Returns on investment in watershed conservation: Application of a best practices analytical framework to the Rio Camboriú Water Producer program, Santa Catarina, Brazil

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Abstract

Watershed management may have widespread potential to cost-effectively deliver hydrologic services. Mobilizing the needed investments requires credible assessments of how watershed conservation compares to conventional solutions on cost and effectiveness, utilizing an integrated analytical framework that links the bio-, litho-, hydro- and economic spheres and uses counterfactuals.

We apply such a framework to a payment for watershed services (PWS) program in Camboriú, Santa Catarina State, Brazil. Using 1m resolution satellite imagery, we assess recent land use and land cover (LULC) change and apply the Land Change Modeler tool to predict future LULC without the PWS program. We use current and predicted counterfactual LULC, site costs and a Soil and Water Assessment Tool model calibrated to the watershed to both target watershed interventions for sediment reduction and predict program impact on total suspended solids (TSS) concentrations at the municipal water intake—the principal program objective. Using local water treatment and PWS program costs, we estimate the return on investment (ROI; benefit/costs) of the program.

Program ROI exceeds 1 for the municipal water utility in year 44, well within common drinking water infrastructure planning horizons. Because some program costs are borne by third parties, over that same period, for overall (social) program ROI to exceed 1 requires delivery of very modest flood and supply risk reduction and biodiversity co-benefits, making co-benefits crucial for social program justification. Transaction costs account for half of total program costs, a result of large investments in efficient targeting and program sustainability. Co-benefits justify increased cost sharing with other beneficiaries, which would increase ROI for the utility, demonstrating the sensitivity of the business case for watershed conservation to its broader social-economic case and the ability to forge institutional arrangements to internalize third-party benefits.

Keywords: Integrated assessment model; watershed management; payments for watershed services; counterfactual; land use change modeling; transaction costs

1. Introduction

The use of "natural infrastructure"—ecosystems or their components—to complement or substitute conventional engineering-based solutions to environmental problems has been receiving widespread interest (Beck et al., 2018; Kroeger et al., 2014; Kroeger et al., 2018; Reguero et al., 2018; Temmerman et al., 2013). In particular, watershed conservation (i.e., protection of existing natural areas from conversion and improvement in land management practices) and restoration (re-establishment of natural vegetation on previously converted lands) have shown promise for improving water quality, flow regulation and flood control (Alcott et al., 2013; De Risi et al., 2018; Furniss et al., 2010; McDonald and Shemie, 2014; McDonald et al., 2016; Opperman et al., 2009).

Three economic rationales are commonly advanced for investing in natural infrastructure solutions: cost-effectiveness, co-benefits and the precautionary principle. Natural infrastructure is cost-effective in producing a specific target service or service bundle if it is at least cost-competitive with conventional engineering-based "grey" infrastructure (Reguero et al., 2018; Kroeger et al., 2014). Natural infrastructure generates co-benefits due to the additional ecosystem services it provides beyond a specific target service(s) (Bennett et al., 2009; Raudsepp-Hearne et al., 2010; Kreye et al., 2014) that competing grey infrastructure generally does not provide (Kroeger and Guannel, 2014; Spalding et al., 2013). Finally, the precautionary principle supports the preservation of the option value of natural systems 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 (Gollier and Treich, 2003; Randall, 1988). In the case of watersheds, the precautionary principle can support conservation and restoration based on the argument that more intact natural systems may be more resilient to climate change (Furniss et al., 2010). This is especially true in a context of broad-scale climate change impacts on freshwater services (Döll et al., 2015; Kundzewicz et al., 2008; Milly et al., 2005) coupled with increasing human demand (Hejazi et al., 2013; Wada el al., 2013) and resulting water stress (McDonald et al., 2014; McDonald et al., 2011). 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).

Apart from the precautionary principle, assessing the economic rationale for natural infrastructure investments requires sufficiently reliable quantitative information about the benefits or "returns" that a natural infrastructure solution delivers in a given place for a given level of investment. Return on investment (ROI) analysis (Reilly and Brown, 2011) is routinely applied in both the private and public sectors to evaluate the performance of competing financial investment opportunities and projects but is equally applicable to conservation projects (Boyd et al., 2015). Indeed, several studies have documented the need for ROI or cost-benefit analysis in conservation decisions (Balmford et al., 2003; Ferraro, 2003a; Murdoch et al., 2007; Naidoo and Ricketts, 2006), demonstrating that the explicit consideration of both conservation returns and costs can dramatically increase conservation outcomes achievable with a given budget (Duke et al., 2014; Ferraro, 2003b; Murdoch et al., 2010; Polasky et al., 2001; Underwood et al., 2008).

Watershed management (conservation and restoration of native vegetation; best management practices) may offer substantial and widespread potential to cost-effectively deliver hydrologic services (McDonald and Shemie, 2014) and thus should be considered alongside engineering solutions in addressing water supply challenges. Mobilizing the needed much larger investments in watershed natural infrastructure (e.g., Asian Development Bank, 2015; Ozment et al., 2015) often will require compelling evidence of their performance in providing desired hydrologic services or associated welfare gains at competitive cost (Bennett and Carroll, 2014). This is especially true for private sector investments, which are seen as key to closing the funding gap for water infrastructure globally (Sadoff et al., 2015). Yet, there exist few analyses of the effectiveness of payments for watershed services (PWS) programs in developing countries (Börner et al., 2017). Fewer still compare service benefits with program costs to assess the ROI of watershed conservation and restoration.

Ferraro et al. (2012) identified only ten credible economic valuation studies of forest hydrological services in developing countries. Of these, only three (Guo et al., 2007; Klemick, 2011; Veloz et al., 1985) also estimate the costs of the interventions they evaluate and calculate, or allow calculating, project ROI. Combined with Quintero et al. (2009), De Risi et al. (2018), Saenz et al. (2014) and Vogl et al. (2017) to our knowledge there exist only seven rigorous, peer-reviewed ROI assessments of forest hydrologic service projects in developing countries. This dearth of credible economic analyses of watershed conservation is disconcerting given the large number of such projects found in tropical and subtropical regions that have the explicit purpose of increasing hydrologic service flows (Salzman et al., 2018; Porras et al., 2013), and given that assessing "land use effects on ecosystem service provisioning in tropical watersheds is still an important unsolved problem" (Ogden and Stallard, 2013, p. E5037). Importantly, none of the available studies are from Brazil's Atlantic Forest, a region experiencing rapid growth in watershed conservation projects with hydrologic service objectives (Bennett and Ruef, 2016; Bremer et al., 2016) and home to over 120 million people (Tabarelli et al., 2010). While few payments for environmental services projects adequately address design and evaluation (Naeem et al., 2015), we apply a best practice framework for economic analysis of ecosystem service projects to target interventions and assess the expected ROI of a recently-created PWS program in Camboriú, Santa Catarina State, Brazil. Importantly, this framework yields natural infrastructure ROI estimates expressed in the same performance metrics routinely used to evaluate engineering alternatives.

1.1. Study area

The Camboriú watershed, located in Santa Catarina state in southern Brazil, has a drainage area of 199.8 km^2 (Figure 1). The municipal drinking water intake is just upstream of the urbanized area, with a drainage area of 137 km^2. The climate is humid subtropical (Köppen classification: Cfa), with a mean annual temperature of 21^o C, no dry season and hot summers. The Camboriú River has a mean monthly discharge of 3.41 $m^3 \cdot s^{-1}$ (maximum: 17.99 $m^3 \cdot s^{-1}$; minimum: is 0.49 $m^3 \cdot s^{-1}$; EMASA, unpublished data). The watershed relief is defined by the Tabuleiro mountain range, featuring steep slopes and deep valleys susceptible to surface runoff and strong erosion, including landslides on cleared areas, and the coastal plain, formed by sedimentary sand-clay and quartz-sand deposits (Urban, 2008).

