduced peak season water losses that result from reduced sediment concentrations in intake water had been used to reduce the size of the recent capacity expansion of the water treatment plant (see Scenario 2 in Table 2). Given the more front-loaded time profile of costs relative to benefits, this outcome is however sensitive to the time horizon of the study.

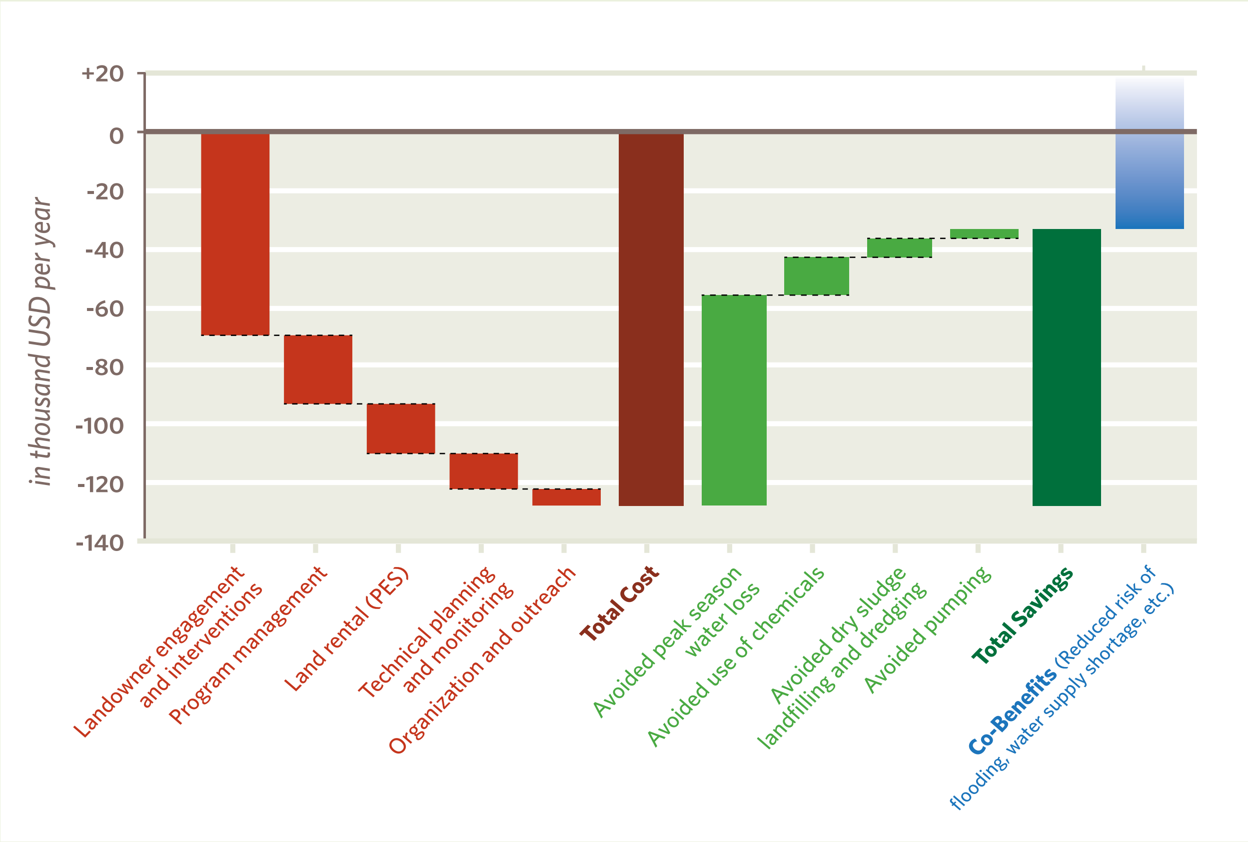

Figure 10: Comparison and composition of annual costs and benefits of the Camboriú PWS program, amortized over 30 years using Brazil's 3.85% social discount rate. Note: The value of co-benefits was not quantified in this analysis

Table 2: Estimated ROI metrics of the Camboriú PWS program as a sediment control measure for municipal water supply, 2015 to 2045

ROI for	Scenario	mg TSS per liter per million USD[1]	kg TSS per USD	B/C
Program overall	1	2.1	1.70	0.59
	2	2.2	1.78	0.63
EMASA	1	2.8	2.24	0.77
	2	3.0	2.39	0.83

Note: All dollar values in 2014 USD present values using a discount rate of 3.85 percent per year.

[1]Average reduction in TSS concentrations during the 30-year period per 1 million USD (present value) invested during 30 years.

Note that only benefits associated with sediment reduction are reflected in these ROI metrics. The biodiversity conservation, peak-season water supply risk and flood risk reduction values produced by the program are treated as co-benefits the quantification of which lies beyond the scope of this study.

COMPARISON WITH OTHER WATERSHED CONSERVATION BENEFIT ESTIMATES

Our benefit estimates are in good agreement with the few estimates of the impact of TSS on municipal drinking water treatment costs reported in other studies. According to our analysis, once the interventions have developed their full functionality with respect to TSS reduction, which is assumed to occur in 2032, the resulting 14 percent reduction in TSS concentrations in intake water will reduce total treatment costs of the water plant (0.21 USD per m^3 water output; EMASA data) by nearly 4 percent.

For comparison, McDonald and Shemie (2014) report that in their sample of more than 100 cities in the United States relying primarily on surface water sources, a 10 percent reduction in sediment concentration on average yields a 2.6 percent reduction in treatment plant operation and maintenance (O&M) costs (excluding pumping, distribution infrastructure O&M costs and reservoir dredging costs).

Similarly, using calibrated OTTER models of four water treatment plants, Grantley et al. (2003) estimate that a 25 percent decrease in TSS and a 15 percent decrease in total organic content (TOC) can reduce treatment costs by 5 percent, where included treatment costs comprise chemical usage, residuals disposal and power consumption of wastewater pumpage. Both McDonald and Shemie's (2014) and Grantley et al.'s (2003) estimates are in line with ours.

Factors that Influence ROI

While our analysis indicates that the Camboriú PWS program has an ROI <1 as a sediment control measure over a 30-year time horizon, this does not necessarily mean that the program does not make economic sense for its supporters overall or business sense for EMASA in particular.