The land use pattern in the watershed resembles that of many other coastal watersheds in Brazil's Atlantic Forest, a biome recognized for its biodiversity and high degree of endemism (Ribeiro et al., 2009) whose historic deforestation was first driven by timber exploitation, followed by sugar cane expansion, widespread conversion to pasture and coffee and, more recently, urban sprawl and expansion of *Eucalyptus* plantations (Teixeira et al., 2009). The urban area in the watershed is heavily concentrated along the coast, with a thin strip of very high-density high-rise ocean front development surrounded by a high to medium-density mixed use area. This is followed by a zone of residential sprawl fast expanding into the alluvial floodplain, which is dominated by pasture and row-crops (primarily rice). The slopes are primarily in native forest but also feature pastures and, increasingly, timber plantations. High rates of both deforestation and regrowth during the past 100 years left a fragmented forest landscape dominated increasingly by younger secondary forests (Teixeira et al., 2009). Family farms in the watershed declined by over two-thirds in number between 1970 and 2006 and currently cover one third of the non-urban portion of the watershed. During the same period, subdivision of rural properties for development of weekend homes and small lodges also increased (Projeto Produtor de Água da Bacia do Rio Camboriú, 2013).

Approximately 95% of the population in the watershed (208,319 in 2016; Instituto Brasileiro de Geografia e Estatística, 2016a, 2016b) resides in the coastal urbanized areas of Balneário Camboriú and Camboriú city, the former a famous beach destination that features Brazil's tallest buildings and attracts increasing numbers of domestic and foreign visitors (Ferreira et al., 2009; Lohmann et al., 2011) who swell population to over 800,000 during the high season (mid-December−early March).

1.2. Water supply challenges and the Rio Camboriú Water Producer Program

Both municipalities rely on the Camboriú River for their drinking water and are supplied by the Balneário Camboriú water company, EMASA. In recent years, high demand during the summer season and the absence of large-scale water storage infrastructure repeatedly led to the threat of intermittent supply shortfalls. High sediment loads at the municipal water intake exacerbate the problem because they increase treatment water losses. EMASA has evaluated several options for increasing supply, including water storage in the watershed through flooding of native forest and agricultural lands; water transfers from a neighboring watershed (Itajai) characterized by substantially lower water quality necessitating advanced treatment; and watershed management including conservation of natural forests and restoration of degraded high sediment loading areas. Due to the high projected costs of the first two options and the promising results of initial feasibility assessments of the third, the utility decided to first invest in the latter while also expanding treatment plant capacity. To implement the watershed conservation strategy, EMASA partnered 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 to create the Camboriú PWS project.

Observational evidence and studies from similar watersheds suggest that major contributors to sediment loading in the watershed include unpaved roads lacking minimal best management practices (Duff, 2010; Guimarães et al., 2011; Minella et al., 2008); pastures on steep slopes (Cerri et al., 2001); stream channel erosion and bank destabilization caused by cattle entering unfenced streams and foraging on regenerating riparian vegetation; stream channel erosion caused by hydraulic energy of high precipitation events (Minella et al., 2008); lateral channel migration; and croplands (Mello et al., 2018).

Exclusion of cattle from streams through fencing of river margins and reforestation of riparian areas (Gumbert et al., 2009; Palhares et al., 2012) and steeply sloped, highly erodible lands with low vegetation cover are recognized as effective soil conservation practices in Brazil (Saad et al., 2018; Teixeira Guerra et al., 2014), and forest cover and riparian restoration have been shown to improve water quality and reduce suspended sediment in other Atlantic forest watersheds (Mello et al., 2017; Monteiro et al., 2016). On pasturelands, upland and riparian reforestation require livestock exclusion (fencing) to permit seedling or tree establishment and enhance tree survival. Best management practices can substantially reduce erosion from unpaved roads (Baesso and Gonçalves, 2003; Kocher et al., 2007), but their impact on sediment loading into streams depends on the hydrologic connectivity of roads and streams (Duff, 2010; Mills et al., 2007).

The PWS program currently implements three interventions whose priority ranking is based on expected sediment loading reductions: 1) restoration of degraded riparian areas and areas surrounding natural springs, through a) fencing for cattle exclusion and b) planting of native tree seedlings or enrichment, depending on the state of degradation; 2) conservation of relatively intact riparian areas featuring regenerating forest, through riparian fencing for cattle exclusion; 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. Interventions are implemented by contractors paid by the program. In July 2012, the program opened a call for proposals from landowners. Landowner selection and implementation of the first interventions began in 2013. Annual implementation capacity is approximately 80 ha per year. For each property submitted for enrolment, the program develops an “ideal” intervention design encompassing all priority areas, with a corresponding annual cash payment based on area size, priority ranking and level of degradation and the official opportunity cost of pasture land in Balneario Camboriú. The latter, known as ‘Unidade Fiscal do Município’ (UFM), in 2015 was BRL 223 (~USD 70 at the average 2015 BRL-USD exchange rate) $ha^{-1}\cdot yr^{-1}$. Priority 1, 2 and 3 areas earn 1.5 UFM, 1 UFM and 0.5 UFM, respectively. The actual intervention design is then negotiated with each landowner and payments are adjusted accordingly. Interventions are inspected every six months by a group of program representatives, who must agree that interventions are well maintained before payment is authorized. Contracts last two years, are renewable and can be terminated if landowner performance is considered unsatisfactory.

2. Methods

2.1. Conceptual model and analysis overview

We synthesized from the literature a best-practice framework for evaluating the economic performance of watershed conservation programs (Appendix A) and used this framework to estimate the ROI of the Camboriú PWS program as a sediment control measure (Figure 2). To identify the relevant sediment metrics for the hydrologic modeling, we constructed empirical sediment cost functions for individual components of EMASA’s treatment operations affected by sediment in intake water. We then used 1 meter [m] spatial resolution land use maps from two recent years (2003, 2012) (Fisher et al., 2017) to develop a LULC change model for the watershed. We used this model to generate counterfactual (i.e., without PWS program) LULC for the year 2025, when the program is expected to have enrolled the lands most crucial for sediment control and most interventions will have attained their full functionality. This counterfactual land use scenario represents the business-as-usual land use needed to estimate sediment outcomes in the absence of the program. To target interventions, we ran the 2012 and counterfactual 2025 land use maps through a Soil and Water Assessment Tool (SWAT) model calibrated to the watershed using the 2012 LULC, daily flow and turbidity, and climate and soil data (Fisher et al., 2017). This allowed us to identify the areas where interventions would produce the largest reductions in sediment yield versus the counterfactual, and allocated the program's interventions to these areas to generate a land use map representing the intervention (i.e., with the program) scenario. We then ran the SWAT model on both the 2025 intervention and counterfactual land use maps to estimate the reduction in TSS at the EMASA intake attributable to the PWS program, and used the sediment cost functions to estimate the value of TSS reductions to EMASA. Finally, we used estimated sediment reduction or value and PWS program costs to calculate three ROI metrics useful for evaluating the economic performance of natural infrastructure projects.

2.2. Identification of target service metrics

The main operational processes of the EMASA treatment plant impacted by sediment in intake water are 1) intake channel dredging; 2) water pumping to and within the treatment plant; 3) chemical use for coagulation and flocculation; 4) settlement basin sludge discharge and disposal; and 5) back-flushing of final gravity filters (Appendix Figure B.1). Because the heavier sediment fraction settles in the intake channel upstream of the treatment plant intake, TSS is the ecosystem service parameter of primary concern for EMASA. Our hydrologic modeling thus was set up to estimate impacts of interventions on TSS at the EMASA intake.

2.3. Land use/land cover change analysis and modeling

To date, to our knowledge there has been no spatially-explicit modeling of future LULC change in the Camboriú watershed. We focused on land use rather than land cover to ensure that temporary land cover change (e.g., plantation harvest) did not bias the model by identifying temporary cover changes as permanent land use change.

We chose LULC data with 1 m spatial resolution for the LULC change and hydrologic analyses, for three reasons. First, individual instances of observed recent forest cover change in the watershed are small, generally <30 m in width, presumably due to forest cover requirements imposed by Brazil's Forest Code. The same is true also for LULC modifications resulting from program interventions, a substantial portion of which consist of riparian reforestation. Much of the recent and future (counterfactual and intervention) LULC change thus may be undetectable even with medium-resolution imagery such as 30 m (Landsat). Finally, 1 m spatial resolution data improved LULC classification accuracy and hydrologic sediment model performance in the watershed (Fisher et al., 2017).

We chose 2003−2012 as the LULC change reference period. While the coastal fringe real-estate construction boom in Balneário Camboriú began in the 1970s (Lohmann et al., 2011), around the year 2000 the urban area entered the phase of maximum densification of the coastal zone and urban sprawl into the hinterland (Ferreira et al., 2009). It is this sprawl that is driving the urban expansion into the watershed, making the period since 2000 an appropriate basis for predictions of future residential land conversion. This period also captures the continuing decline in cattle farming and expansion of plantations and second-home development in the rest of the watershed (Projeto Produtor de Água da Bacia do Rio Camboriú, 2013). Moreover, the earliest cloud-free 1 m resolution imagery for the entire watershed is available for 2003/2004 (Fisher et al., 2017).

We used Land Change Modeler (LCM) for ArcGIS 2.0 (http://www.clarklabs.org/; Pérez-Vega et al., 2012) to identify spatially-explicit land use change between 2003 and 2012 land use maps with 1 m spatial resolution for the watershed upstream of the EMASA intake (Fisher et al., 2017), and to predict land use in 2025. The seven land use classes (forest, plantation, rice, pasture, bare, impervious, water) resulted in 42 possible transitions (7^2 minus 7 where no change occurred). To keep the analysis computationally tractable and exclude minor transitions (by area) unlikely to correlate with predictive variables, we limited the transitions to the eight most significant ones (by area) during 2003 to 2012. Together these represent 90.5% of all land use change observed during that period (Table C.1).

Out of the large set of potential LULC change drivers (Blackman, 2013; Busch and Ferretti-Gallon, 2017; Soares-Filho et al., 2004), we selected for consideration eleven (Table C.2) significant drivers of LULC change in Atlantic Forest areas experiencing the same land use change patterns observed in Camboriú (Appendix C). These include distance to roads, urban centers, and rivers; slope; elevation (Teixeira et al., 2009); and distance to already-converted lands (which has been found to drive forest change; Soares-Filho et al., 2004) in impervious, bare, pasture, plantation, or rice, respectively. We did not include protection status because the watershed is almost exclusively privately owned; thus, the main source of protection is the national Forest Code, compliance with which is generally low due to low levels of enforcement (Appendix C). Slope and distance to rivers or plantations had almost no predictive power and were excluded from the final LCM model.

Change prediction to 2025 via LCM was accomplished using a Markov Chain analysis without added restrictions or incentives for any modeled transition, that is, assuming no change from their 2003-2012 levels in legal or economic factors affecting land use change. This yielded an estimate of pixel-level land use change probability. For the sub-models for each land use transition, we used all eight predictive variables and let LCM determine the appropriate weights of each using the SimWeight method. Although five of the predictors are based on distance to land cover (which changes over time) we left all variables as static rather than dynamic to avoid over-training the model from its early predictions given the low rate of land use change in the study area. To produce a specific "hard" prediction of expected baseline land use in 2025, LCM uses a multi-objective land allocation algorithm that determines which land use classes will expand or shrink, respectively (based on the probability of all transitions). It then uses a Markov chain run to allocate the specific changes to each pixel (Eastman et al., 1995).

2.4. Hydrologic modeling

We modeled the impact of interventions on TSS concentrations at the EMASA intake using SWAT (SWAT 2012 rev. 637; Arnold et al., 1998; Bressiani et al., 2015; Gassman et al., 2007), a physically-based, continually evolving public-domain watershed modeling tool and the most widely-applied hydrology model globally (Dile et al., 2016; Francesconi et al., 2016; Krysanova and White, 2015). The SWAT model was built for the watershed portion upstream of the EMASA intake using 1) 1m land use and digital elevation data from 2012, and 2) daily flow and sediment load data (aggregated from hourly flow and 15-minute turbidity monitoring data, respectively) from local gauge stations and optical turbidity sondes. To avoid over-fitting the model to calibration data, model parameters were calibrated using a split-sample calibration method, with a training (5/27/2014-12/31/2014) and a validation (1/1/2015-11/06/2015) period (Fisher et al., 2017). The daily-modeled flow and sediment load both met satisfactory performance criteria for monthly models as recommended by Moriasi et al. (2007) over the combined training and testing period (flow: Nash-Sutcliffe efficiency=0.63, PBIAS=-5.3; sediment: NSE=0.56, PBIAS=11.45; Fisher et al., 2017). Because daily-scale models are likely to have poorer performance statistics than coarser time-step models and their evaluation criteria therefore should be more relaxed (Moriasi et al., 2007), performance of our model might be rated as good by daily-scale criteria.

We ran the SWAT model with land use defined by the 2025 LULC maps for intervention (2.5) and counterfactual (2.3) scenarios to test the effects of the program interventions on TSS loads at the EMASA treatment plant. These models isolated the effects of the land-use differences among the two scenarios by adopting the 2014 climate data and identical parameters to those found through calibration. Climate change may increase or decrease the expected intervention effects, but was deemed beyond the scope of this study.

2.5. Targeting of interventions based on SWAT and LCM results

Cost-effective portfolio selection requires targeting of interventions based on costs and benefits (Duke et al., 2014). To target restoration activities, we first identified potential intervention sites as lands currently in pasture or bare (excluding roads) and located in riparian areas or near natural springs, defined following the Brazilian Forest Code (Soares-Filho et al., 2014) as a 30-m buffer on both sides of a stream and a radius of 50 m around springs. We focused on riparian and spring areas because the aquatic-terrestrial ecotone governs the transfer of sediment between terrestrial areas and waterways. From these lands, we excluded all areas that the LCM analysis predicted to revert to forest by 2025, and then selected as targets for the restoration activities those lands that our SWAT model estimated as having the highest sediment yields in 2012, until reaching the estimated total program restoration implementation capacity of 326 ha by 2022, the expected end of the intervention phase.

To target conservation activities, we selected the 313 ha in priority areas that our LCM model predicted to change from forest in 2012 to non-forest in 2025 in the counterfactual scenario. To generate the 2025 intervention scenario land use map, land use on intervention sites was changed to forest in the counterfactual 2025 land use map.

2.6. PWS program costs

We compiled information about the full costs of PWS program-related activities during 2009-2015 and projected future annual costs based on expected activity time profiles. Activities include hydrologic, political and economic feasibility studies; coordination, communication and program design; program management (administration, external communication, landowner compliance monitoring); surveying; landowner engagement and contract development; planning and implementation of restoration (plan design for each property; fencing, planting, enrichment) and conservation (fencing) interventions and their maintenance (follow-up inspection to ensure tree survival; replanting where necessary); and payments to landowners. We included all costs irrespective of who bears them, including grants from multilateral institutions and private foundations that supported several aspects of program development including feasibility studies and hydrologic monitoring infrastructure, and staff time of EMASA and other program partners (The Nature Conservancy; EPAGRI-CIRAM).