In fact, our ROI estimates are sensitive to several key assumptions. These include:

1. the time horizon and discount rate used in the analysis;
2. the omission of co-benefits produced by the program, including biodiversity conservation and peak-season water supply risk and flood risk reduction. These represent positive externalities that affect the overall economic case for the program and, to the extent that they can be internalized, the business case of watershed conservation for EMASA vis-à-vis grey infrastructure sediment control alternatives;
3. the scale of intervention;
4. a conservative bias in our benefit estimates from sediment reduction;
5. full accounting for transaction costs

1. TIME HORIZON AND DISCOUNT RATE USED IN THE ANALYSIS

As a result of the time needed for restoration interventions to become fully effective in controlling sediment export as well as the spreading out of intervention implementation over eight years, annual benefits from sediment reduction initially are low but show a sharp increase over time, from an estimated less than USD 9,000 in 2015 (from prevented forest conversion) to USD 313,000 in 2045 (all current year undiscounted values at 2014 USD). Annual benefits from sediment reduction are projected to exceed annual costs in 2024 (Figure 11). In the preceding years, the program generates net costs due to front-loaded design, coordination and implementation costs and high transaction costs associated with program rollout.

Because of the inverted time profile of costs and benefits, longer time horizons will increase the ROI because more years are included in which the program generates net benefits. For example, the ROI for EMASA will exceed 1 if the time horizon is extended by 14 years (from 2045 to 2059) or more.

Similarly, because of the time profiles of costs and benefits, higher discount rates will lower the ROI while lower rates will increase it. If EMASA were a private corporation rather than a public entity, its discount rate would be based on its cost of capital or its rate of return from competing investments, which likely would exceed the rate used in our analysis.

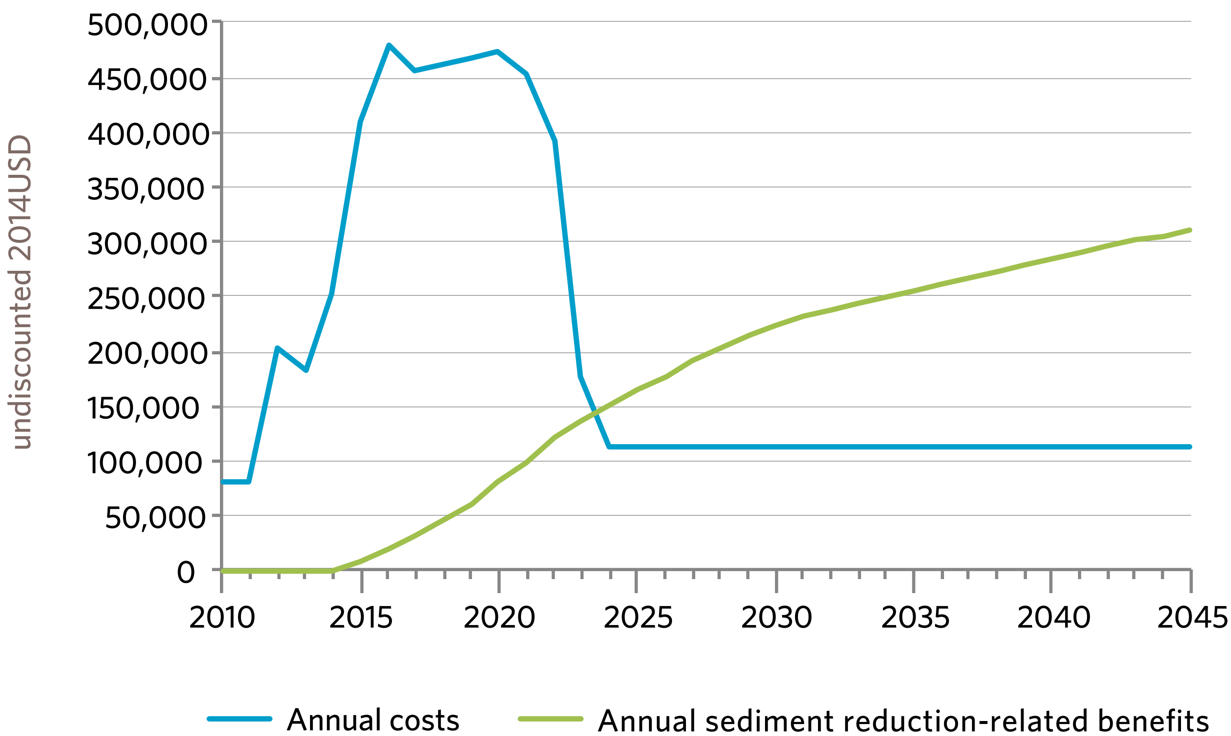

Figure 11: Time profile of annual costs and sediment reduction benefits

In our analysis, we target interventions based on the predicted likelihood of conversion of a site by a particular future point in time (2025 in our case). This can result in not protecting sites that over longer time frames may produce more of the target outcome (TSS reduction) but that are predicted to be converted only at a later point in time and therefore did not make it into the intervention portfolio. Changing the time horizon of the analysis thus may affect the "optimal" intervention portfolio. While this is not surprising, it does make the choice of time horizon very important, and may justify including this parameter in a formal sensitivity analysis.

2. CO-BENEFITS

The Camboriú PWS program produces important co-benefits in addition to reducing sediment concentrations in municipal intake water. Because of these additional benefits, the program's overall ROI is likely to substantially exceed its ROI in sediment control. Such divergence between the broader economic case and the specific business case for a particular objective or investor is, of course, neither surprising nor unique to the Camboriú program, but it does highlight the importance of careful selection of the scope of ROI analyses and the interpretation of their results.

Besides biodiversity conservation, two important co-benefits of the Camboriú program that are likely to drive a wedge between the broader economic and the specific business case are (1) reduced risk of flooding and (2) reduced risk of water supply interruptions.

By 2022, the Camboriú PWS program will increase forest cover equivalent to 5 percent of the watershed upstream of the EMASA intake. This is expected to increase infiltration and result in reduced surface runoff and river discharge during storm events, and potentially increased discharge during low precipitation periods. Our SWAT model predicts modeled PWS interventions to reduce peak flow levels (>12 cm at the EMASA intake monitoring point) by more than 3 percent on average and increase low flow levels (<2 cm at the EMASA intake) by 0.4 percent on average. While these impacts are small in relative terms, they nevertheless are likely to carry economic value.

Clearly, risk reduction is a strong reason for diversified investments in water infrastructure, including catchment restoration. Although it is impossible to say without detailed analysis how large the value of reduced risks of supply deficits and flooding is relative to the value of the reductions in sediment treatment plant operation and capital costs quantified in this analysis, evidence from other studies certainly suggests that it can be substantial. While the value of supply or flood risk reduction is strongly context-dependent, even if those values in Camboriú were one order of magnitude smaller than those reported for other cities in Brazil and elsewhere in South America (e.g., Zapata et al., 2012; Machado et al., 2014; Fuks and Chatterjee, 2008), they would dwarf the value associated with sediment reductions in the municipal water supply.

The importance of reduced supply and flood risks may be particularly high in the case of Balneario Camboriú, because of the importance of the tourism and real estate sectors to the municipality's economy and the presence of nearby substitute beach destinations along southern Brazil's Atlantic coast.

Those supply and flood risk reduction benefits accrue to local businesses, residents and visitors, directly or via reduced municipal spending on flood damages or grey water storage infrastructure. Thus, cost sharing among PWS program beneficiaries clearly is justified. This could be achieved by incorporating the cost of watershed maintenance into water user fees or, alternatively, levying a watershed conservation fee on high-season visitors based on the rationale that most of the benefits of reduced supply interruption risk accrue to visitors during the high season when demand is stressed and tourists account for three-quarters of the combined population of the two municipalities.

Incorporating a watershed conservation fee of only USD 0.005 per cubic meter water use — less than 0.4 percent of the current average rate paid by municipal water customers, or USD 1.25 for the average household per year — or of USD 0.15 per high-season visitor would lift the ROI of the program above 1. A surcharge of that amount would be justified as long as the combined value of the reduced risk of flooding and water supply shortages presently is at least USD 88,000 per year [USD 142,000 per year (undiscounted) on average during 2015 to 2045].

3. SCALE OF INTERVENTIONS

Our analysis assumes that the current rate of restoration and conservation interventions continues through 2022, when the current phase of landowner enrollment ends. This results in more than 50 ha of high sediment loading areas that remain unaddressed (see red areas in right panel in Figure 7). Because transaction costs account for a high share of total program costs and because many of these transaction costs are either independent of total intervention extent or increase less than proportionally with that extent, the ROI of the program could be improved by expanding interventions to those remaining high-loading areas. For example, increasing total conservation and restoration areas by 10 percent (64 ha) compared to our analysis would increase total program costs by approximately 6 percent, but benefits would increase by nearly 10 percent.

4. CONSERVATIVE BENEFITS ESTIMATES

We do not include the value of reduced water treatment plant maintenance costs as a result of reduced sediment loads and reduced processing during off-peak months, for either the existing treatment plant or the expansion unit. This omission is likely to impart a negative (low) bias in our benefit estimates, and, consequently, will bias our ROI estimates downward. Maintenance costs may not be proportionally related to TSS concentrations and water throughput, and their estimation is beyond the scope of this study. If the estimated average annual 11 percent reduction in TSS concentrations in intake water during 2015 to 2045 is sufficient to increase the lifetime of plant equipment, our benefits estimates may have a substantial downward bias.

5. FULL ACCOUNTING FOR TRANSACTION COSTS

In the Camboriú PWS program, transaction costs (TAC) for organization and outreach during the program planning stage; legal and hydrologic studies; hydrologic and compliance monitoring; landowner engagement and project administration account for over 50 percent of total program costs. Few literature estimates of PES or PWS programs include TAC, and the ones that do likely underestimate the latter (Finney, 2015). The large share of TAC in the Camboriú PWS program is explained in part by pre-project feasibility studies; assembly of, and coordination among, a technically strong and diverse group of program partners; and a substantial investment in efficient intervention targeting and impact and compliance monitoring, all of which enhance the performance and sustainability of the program. All of these activities imply labor and other costs incurred by program partners, which we attempted to realistically account for.


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Conclusion

Our analysis indicates that the Camboriú PWS program is expected to be an effective tool for reducing TSS concentrations at the municipal drinking water treatment plant intake. Through fairly sophisticated targeting of restoration and conservation interventions based on both costs and benefits, focusing on lands that would contribute the highest sediment loads, the program will enroll nearly 5 percent of lands in the watershed upstream of the treatment plant yet will reduce TSS concentrations at the intake by an estimated 14 percent once interventions attain their full functionality.

Yet, despite this effective targeting, those large impacts also come at a substantial cost. These result from both high transaction costs and a large share of active restoration. The transaction costs include:

- extensive land use and land cover change and hydrologic analyses aimed at ensuring additionality of impacts;
- the assembly of, and coordination among, a diverse group of program partners, and collaborative program management;
- extensive landowner engagement activities and interventions tailored to individual sites;
- outreach activities aimed at creating public awareness of the program and its purpose, both among the general public and public decision makers, to ensure political support;
- ongoing hydrologic monitoring to demonstrate program impacts.

We consider all of these activities necessary for ensuring the effectiveness, cost-effectiveness and sustainability of the program. It is important to note that while the cost of the Camboriú PWS program may appear high relative to those reported for other PWS programs, this is caused in large part by the failure of most analyses to fully account for transaction costs. In other words, we regard our cost estimates of the Camboriú program as representative of the actual costs faced by similar programs elsewhere in the Atlantic Forest that aim to produce similarly extensive land cover changes and achieve sustainability.

Unlike the heavily front-loaded program costs, the benefits the treatment plant receives from reduced TSS concentrations grow over time with treatment plant output and development of full sediment control functionality of the interventions.

The combination of high, front-loaded costs and benefits that start out small and steadily grow over time results in a 30-year ROI<1 of the program for sediment control. However, the ROI for the municipal drinking water treatment plant exceeds 1 for time frames longer than 43 years.

Yet the PWS program generates important additional benefits of high concern to local residents and businesses in the form of flood attenuation and a reduction in the risk of water supply shortages during the tourist season. With tourism being a crucial part of the local economy, reductions in the risk of flooding and supply shortages likely carry large total economic value for those beneficiary groups. Thus, there is a strong case for cost sharing among program beneficiaries. Such cost sharing would align the business case for the water treatment company and the broader economic case for catchment protection. Incorporating the cost of watershed protection into water use fees — an amount equivalent to a mere 0.4 percent of current average user charges — or levying a conservation fee of USD 0.15 per high-season visitor both represent low TAC vehicles for such cost sharing. Recognizing the broadly distributed benefits of watershed protection, the incorporation of watershed protection costs into water user fees presently is being discussed in Camboriú among PWS program partners, local governments, and the state's water and sanitation regulatory agency.

While the Camboriú PWS program displaces a portion of the conventional "grey" water treatment and could even have avoided a portion of the expansion of treatment infrastructure currently under way, it could not displace all grey treatment. Thus, the program is an example of how natural infrastructure can serve as a cost-effective complement to grey infrastructure.