2.7. Benefits estimation

We estimated the avoided costs for EMASA that result from the reductions in TSS concentrations in intake water in the intervention scenario. To do so, we used EMASA data to estimate empirical relationships between sediment concentrations in intake water and operational costs for five discrete processes: intake channel dredging; pumping; chemicals use; sludge disposal; and treatment water loss (Table 1). We distinguished between peak (December-March tourist high season) and off-peak demand periods. We assumed that in off-peak months there is no demand for any additional water output; thus, reduced water loss from lower TSS concentrations and consequent lower sedimentation basin sludge discharge and filter backwashing is used to reduce water intake. During peak months, when excess supply frequently approaches zero, we assumed that the reduced TSS-related water loss is used to increase plant water output to permit keeping short-term storage infrastructure at capacity. This infrastructure comprises two municipal water towers, industrial and commercial water storage tanks as well as the cisterns now required in apartment buildings and condominiums to reduce supply interruption risk. Thus, during peak months, the benefits for EMASA of reduced TSS concentrations, in addition to reduced treatment costs, also include revenue gains from increased water sales. We valued these gains using the August 2015, user type and use volume-weighted marginal price of water and sewer (automatically billed at 80% of water use) of USD 1.90 (BRL 6.08) m^{-3} (Appendix D).

We assumed that recent (2008-2014) average absolute increases in municipal peak (274,000 m^3) and off-peak season (398,000 m^3) water supply and average peak (16.9%) and off-peak season (14.9%) inflow losses will remain constant and used supply and losses to calculate future plant intake.

In addition to operational costs, we also considered potential avoided capital costs of reduced TSS levels. Our base case takes the recent (2015) treatment plant capacity expansion as given and assesses the effect of reduced TSS concentrations on plant operational costs only. In contrast, our hypothetical avoided capitol cost case assumes that this expansion would have been reduced in size in proportion to the reduction in plant output losses that results from the lower TSS concentrations in the intervention scenario, and counts the corresponding avoided capital cost as an additional benefit for EMASA (Appendix E).

2.6.1. Temporal incidence of benefits

The SWAT-modeled TSS concentration difference at the EMASA intake between the intervention and counterfactual scenarios represents the full impact once all interventions have been implemented and developed their full sediment loading reduction functionality.

We calculated the actual TSS reduction achieved in each year as a function of the age composition of the total intervention area implemented to that year and the age-specific TSS control efficiency of interventions (Table F.1), assuming very conservatively (compare Borin et al., 2005; Vogl et al., 2017) that the impact of forest restoration on TSS increases linearly from zero in year one to 100% in year ten. Conservation activities avoid forest loss and therefore achieve full functionality in the year they are implemented. Total conservation (313 ha) and restoration (326 ha) interventions were spread evenly over 2015-2022, meaning the full TSS control potential is first achieved in 2032.

2.7. ROI calculation

We calculated three ROI metrics for the Camboriú PWS program, separately for EMASA and the program overall: 1) The cost-effectiveness in reducing TSS, expressed as average reduction in mg $TSS \cdot l^{-1}$ removed from intake water per USD invested, or as 2) average kg sediment load removed from intake water per USD invested; and 3) the benefit-cost ratio or monetized ROI, calculated by dividing the value of the benefits of TSS reductions in municipal treatment plant intake water by PWS program costs. Because investments in grey drinking water treatment infrastructure have economic lifetimes of 15-25 years (mechanical and electrical treatment plant systems and pumping stations) to 60-70 years (concrete structures) (U.S. EPA, 2002), we calculated ROI metrics for 30- and 50-yr time frames.

Social 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). We discounted all costs and benefits to their 2014 present value (PV) equivalents using Brazil's estimated social consumption discount rate of 3.85% (Fenichel et al., 2017).

3. Results

3.1 Observed (2003-2012) and predicted (2012-2025) land use change

A total of 1,125 ha of gross land use change was observed between 2003 (Figure C.1) and 2012 (Figure C.2), or 8% of the 13,668-ha watershed area upstream of the EMASA intake (Figure 3). Due to transitions between land use classes, net change was approximately half that (562 ha; Table C.3). The single largest net change was a reduction in pasture, balanced by increases in plantation, bare, impervious and forest.

For 2003-2012, the LULC change model correctly predicts the included transitions 43-72% of the time as indicated by the hit rate. The overall model hit rate (both area-weighted and unweighted) is 55%, meaning that, on average, included transitions are predicted correctly more often than not, and more often than if predicted transitions were chosen randomly. Model predictive ability is constrained by the complex composition and large number of land cover transitions in the watershed and the omission of socio-economic and demographic drivers of LULC change, for which we lacked data.

Absent the PWS program, predicted total net land use change by 2025 is 582 ha (4.2% of the area upstream of the EMASA intake), dominated by a reduction in pasture (-2%) and increase in plantation (1.3%), followed by increases in impervious (0.4%), bare (0.3%) and forest (0.2%; Table C.4). Analysis of individual land use transitions (Table C.5) reveals that while forests show a net increase fueled by abandonment of some pastures, 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 4). Conversely, while pastures are being replaced by plantations and forest throughout the watershed, this effect is most pronounced in the headwater areas (green areas in the lower portion of Figure 4). These predictions are consistent with the empirical observations of mature native Atlantic Forest continuing to be replaced by regrowing forest patches (Joly et al., 2014) and forest regrowth being highest at higher elevations and farther from urban areas and roads (Teixeira et al., 2009).

3.2 PWS program intervention areas and impact on sediment yield

Figure 5 shows the areas selected for restoration and conservation activities, selected based on modeled current (Figure 6, top panel) and counterfactual 2025 (Figure 6, bottom left panel) contribution of all sites above the EMASA intake (point 1 in the figure) to sediment loads in the Camboriú River at the EMASA intake. A comparison of intervention and counterfactual scenarios (Figure 6) shows that the interventions will substantially reduce sediment yield from most high-yield sites.

3.3 Reduction in TSS concentrations at municipal water intake

In the counterfactual scenario, modeled TSS concentrations in 2025 are predicted to be 10.2% lower than in 2012 (Table 2). In the intervention scenario, by 2032 average annual TSS concentrations at the municipal intake are reduced by an estimated 14.2% (13 $mg \cdot l^{-1}$) compared to the counterfactual scenario (Figure G.1), with an average intake volume-weighted annual reduction during 2015-2045 of 11.1 mg·l^{-1}.

3.4 PWS program benefits and costs

Sediment-related benefits of the PWS program for municipal water provision average USD 194,000 (USD 202,000 in the hypothetical avoided capital cost case) per year (undiscounted) during 2015-2045 and are dominated by avoided revenue losses to EMASA from reduced peak-season water loss (76%), followed by avoided chemicals use (15%) and sludge disposal (6%) (Table H.2). Benefits continue to increase with municipal water supply even after interventions have attained full functionality. Costs during 2015-2045 average USD 176,000 per year for EMASA, and USD 228,000 per year (all undiscounted) for the project overall (Table H.3), with transaction costs (TAC; all program activities except intervention design, implementation and maintenance, and payments to landowners) accounting for 39% of EMASA and 53% of overall program cost. Because of markedly different time profiles of benefits (steadily increasing over time from zero) and costs (heavily front-loaded) (Figure H.1), average annual benefits (Table 3) decline relative to costs (Table 4) in PV terms.

3.5 Camboriú PWS program ROI

For EMASA, program ROI (i.e., PV benefit-cost ratio) for sediment control exceeds 1 for analysis horizons exceeding 43 years (Figure H.2), a timeframe common for evaluating the economics of water supply infrastructure (U.S. EPA, 2002). If peak season water savings produced by the program had been used to reduce the size of the treatment plant expansion, break-even time would decline to 40 years. Overall (i.e., including program costs not borne by EMASA) ROI for sediment control surpasses 1 only after more than 70 years. Table 5 shows the three ROI metrics for the program for time horizons of 30 and 50 years, respectively.