We believe our findings are indicative of the ROI of many other similar catchment protection projects. Many municipal drinking water treatment plants in Brazil and elsewhere are of the same type found in Camboriú. As a result, programs that produce comparably sized TSS reductions in watersheds without storage reservoirs should generate similar values in the form of avoided treatment costs. For plants employing different treatment types (including sludge water recovery), benefits may differ. The transferability of our benefit estimates to a given catchment and plant thus hinges more on the size of the watershed and the portion of catchment discharge captured by the treatment plant: The larger the watershed, the larger the scope of interventions needed (McDonald and Shemie, 2014); the smaller the share of discharge captured by the plant, the lower the benefits relative to the costs.

Irrespective of whether or not our case study findings are easily transferable to similar programs elsewhere, the findings of our case study do lead to a clear recommendation: Payment for watershed services programs should strive to assess program impacts on high-value "ancillary" ecosystem services received by parties other than the intended core target beneficiaries of the PWS program. Credible quantification of the economic value of the benefits associated with those third-party service gains will facilitate internalization of those values into program financing via cost-sharing. This will improve its business case or may even help establish the business case in the first place.

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References

Alcott, E., Ashton, M.S., and Gentry, B.S. (eds). (2013). *Natural and Engineered Solutions for Drinking Water Supplies: Lessons from the Northeastern United States and Directions for Global Watershed Management.* Boca Raton: CRC Press. 303 pp.

Anderson, C.J., and Lockaby, B.G. (2011). The effectiveness of forestry best management practices for sediment control in the Southeastern United States: A literature review. *Southern Journal of Applied Forestry* 35(4):170-177.

Arnold, J.G., Srinivasan, R., Muttiah, R.S., and Williams, J.R. (1998). Large area hydrologic modeling and assessment part I: Model development1 *JAWRA Journal of the American Water Resources Association* 34(1):73–89.

Arrow, K., Cropper, M., Gollier, C., Groom, B., Heal, G., Newell, R., Nordhaus, W., Pindyck, R., Pizer, W., Portney, P., Sterner, T., Tol, R.S.J., and Weitzman, M. (2013). Determining benefits and costs for future generations. *Science* 341:349-350.

Balmford, A., Gaston, K.J., Blyth, S., James, A., and Kapos, V. (2003). Global variation in terrestrial conservation costs, conservation benefits, and unmet conservation needs. *Proceedings of the National Academy of Sciences* 100(3):1046-1050.

Banco Central do Brasil. (2015). Economic Indicators. Chapter III – Financial and capital markets: Effective interest rates. 7 October 2015. Online at http://www.bcb.gov.br/?INDICATORS. Last accessed 13 October 2015.

Banco Nacional de Desenvolvimento Econômico e Social. (2013). *Programa para Redução da Emissão de Gases de Efeito Estufa na Agricultura – Programa ABC* (BNDES, Rio de Janeiro, 2013; http://www.bndes.gov.br/SiteBNDES/bndes/bndes_pt/Institucional/Apoio_ Financeiro/Programas_e_Fundos/abc.html).

Banks-Leite, C., Pardini, R., Tambosi, L.R., Pearse, W.D., Bueno, A.A., Bruscagin, R.T., Condez, T.H., Dixo, M., Igari, A.T., Martensen, A.C., and Metzger, J.P. (2014). Using ecological thresholds to evaluate the costs and benefits of set-asides in a biodiversity hotspot. *Science* 345 (6200):1041-1045.

Baptista, S.R. (2008). Metropolitanization and forest recovery in southern Brazil: A multiscale analysis of the Florianópolis city-region, Santa Catarina State, 1970 to 2005. *Ecology and Society* 13(2):5. [online] URL: http://www.ecologyandsociety.org/vol13/iss2/art5

Baptista, S.R., and Rudel, T.K. (2006). A re-emerging Atlantic forest? Urbanization, industrialization and the forest transition in Santa Catarina, southern Brazil. *Environmental Conservation* 33:195–202.

Bennett, G., and Carrol, N. (2014). Gaining Depth: State of Watershed Investment 2014. Washington, D.C.: Forest Trends Ecosystem Marketplace. 127 pp

Bennett, E.M., Peterson, G.D., and Gordon, L.J. (2009). Understanding relationship among multiple ecosystem services. *Ecology Letters* 12(12):1394–1404.

Betrie, G.D., Mohamed, Y.A., van Griensven, A., and Srinivasan, R. (2011). Sediment management modelling in the Blue Nile Basin using SWAT model. *Hydrological Earth Systems Science* 15(3):807–818, doi:10.5194/hess-15-807-2011.

Bishop, R.C. (1978). Endangered species and uncertainty: The economics of a safe minimum standard. *American Journal of Agricultural Economics* 60(1):10-18.

Blackman, A. (2013). Evaluating forest conservation policies in developing countries using remote sensing data: An introduction and practical guide. *Forest Policy and Economics* 34:1-16.

Boyd, J., Epanchin-Niell, R., and Siikamäki, J. (2012). Conservation Return on Investment Analysis: A Review of Results, Methods, and New Directions. Resources for the Future Discussion paper 12-01.

Boyd, J., and Banzhaf, S. (2007). What are ecosystem services? The need for standardized environmental accounting units. *Ecological Economics* 63:616-626.

Brown, T.C., Bergstrom, J.C., and Loomis, J.B. (2007). Defining, valuing and providing ecosystem goods and services. *Natural Resources Journal* 47(2):329-376.

Das, S., and Vincent, J.R. (2009). Mangroves protected villages and reduced death toll during Indian super cyclone. *Proceedings of the National Academy of Sciences USA* 106(18):7357–7360.

Duke, J.M., Dundas, S.J., Johnston, R.J., and Messer, K.D. (2014). Prioritizing payment for environmental services: Using nonmarket benefits and costs for optimal selection. *Ecological Economics* 105:319–329.

Fenichel, E.P., Kotchen, M.J., and Addicott, E. (2016). The representative agent must die: Using demographics to inform social discount rates. Paper presented at the European Association of Environmental and Resource Economists (EAERE) 22nd Annual Conference, 22-25 June 2016, Zurich, Switzerland.

Ferrario, F., Beck, M.W., Storlazzi, C., Micheli, F., Shepard, C., and Airoldi, L. (2014). The effectiveness of coral reefs for coastal hazard risk reduction. *Nature Communications* 5:3794. doi: 10.1038/ncomms4794

Ferraro, P.J. (2009). Counterfactual Thinking and Impact Evaluation in Environmental Policy. In M. Birnbaum & P. Mickwitz (Eds.), *Environmental program and policy evaluation. New Directions for Evaluation* 122:75–84.

Ferraro, P.J., Lawlor, K., Mullan, K.L., and Pattanayak, S.K. (2012). Forest figures: Ecosystem services valuation and policy evaluation in developing countries. *Review of Environmental Economics and Policy* 6(1):20-44.

Ferraro, P.J., and Pattanayak, S.K. (2006). Money for nothing? A call for empirical evaluation of biodiversity conservation investments. *PLoS Biology* 4(4):e105.

Ferreira, J.C., Silva, L., and Polette, M. (2009). The coastal artificialization process: Impacts and challenges for the sustainable management of the coastal cities of Santa Catarina (Brazil). *Journal of Coastal Research* SI 56:1209-1213.

Ferretti-Gallon, K., and Busch, J. (2014). What drives deforestation and what stops it? A meta-analysis of spatially explicit econometric studies. CGD Working Paper 361. Washington, D.C.: Center for Global Development.

Finney, C. (2015). Comment on "Using ecological thresholds to evaluate the costs and benefits of set-asides in a biodiversity hotspot." *Science* 347(6223):731.

Fisher, J.R.B., Acosta, E., Dennedy-Frank, P.J., Boucher, T., Kroeger, T., and Giberti, S. (2017). The impact of satellite imagery's spatial resolution on land use classification and modeled water quality. Manuscript in preparation.

Fuks, M., and Chatterjee, L. (2008). Estimating the willingness to pay for a flood control project in Brazil using the contingent valuation method. *Journal of Urban Planning and Development* 134(1). DOI: http://dx.doi.org/10.1061/(ASCE)0733-9488(2008)134:1(42) - See more at: http://ascelibrary.org/doi/full/10.1061/(ASCE)0733-9488(2008)134:1(42)#sthash.aqggtY7U.dpuf

Furniss, M.J., Staab, B.P., Hazelhurst, S., Clifton, C.F., Roby, K.B., Ilhadrt, B.L., Larry, E.B., Todd, A.H., Reid, L.M., Hines, S.J., Bennett, K.A., Luce, C.H., and Edwards, P.J. (2010). Water, climate change, and forests: Watershed stewardship for a changing climate. Gen. Tech. Rep. PNW-GTR-812. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 75pp.

Gassman, P.W., Reyes, M.R., Green, C.H., and Arnold, J.G. (2007). The Soil and Water Assessment Tool: Historical Development, Applications, and Future Research Directions. *Transactions of the American Society of Agricultural and Biological Engineers* 50(4):1211-1250.

Gollier, C., and Treich, N. (2003). Decision-making under scientific uncertainty: The economics of the precautionary principle. *The Journal of Risk and Uncertainty* 27(1):77–103.

Grantley, W.P., Becker, W.C., Head, R., and O'Melia, C.R. (2003). Impacts of Major Point and Non-Point Sources on Raw Water Treatability. AWWA Research Foundation Report No. 90959F. Denver, CO: AWWA Research Foundation.

Griffiths, C., Klemick, H., Massey, M., Moore, C., Newbold, S., Simpson, D., Walsh, P., and Wheeler, W. (2012). US Environmental Protection Agency valuation of surface water quality improvements. *Review of Environmental Economics and Policy* 6(1):130–146.

Guedes, F.B., and Seehusen, S.E. (2011). Pagamentos por Serviços Ambientais na Mata Atlânica: Lições aprendidas e desafíos. Ministério do Meio Ambiente, Brasil.

Guimarães, R.Z., Lignau, C., Rizzi, N.R., Schechi, R.G., and Bianchi, R.C. (2011). *Espacialização da perda de solo por erosão laminar na micro-bacia do rio Campinas*, Joinville, Santa Catarina. Ra´E Ga 23:32-64.

Gumbert, A.A., Higgins, S., and Agouridis, C. (2009). Riparian buffers: A livestock best management practice for protecting water quality. ID-175. Lexington, KY: University of Kentucky Cooperative Extension Service.

Guo, Z., Li, Y., Xiao, X., Zhang, L., and Gan, Y. (2007). Hydroelectricity production and forest conservation in watersheds. *Ecological Applications* 17(6):1557–1562.

Holmes Jr., R.R., Terrio, P.J., Harris, M.A., and Mills, P.C. (2001). *Introduction to field methods for hydrologic and environmental studies.* Open-File Report, USGS Numbered Series.

Instituto Brasileiro de Geografia e Estatística. (2016a). Balneário Camboriú. http://www.cidades.ibge.gov.br/v3/cidades/municipio/4202008

Instituto Brasileiro de Geografia e Estatística. (2016b). Camboriú. http://www.cidades.ibge.gov.br/v3/cidades/municipio/4203204

Johnston, R.J., and Russell, M. (2011). An operational structure for clarity in ecosystem service values. *Ecological Economics* 70:2243–2249.

Joly, C.A., Metzger, J.P., and Tabarelli, M. (2014). Experiences from the Brazilian Atlantic Forest: ecological findings and conservation initiatives. *New Phytologist* 204(3) DOI:10.1111/nph.12989

Keeler, B.L., Polasky, S., Brauman, K.A., Johnson, K.A., Finlay, J.C., O'Neille, A., Kovacs, K., and Dalzell, B. (2012). Linking water quality and well-being for improved assessment and valuation of ecosystem services. *Proceedings of the National Academy of Sciences USA* 109(45):18618-18624.

Kroeger, T. (2013). The quest for the "optimal" payment for environmental services program: Ambition meets reality, with useful lessons. *Forest Policy and Economics* 37:65-74.

Kroeger, T., and Guannel, G. (2014). Fishery Enhancement and Coastal Protection Services Provided by Two Restored Gulf of Mexico Oyster Reefs. Pp. 334-358 in: K. Ninan (ed.), *Valuing Ecosystem Services-Methodological Issues and Case Studies*. Cheltenham: Edward Elgar. 464 pp.

Kroeger, T., Escobedo, F.J., Hernandez, J.L., Varela, S., Delphin, S., Fisher, J.R.B., and Waldron, J. (2014). Reforestation as a novel abatement and compliance measure for ground-level ozone. *Proceedings of the National Academy of Sciences USA* 111(40):E4204-4213.

Kundzewicz, Z.W., Mata, L.J., Arnell, N.W., Döll, P., Jimenez, B., Miller, K., Oki, T., Sen, Z., and Shiklomanov, I. (2008). The implications of projected climate change for freshwater resources and their management. *Hydrological Sciences Journal* 53(1):3-10.

Landers, D.H., and Nahlik, A.M. (2013). Final Ecosystem Goods and Services Classification System (FEGS-CS). EPA/600/R-13/ORD-004914. Washington, D.C.: U.S. Environmental Protection Agency, Office of Research and Development.

Lohmann, G., Glauber, S., and Allis, T. (2011). 'Los Hermanos' Visiting the South Region of Brazil: A Comparison Between Drive Tourists and Coach Tourists from Argentina, Paraguay and Uruguay. Pp. 49-60 in B. Prideaux and D. Carson (Eds.) *Drive Tourism: Trends and Emerging Markets.* New York: Routledge.