4. Discussion

Our analysis indicates the Camboriú PWS program will be a cost-effective tool for the utility for reducing TSS concentrations in municipal intake water. We expect interventions, once fully implemented and functional, to lower TSS concentrations at the utility intake by over 14% vs the baseline (i.e., the counterfactual). Based on local utility data on sediment-related treatment costs, we predict this TSS reduction to lower total annual treatment costs for the utility (USD 0.21 per m^3 water output in 2011; EMASA data) by 3.8%. This estimate is in good agreement with the few reported estimates of the impact of TSS on municipal drinking water treatment costs. McDonald and Shemie (2014) report that in their sample of more than 100 U.S. cities relying primarily on surface water sources, a 10% reduction in sediment concentration reduces treatment plant operation and maintenance (O&M) costs (excluding pumping, distribution infrastructure O&M and reservoir dredging) by 2.6% on average. Using calibrated OTTER models for four water treatment plants, Grantley et al. (2003) estimate that a 25% decrease in TSS and a 15% decrease in total organic content can reduce treatment (chemicals use, residuals disposal and power consumption of wastewater pumping) costs by 5%. Warziniack et al. (2017) find that in a sample of 26 conventional treatment plants in the U.S. with mean percent source watershed in forest cover (53%) similar to the Camboriú watershed, a 1% reduction in turbidity was associated with 0.19% lower treatment cost. Price and Heberling (2018) review 12 studies from the U.S. and other countries that statistically estimate the effect of turbidity on drinking water treatment costs. They find that costs increase by 0.14% on average for each 1% increase in turbidity. Given our estimated 14% reduction in TSS concentrations and the turbidity-TSS relationship in our watershed (Figure H.3), the elasticities of treatment cost with respect to turbidity reported in these four studies would result in treatment costs reductions of 3.2-5.9%, bracketing our estimate of 3.8%.

Our finding that the ROI of the PWS program exceeds 1 for EMASA in year 44 indicates that the utility's investment in the program as a sediment control measure is financially justified. Importantly, its ROI increases if the utility manages to attract additional cost sharing due to third-party positive externalities. If costs borne by entities other than EMASA are included, the program is unlikely to be justified economically solely by its sediment control effect, as overall program ROI for just sediment control surpasses 1 only after more than 70 years. For the program's social ROI, that is, the ratio of the value of all program benefits and costs, to surpass 1 after 43 (30; 50) years, the program would need to produce co-benefits with a PV of USD 31,100 (USD 69,400; USD 19,900) per year on average. A preliminary analysis of those co-benefits (4.1.1) indicates that social program ROI very likely does exceed 1.

In the Camboriú program, TAC account for half of total program costs. While such a high TAC share is not unheard of (Jayachandran et al., 2017), it is much higher than the share reported in the majority of the few PWS studies that estimate TAC (Alston et al., 2013; Finney, 2015; Wunder et al., 2008). We attribute this divergence to our attempt to account for TAC incurred by all program partners, something rarely done (Finney, 2015), and to account comprehensively for all program-related activities including assembly of, and coordination among, a technically strong and diverse group of program partners; legal and hydrologic studies; hydrologic and compliance monitoring; efficient targeting of site-specific interventions that incorporate individual landowner concerns; maintaining good landowner relations; and ongoing public communication. The high TAC thus result from a substantial investment in ensuring program performance and sustainability, and necessarily exceed those of programs characterized by generic or collective agreements (Alston et al., 2013; Kerr et al., 2014), low additionality (Blackman, 2013) or low conditionality (Kroeger, 2013; Wunder et al., 2008). Importantly, TAC explain nearly 90% of the nearly two-fold discrepancy between our program cost (USD 356 $ha^{-}1\cdot yr^{-1}$ over 30 years, undiscounted) and the average cost reported for several other Atlantic Forest projects (USD 133 $ha^{-1}\cdot yr^{-1}$; Banks-Leite et al., 2015 based on Guedes and Seehusen, 2011), which exclude transaction costs (Finney, 2015). Efforts to reduce TAC thus are important, beginning with the careful choice of the scientific analyses used to support program design. In the case of the Camboriú program, use of 30m instead of 1m resolution satellite imagery reduced hydrologic model performance and estimated program ROI (Fisher et al., 2017). However, given the utility’s strong focus on risk reduction, it is doubtful that this would have changed the decision to invest in the program, while at the same time it would have substantially lowered the costs of impact analysis (Fisher et al., 2017).

4.1 Sensitivity analysis and caveats

Both EMASA and overall ROI are sensitive to the treatment of co-benefits, choice of discount rate; intervention scale and time needed to attain full functionality; and assumptions about future increases in municipal water supply, targeting efficiency and leakage effects.

4.1.1. Co-benefits

Because the Camboriú PWS program produces multiple benefits for diverse stakeholders, social program ROI exceeds ROI for sediment control. Such divergence between the broader economic and the specific business cases for a specific objective or supporter is not surprising but highlights the importance of carefully scoping ROI analyses and interpreting their results. While quantitative analysis of the co-benefits of the Camboriú PWS program is beyond the scope of our study, the high degree of endemism and small remaining percentage (<12%) of Brazil’s historic Atlantic forest extent (Ribeiro et al., 2009) suggest that the program may produce biodiversity benefits by increasing (vs the counterfactual) forest cover by five percent of the watershed upstream of the EMASA intake. Studies in other Atlantic Forest watersheds found that overland flow from forest is significantly lower than from pasture (Pereira et al., 2014; Salemi et al., 2013), in line with the observed generally negative correlation between forest cover and peak flows and flooding (Filoso et al., 2017). The program thus is expected to lower flood risk during storm events. Finally, reduced water losses in the TSS treatment process and increased infiltration (Salemi et al., 2013) and dry season low flows (Pereira et al., 2014) associated with reforestation also lower the risk of supply shortfalls. Such risk reduction is an important reason for diversified investments in water infrastructure especially given projected increases in climate extremes in southeastern Brazil (Grimm, 2011; Marengo, 2009).

Riverine flooding historically has been a serious concern in the densely developed urban portion of the Camboriú watershed (CEPED, 2014), prompting in 2013 the installation of a flood early warning system that monitors streamflow at various points in the watershed in real time.

Our SWAT model (Fisher et al., 2017) predicts program interventions, once fully functional, to reduce the four highest annual river flood levels at the EMASA intake by 4% on average (Figure G.2). Evidence from other studies suggests that this value could be substantial. Even if in Camboriú it were only one tenth of the average value per household reported in other cities in Brazil and Ecuador (Table H.1), it would be 2 to 9 times as high (USD 169,000 to USD 856,000 per year) as the value associated with sediment reductions in municipal water supply (Table 3) and would lift social program ROI to 1 within 2 to 22 years, and to 1.2 to 3.6 within 30 years. We note, however, that our hydrologic model covers only two years so the peak flow reduction may be less for the largest events.

Because flood and supply risk reduction benefits accrue to local businesses, residents and visitors, either directly or via reduced municipal spending on flood damages and emergency response, PWS program cost-sharing with those beneficiaries would be justified. This could be achieved by incorporating watershed conservation costs into water user rates or levying a watershed conservation fee on high-season visitors, the latter based on the rationale that a large share of flood and supply risk reduction benefits occur during the tourist high season that encompasses the three months of the year when consumption and precipitation are highest and when tourists account for three-quarters of the combined population of the two cities.

A watershed conservation fee of only USD 0.009 m^{-3} water used— less than 0.7% of the current average rate paid by municipal water customers, or USD 2.50 for the average household per year — that declines to USD 0.003 in 2065, or of USD 0.28 per high-season visitor would result in the internalization of the low-end supply and flood risk reduction benefits estimate (USD 169,000 per year) and would lift EMASA's 30-year ROI of the program to 1.6, and 50-year ROI to

2.0. Recognition of the multiple benefits provided by the Camboriú PWS program has resulted in the state water and sanitation regulator's approval in June 2017 of a revised municipal water tariff structure that includes the Camboriú program's operational costs in water tariffs.