Machado, F.H., Silva, L.F., Dupas, F.A., Mattedi, A.P., and Vergara, F.E. (2014). Economic assessment of urban watersheds: developing mechanisms for environmental protection of the Feijão river, São Carlos - SP, Brazil. *Brazilian Journal of Biology* 74(3):677-684.

McDonald, R.I., Weber, K.F., Padowski, J., Boucher, T., and Shemie, D. (2016). Estimating watershed degradation over the last century and its impact on water-treatment costs for the world's large cities. *PNAS* 113(32):9117-9122.

McDonald, R.I., and Shemie, D. (2014). Urban Water Blueprint: Mapping conservation solutions to the global water challenge. Arlington, VA: The Nature Conservancy.

McDonald, R.I., Weber, K., Padowski, J., Flörke, M., Schneider, C., Green, P.A., Gleeson, T., Eckman, S., Lehner, B., Balk, D., Boucher, T., Grill, G., and Montgomery, M. (2014). Water on an urban planet: Urbanization and the reach of urban water infrastructure. *Global Environmental Change* 27:96-105.

McDonald, R.I., Green, P., Balk, D., Fekete, B., Revenga, C., Todd, M., and Montgomery, M. (2011). Urban growth, climate change, and freshwater availability. P*roceedings of the National Academy of Sciences USA* 108:6312–6317.

Mills, K., Dent, L., and Cornell, J. (2007). Rapid survey of road conditions to determine environmental effects and maintenance needs. *Transportation Research Record: Journal of the Transportation Research Board* 1989(1):89-97.

Milly, P.C.D., Dunne, K.A., and Vecchia, A.V. (2005). Global pattern of trends in streamflow and water availability in a changing climate. *Nature* 438:347-350.

Moriasi, D.N., Arnold, J.G., Van Liew, M.W., Bingner, R.L., Harmel, R.D., and Veith, T.L. (2007). Model evaluation guidelines for systematic quantification of accuracy in watershed simulations. *Transactions ASABE* 50(3):885–900.

National Research Council. (2005). *Valuing Ecosystem Services: Toward Better Environmental Decision-Making*. Washington, D.C.: National Academies Press.

Opperman, J.J., Galloway, G.E., Fargione, J., Mount, J.F., Richter, B.D., and Secchi, S. (2009). Sustainable floodplains through large-scale reconnection to rivers. *Science* 326: 1487-1488.

Pérez-Vega, A., Mas, J-F., and Ligmann-Zielinska, A. (2012). Comparing two approaches to land use/cover change modeling and their implications for the assessment of biodiversity loss in a deciduous tropical forest. *Environmental Modelling and Software* 29:11-23.

Soares-Filho, B., Rajão, R., Macedo, M., Carneiro, A., Costa, W., Coe, M., Rodrigues, H., and Alencar, A. (2014). Cracking Brazil's Forest Code. *Science* 344:363-364.

Soares-Filho, B., Alencar, A., Nepstad, D., Cerqueira, G., Diaz, M.C.V., Rivero, S., Solórzano, L., and Voll, E. (2004). Simulating the response of land-cover changes to road paving and governance along a major Amazon highway: the Santarém-Cuiabá corridor. *Global Change Biology* 10:745–764.

Spalding, M.D., Ruffo, S., Lacambra, C., Meliane, I., Hale, L.Z., Shepard, C.C., and Beck, M.W. (2013). The role of ecosystems in coastal protection: adapting to climate change and coastal hazards. *Ocean & Coastal Management* 90:50-57.

Sterner, T., and Persson, U.M. (2008). An Even Sterner review: Introducing relative prices into the discounting debate. *Review of Environmental Economics and Policy* 2(1):61-76.

Tallis, H., and Polasky, S. (2009). Mapping and valuing ecosystem services as an approach for conservation and natural resources management. *The Year in Ecology and Conservation Biology: Annals of the New York Academy of Science* 1162:265-283.

Teixeira, A.M.G., Soares-Filho, B.S., Freitas, S.R., and Metzger, J.P. (2009). Modeling landscape dynamics in an Atlantic Rainforest region: Implications for conservation. *Forest Ecology and Management* 257:1219-1230.

Temmerman, S., Meire, P., Bouma, T.J., Herman, P.M.J., Ysebaert, T., and De Vriend, H.J. (2013). Ecosystem-based coastal defence in the face of global change. *Nature* 504:79–83.

U.S. Environmental Protection Agency (EPA). (2002). The clean water and drinking-water infrastructure gap analysis. Office of Water (4606M). EPA-816-R-02-020. September.

Vörösmarty, C.J., Green, P., Salisbury, J., and Lammers, R.B. (2000). Global Water Resources: Vulnerability from Climate Change and Population Growth. *Science* 289:284-288.

Wilson, M.A., and Carpenter, S.R. (1999). Economic valuation of freshwater ecosystem services in the United States: 1971-1997. *Ecological Applications* 9:772-783.

Wunder, S. (2013). When payments for environmental services will work for conservation. *Conservation Letters* 6(4):230-237.

Zapata, S.D., Benavides, H.M, Carpio, C.E., and Willis, D.B. (2012). The economic value of basin protection to improve the quality and reliability of potable water supply: Some Evidence from Ecuador. *Water Policy* 14(1):1-13.

Appendix

A1. SEDIMENT IMPACTS ON MUNICIPAL WATER TREATMENT PLANT

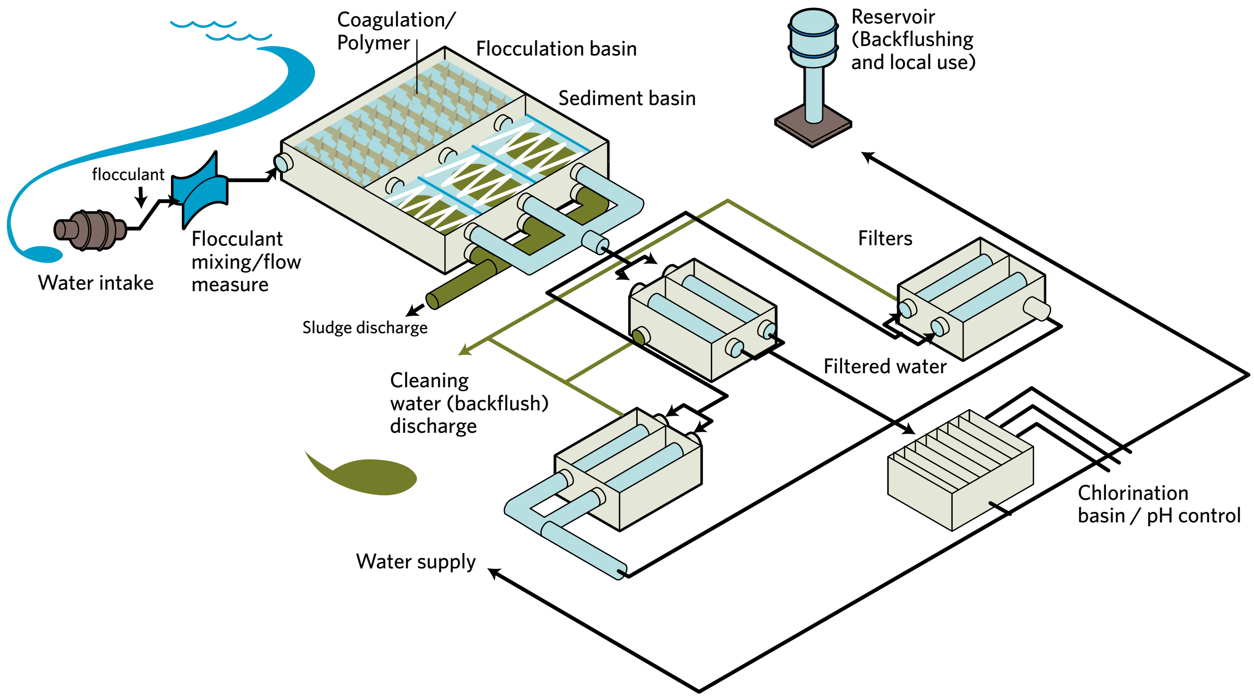


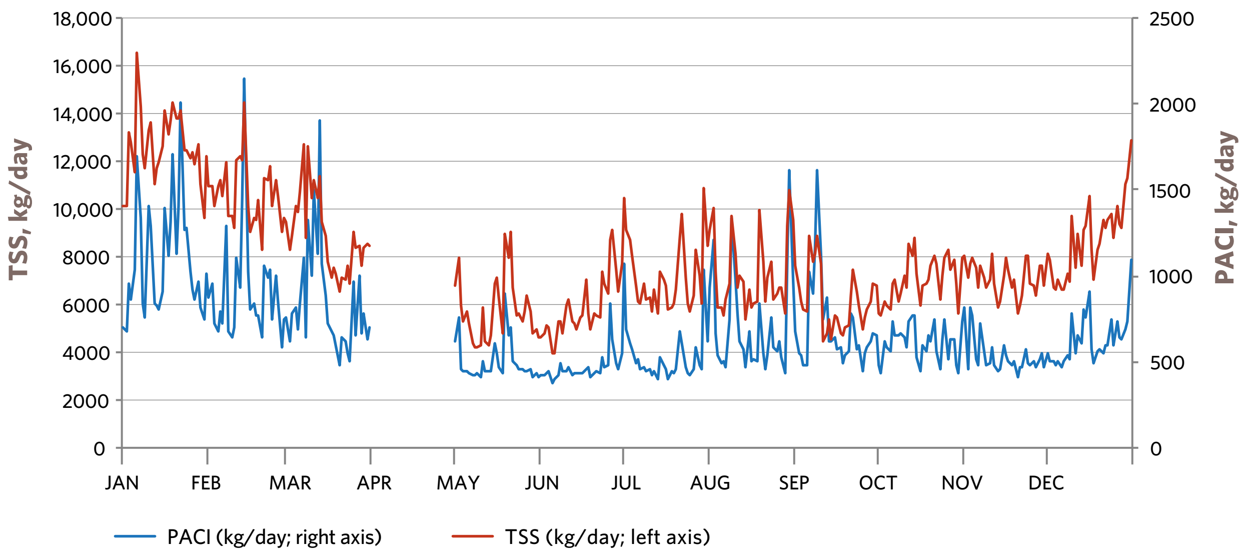
Figure A1: Drinking water treatment plant processes affected by sediment in intake water

DREDGING — Higher sediment loads require more frequent intake channel dredging. The river at the intake and the intake channel itself currently are dredged bi-annually, with 2,000 to 2,500 m^3 of sediment removed in each dredging. The dredge material is composed of heavier sediment fraction that moves along the base of the stream channel. We assume that the PWS interventions reduce that heavier fraction by the same proportion as TSS.

PUMPING — Pumping of water from the river outtake to the treatment plant requires 0.245 kWh/m^3 on average; pumping within the plant requires 0.345 kWh/m^3 on average.

CHEMICALS USE — The chemicals used to remove TSS comprise aluminum polychloride (PACI), a coagulate added to the water to achieve flocculation, and a polymer added as an auxiliary flocculent in the flocculation basins under high inflow conditions. PACI and polymer application are highly correlated ($R^2 = 0.95$; $p = 0.000$) with TSS concentrations in intake water (Figure A1.5).

Figure A1.5: Daily TTS load and PACI application in EMASA treatment plant in 2011



SLUDGE PRODUCTION — The floccus (a coagulate of TSS, PACI and polymer) settles in the sedimentation basins and is regularly discharged as sludge, with the frequency depending on TSS loads and quantity of water processed. EMASA reports production of 923 m^3 of sludge per day under normal operating conditions and intake levels (0.64 m^3/s). The sludge then is pumped to immediately outside the plant, where it is left to dry and then trucked to a landfill. Sludge transport records indicate that an average of 9.24 t of dried sludge material are landfilled per day.

SEDIMENT-RELATED WATER LOSS — A 2006 analysis of a single sludge sample of the plant revealed a total mass of dry solids of 7.24 g/l-1, equivalent to a sludge water content of 99.3 percent. Thus, each m^3 of sludge dry solids is associated with a loss of 137 m^3 of water. Given the reported average daily sludge production of 923 m^3, estimated average monthly water loss in sludge thus is 27,870 m^3, equivalent to 1.7 percent of inflow. Using May 2014 to August 2015 turbidity monitoring data at the water intake, the turbidity-TSS rating curve developed for the EMASA intake and the monitored daily water treatment plant inflow volumes during the same period, the plant receives an estimated average daily TSS load of 5.08 t. Given the average coagulant (PACI) application rate in the plant of 46.4 t per month, TSS accounts for an estimated 77 percent of the average total solids mass (6.60 t per day) entering the coagulation-flocculation-sedimentation treatment train. Thus, each m^3 TSS is associated with the loss of 178 m^3 water.