4.1.2. Discounting

Due to the time profile of benefits and costs (Figure H.1), discount rate and program ROI are inversely related. With a rate of 6% (the yield on recent 10-year Brazilian government bonds; Parra-Bernal and Kilby, 2017) rather than the 3.85% social rate used in this analysis, EMASA's 50-year ROI of the program for sediment control declines from 1.08 to 0.81. Consequently, using the utility's historical program cost share, the program would not be financially viable as a sediment control measure if the utility were required to use its cost of capital as discount rate.

4.1.3. Intervention scale

The currently planned program portfolio will leave more than 50 ha of high sediment loading areas untreated (red areas in bottom right panel in Figure 6). Because TAC account for a high share of total program costs and because many of these costs are independent of, or increase less than proportionally with, intervention extent, program ROI would increase if interventions were expanded to remaining high-loading areas. For example, increasing conservation and restoration extent each by 10% (64 ha total) compared to our analysis would increase total program costs by 6% but benefits by nearly 10%, EMASA's 30-year ROI from 0.77 to 0.85, and 50-year overall program ROI for sediment control from 0.82 to 0.86.

4.1.4. Timing of benefits

If interventions develop their full TSS control effect after three (e.g., Borin et al., 2005; Vogl et al., 2017) instead of the 10 years assumed here, EMASA's ROI reaches 1 in year 39 (vs 43) and 30 and 50-yr ROIs are 0.84 (vs 0.77) and 1.14 (vs 1.08) for EMASA and 0.63 (vs 0.59) and 0.86 (vs 0.82) overall, respectively (ignoring hypothetically avoidable capital costs).

4.1.5. Targeting efficiency and leakage

Our ROI estimates assume accuracy of our land use change predictions. Complete accuracy is unlikely due to potential model estimation error or possible future changes in the size (e.g., demand for beef or rural homes), effect strength (i.e., changes in the size or direction of the coefficients on the variables) or composition of ultimate LULC change drivers. Our LULC change model hit rate of 55% suggests model estimation error as the most likely source of error. However, we expect actual targeting efficiency to be higher than the hit rate, for two reasons. First, PWS program managers incorporate additional information omitted from the modeling due to a lack of data for most properties. Second, the hit rate indicates overall model accuracy in retroactively and spatially-explicitly predicting all specific included past land use transitions (e.g., from forest to pasture). Because in many cases the model correctly predicted a change in land use but incorrectly predicted the specific transition, overall model accuracy in spatially-explicit prediction of land use change *per se* exceeds the hit rate. It is the former that matters for targeting and additionality.

Because our land use change model was estimated over a very recent period (2003-2012) and our projection spans only 13 years, changes in land use change drivers are less likely to be of concern. Still, changes in national Forest Code enforcement, which has remained inconsistent (Schmitt et al., 2013; Soares-Filho et al., 2012); in agricultural conservation programs such as the ABC (low-carbon agriculture) investment program which supports activities such as recovery of degraded pastures or Forest Code compliance (Banco Nacional de Desenvolvimento Econômico e Social, 2013); in agricultural input or output prices; or in real estate development-related policies could change the economics of land use in the watershed.

Because land use is a continuous process, the optimal intervention portfolio is sensitive to the choice of modeling time horizon: extending the LULC change analysis beyond 2025 may identify sites with high TSS yields excluded in our portfolio because they are not predicted to be converted by 2025.

In targeting interventions solely based on expected additionality of TSS loadings and site costs, our site portfolio assumes risk-neutrality. Under risk aversion, this purely cost-effectiveness-based portfolio may change in favor of including sites with lower likelihood of conversion but the potential to yield large and difficult-to-mitigate sediment loading, thus trading off expected cost-effectiveness against certainty in avoiding highly undesirable outcomes (Bishop, 1978; Ciriacy-Wantrup, 1952).

Our analysis assumes no leakage, that is, displacement of land management activities targeted by interventions to non-intervention priority areas. Leakage within the EMASA drainage area could lower program ROI but we expect this outcome to be unlikely. Leakage is unlikely to occur on participating properties because contract terms do not permit internal relocation of land cover degrading management practices to other potential priority areas on a property, and effectively enforced conditionality of payments to date has ensured compliance with contracts. Leakage to priority areas on non-participating properties also is unlikely because most high-priority areas will be enrolled by the program and cattle raising is declining in the watershed (Figure 3).

4.1.6. *Other assumptions*

Our analysis assumes that municipal water demand continues to increase by the same annual increment as during 2008-2014. Given the projected growth in Balneario Camboriú's year-round population (Tischer et al., 2015), the continued growth of the real estate and tourism sector and the fact that EMASA currently abstracts less than one-fifth of annual river discharge, this assumption is reasonable. Lower increases would reduce program ROI while higher increases would increase ROI.

We also assume that PWS payments will remain constant in real terms. If payments were to increase due to increasing opportunity costs for landowners, program ROI would decline, all else equal.

Furthermore, based on experience to date, our estimates assume that once enrolled, lands remain in the PWS program. Turnover of participating lands would reduce program ROI.

4.2 Transferability of findings

We expect our finding of the importance of watershed management for municipal water supplies to apply to many other Atlantic Forest watersheds. The transferability of our particular ROI results to other catchments depends on similarities of major drivers of benefits and costs. These include drinking water treatment technology (e.g., with or without sludge water recovery); proximity to sediment thresholds for plant operation (e.g., avoided shutdowns due to excessive sediment); watershed size (the larger the watershed, the larger the scope of interventions needed to achieve a given TSS reduction [McDonald and Shemie, 2014]); portion of stream flow and hence intervention impacts captured by the treatment plant; watershed hydrologic properties (soils, slopes, instream-processes between intervention and beneficiary sites); presence of additional beneficiaries of sediment reduction (e.g., reservoir operators, canal owners, harbor authorities) or co-benefits and their willingness to cost-share; opportunity cost of interventions and hence PES payment levels; land use change patterns; conservation and transaction costs; and targeting efficiency.

5. Conclusions

We synthesized from the literature a best practice analytical framework and applied it to the Camboriú PWS program in Brazil to contribute to the limited evidence base on the ROI of natural infrastructure solutions to water supply challenges, and to inform future analyses that assess the performance of such solutions to hydrologic challenges.

Our findings indicate that the municipal utility's investments in watershed management to control TSS concentrations are justified on ROI grounds. Moreover, a preliminary analysis of program co-benefits indicates that the program also generates social net benefits for local stakeholders overall. The program's private and public ROI therefore both exceed 1.

Our analysis highlights the formidable challenge of reliably assessing PWS program performance *ex-ante*. Constructing rigorous *ex-ante* counterfactuals, benefit functions and calibrated hydrologic models entails significant information requirements and associated costs, and therefore often may be infeasible due to time or budget constraints. This highlights the need to select analyses based on their value of information (Fisher et al., 2017): analytical sophistication (and hence, generally, cost) should be defined by the level of uncertainty of results that is acceptable to decision-makers.

By targeting interventions based on both costs and benefits as well as a counterfactual baseline and by employing a quasi reverse-auction format, the Camboriú program incorporates key efficiency-enhancing features. Yet these features, together with extensive hydrologic and compliance monitoring and ongoing landowner engagement and public communications aimed to ensure long-term program sustainability, also lead to transaction costs that account for over half of total program costs. We expect these findings to be broadly representative: PWS programs rigorously designed to achieve high additionality and cost-effectiveness in target service provision as well as sustainability generally will have higher transaction costs and therefore higher total costs than programs lacking these features. Finally, our analysis highlights that the business case for a given stakeholder and program is sensitive to the program's ability to forge institutional arrangements that facilitate cost-sharing with recipients of program co-benefits.

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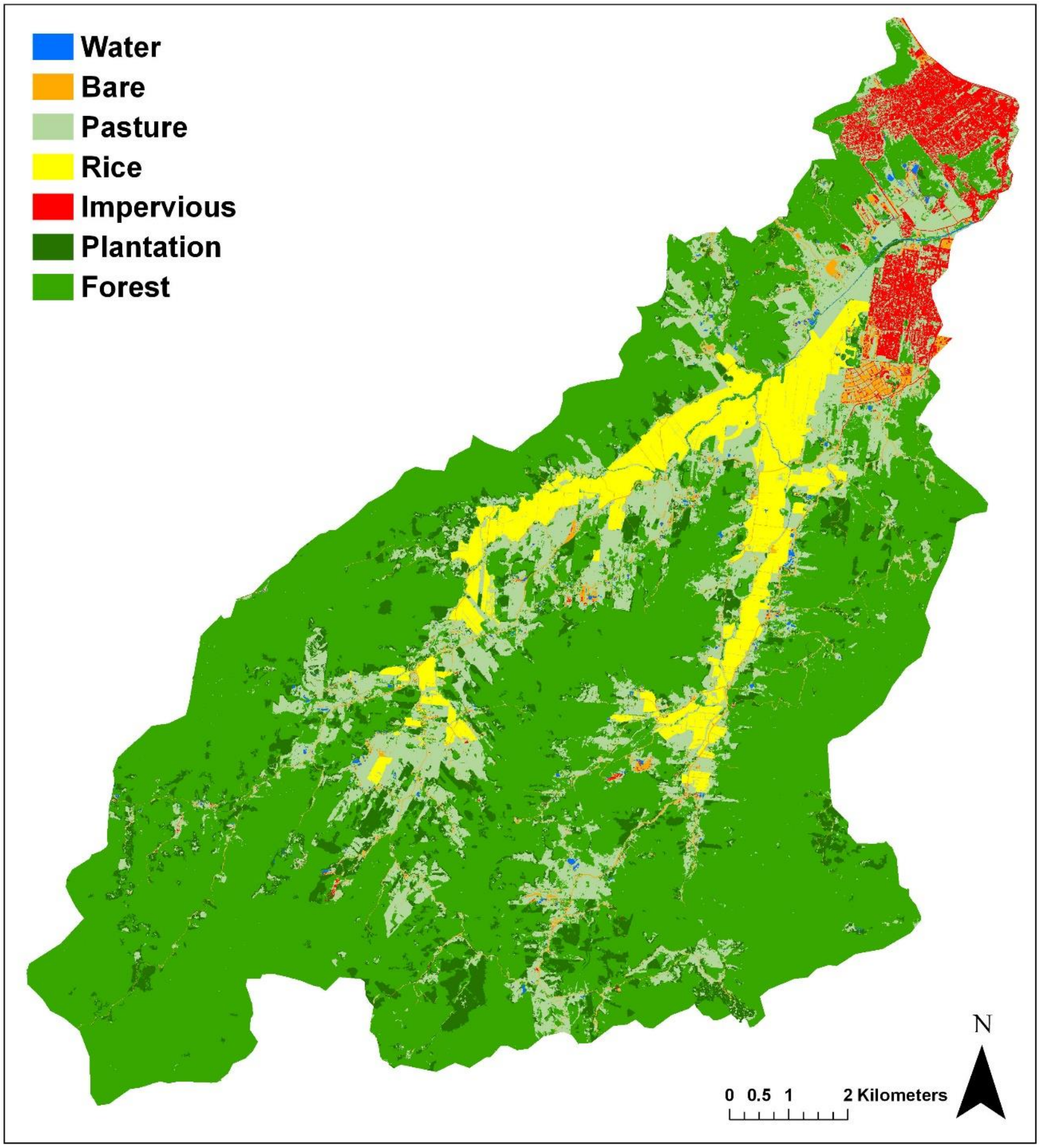

Figure 1: Land use map of the Camboriú watershed *[1.5-column fitting image]*

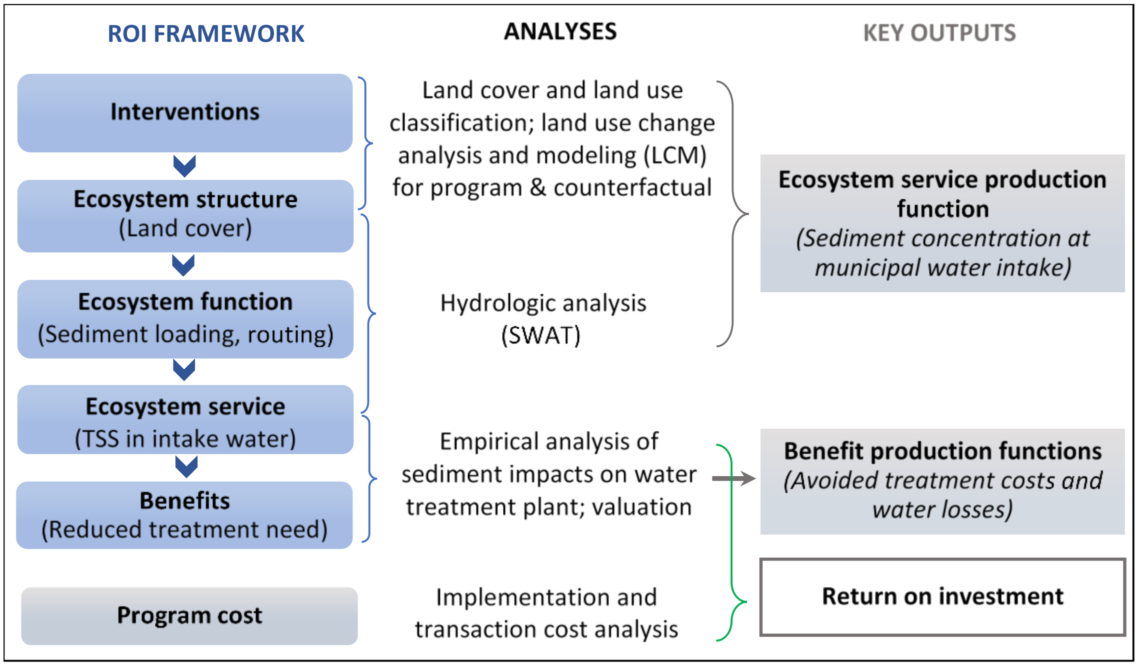

Figure 2: Analytical framework and analyses used to assess the return on investment of the Camboriú watershed conservation program for sediment management
[1.5-column fitting image]

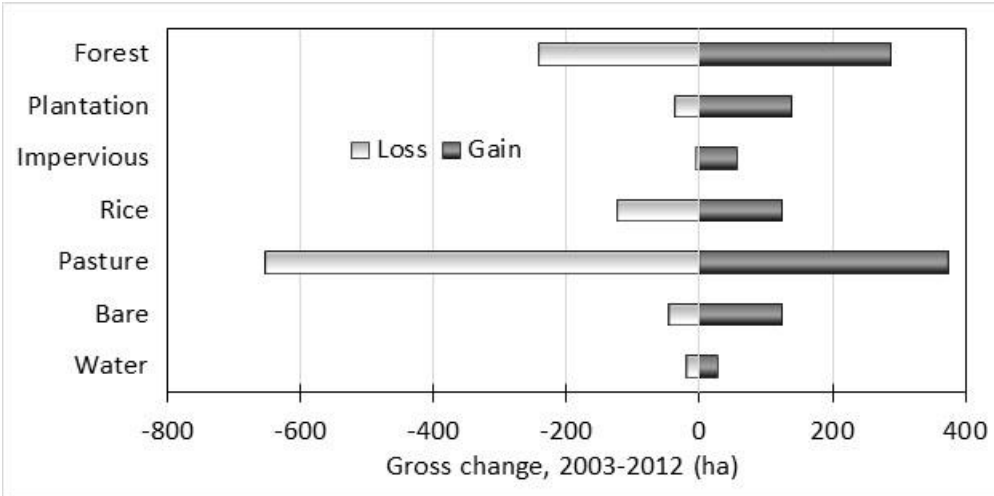

Figure 3: Gross land use change in the study area, 2003 to 2012 *[1-column-fitting image]*