Table A1: Sediment-related water treatment plant unit costs and quantities

	Value	Unit	Quantity #
Pumping: from intake channel to treatment plant	0.08	USD/kWh	0.245 kWh/m^3 ‡
Pumping: within treatment plant	0.08	USD/kWh	0.345 kWh/m^3 ‡
Coagulate (PACl)	0.38	USD/kg	46,436 kg/month (25 mg/L)
Flocculent (polymer)	3.71	USD/kg	59 kg/month (0.03 mg/L)
Water lost in sludge	depends**		992.8 g/l sludge
Water lost in filter backwashing	depends**		766,500 m^3 /yr §
Peak season revenue forgone from water lost in sediment removal*	1.37	USD/m^3	389,107 m^3/peak season †
Treatment plant sludge disposal	18.75	USD/ton	9.24 t/day
Intake channel dredging ˆ	4.70	USD/m^3	1,250 m^3/yr

Notes: All data from EMASA. Converted from USD to US using the average 2014 BRL-USD exchange rate of 3.2 (from www.xe.com).

‡ At normal (design) operating rate of 0.64 m^3 /s (latest year-on-year operating rate is 0.69 m^3 /s).

* Forgone revenue from sale of water during peak season. Assumes marginal user charge is equivalent to estimated average charge across all municipal use of BRL 4.39 per m^3 (charge is per m^3 of water consumption including 0.8 m^3 of sewer discharge per m^3 consumption).

ˆ Collected for free by third party who sells the dredging material.

\# Average of reported quantities.

** Estimated value depends on whether loss occurs in peak (lost revenue) or off-peak season (no lost revenue).

§ Based on 350 m^3 water used per filter backflushing and six flushings per day. 67 percent and 34 percent of this loss are assumed to occur during the off-peak and peak seasons, respectively.

† Includes 133,607 m^3 lost in sediment basin sludge discharge and 255.500 m^3 lost in filter backwashing per peak season.

From the sedimentation basins, the water is pumped to the final sediment treatment stage, where it passes through two large gravity filters composed of layers of gravel, sand and activated charcoal that remove the majority of the remaining particles. These filters each are backwashed two to three times daily using already treated water. Higher polymer use leads to polymer buildup on the filters, necessitating more frequent cleaning. Each backwash cycle takes 30 to 45 minutes and requires at least 350 m^3 of treated water. The water used for filter backwashing is then discharged as wastewater. We assume that in the future given the plant expansion and fast-rising water demand, each final TSS filter will be backwashed on average three times per day using 350 m^3 per event, resulting in a total annual water loss for filter backwashing of an estimated 766,500 m^3, or 4.2 percent of total average annual 2008-2014 water intake.

Total sediment removal-related water losses thus sum to 5.9 percent of intake water. Treatment plant data for 2008 to 2014 indicate that total measured water outflow is 15.5 percent less than total raw water intake. The remainder of this difference is explained by abstraction of water ahead of the outflow monitoring point that is used for the filling of water trucks that supply neighboring Camboriú Municipality when the latter faces supply shortfalls, as well as by internal plant use and evaporation.

A2. HYDROLOGIC MODEL CALIBRATION

We calibrated SWAT to 2014 discharge and TSS concentration data using a split-sample calibration method.

For discharge, we calibrated SWAT with the in-sample calibration occurring from 1/1/2014 to 12/31/2014 and the out-of-sample test occurring from 1/1/2015 to 11/06/2015. First, the model was calibrated for discharge. At the EMASA station, the model achieved a Nash-Sutcliffe Efficiency (NS) of 0.71 for in-sample daily flows during calibration, and 0.53 for out-of-sample daily flows, qualifying the model as better than satisfactory according to Moriasi et al.'s (2007) monthly calibration criteria (Table A2). The NS was chosen as the calibration statistic because it strongly weights large flows, which in many locations drive the transport of the majority of sediment. Calibration statistics were calculated for both the Canoas and EMASA stations, and these two stations were weighted equally for the purpose of model calibration. A sensitivity analysis was performed to estimate the most sensitive parameters in the model before calibration. See Fisher et al. (2017) for the parameter values used in the calibration.

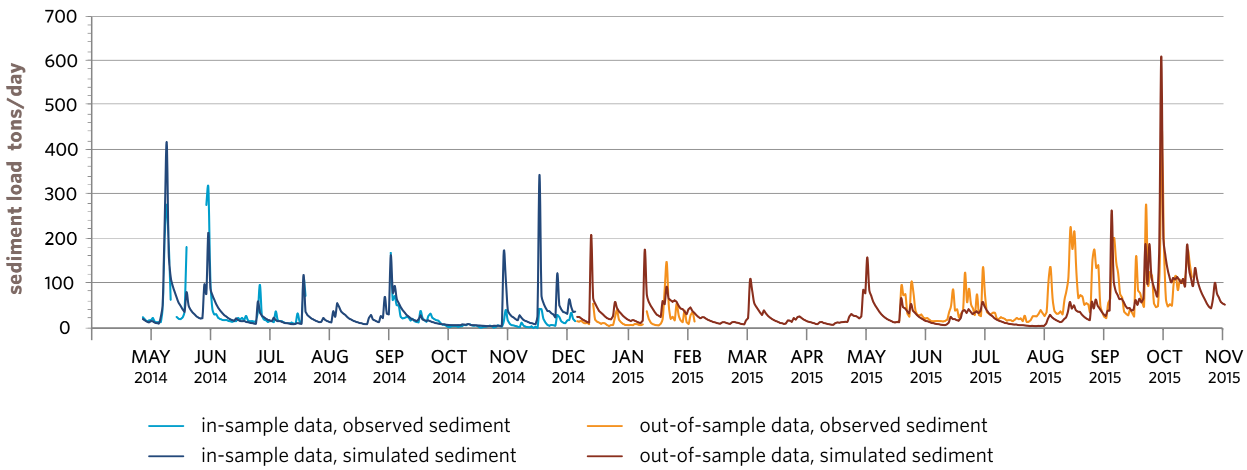
Table A2. Sediment and flow calibration statistics for in-sample calibration and out of sample validation data.

	In-sample data (1/1/2014-12/31/2014)				Out-of-sample data (1/1/2015-11/06/15)		
	NSE	PBIAS	R^2		NSE	PBIAS	R^2
flow (1 m)	0.71	-4.54	0.74		0.53	-6.17	0.81
sediment (1 m)	0.63	8.42	0.63		0.48	15.01	0.57

Source: Fisher et al. (2017)

After discharge, the SWAT model was calibrated for sediment load in the watershed at both the Canoas and EMASA stations at the daily timescale.

Figure A2: Comparison of observed (based on measured turbidity and turbidity-TSS concentration curve) and simulated TSS loads at the EMASA intake, 2014 to 2015.



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