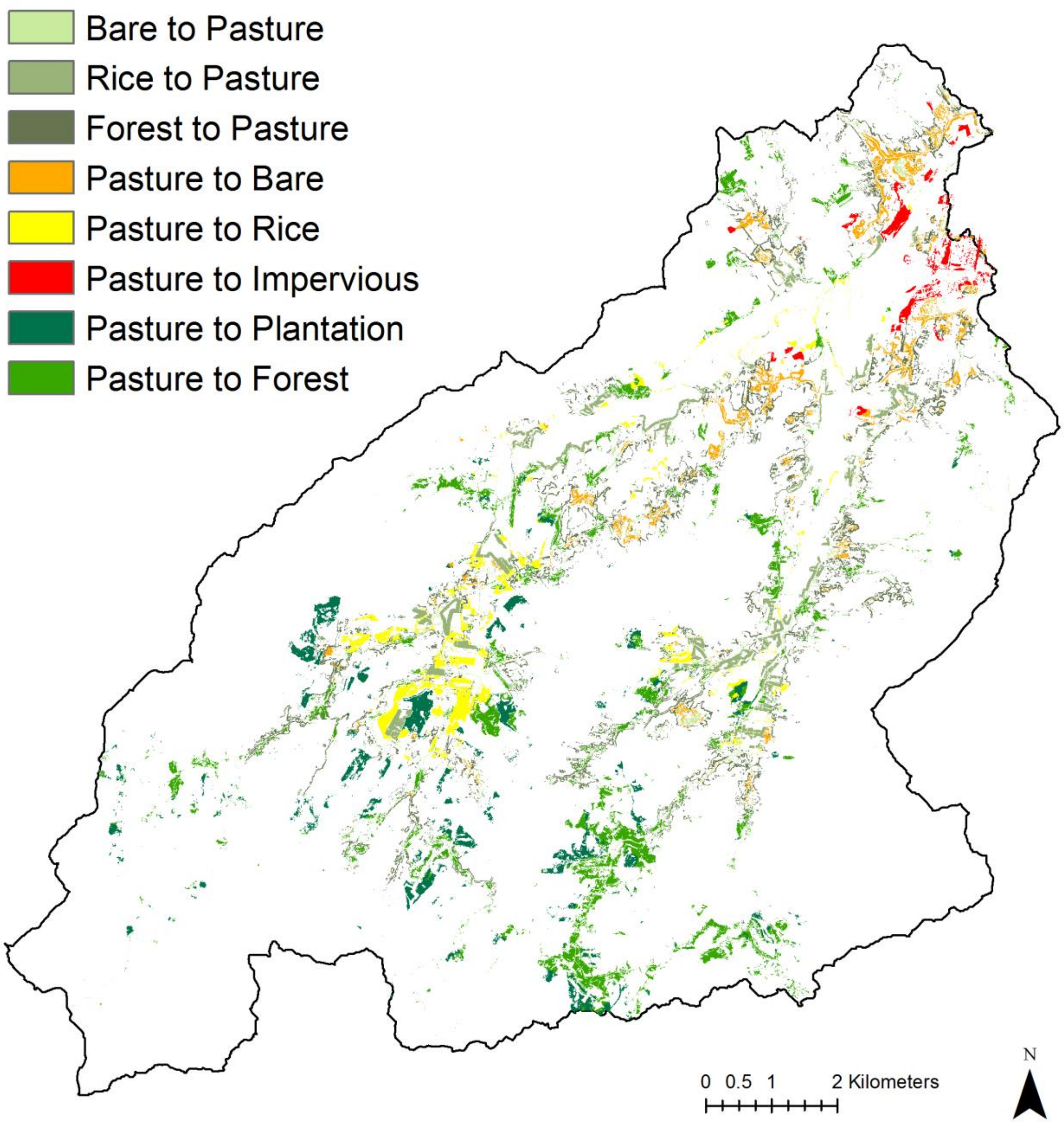

Figure 4: Predicted 2012-2025 land use change in study area absent PWS program *[2-column fitting image]*

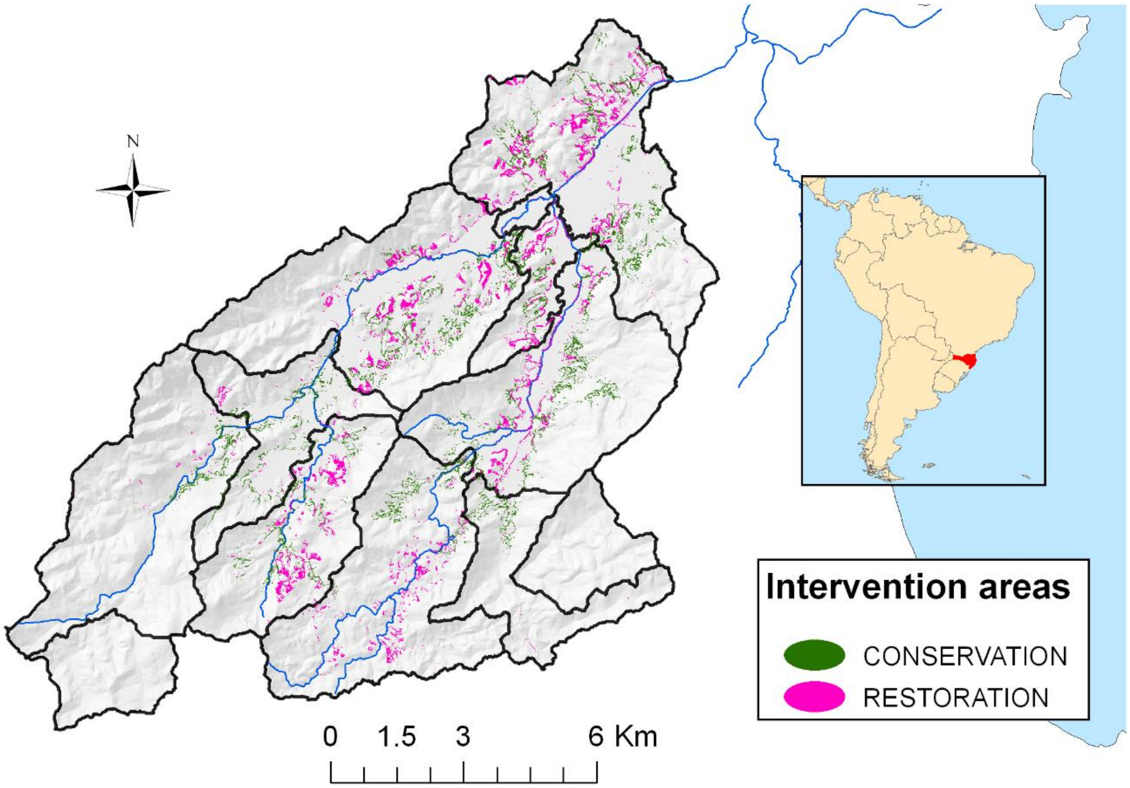

Figure 5: Conservation and restoration interventions in the Camboriú watershed in the area upstream of the municipal water intake. Shading shows elevation. Red area in inset indicates Santa Catarina state *[1-column fitting image]*

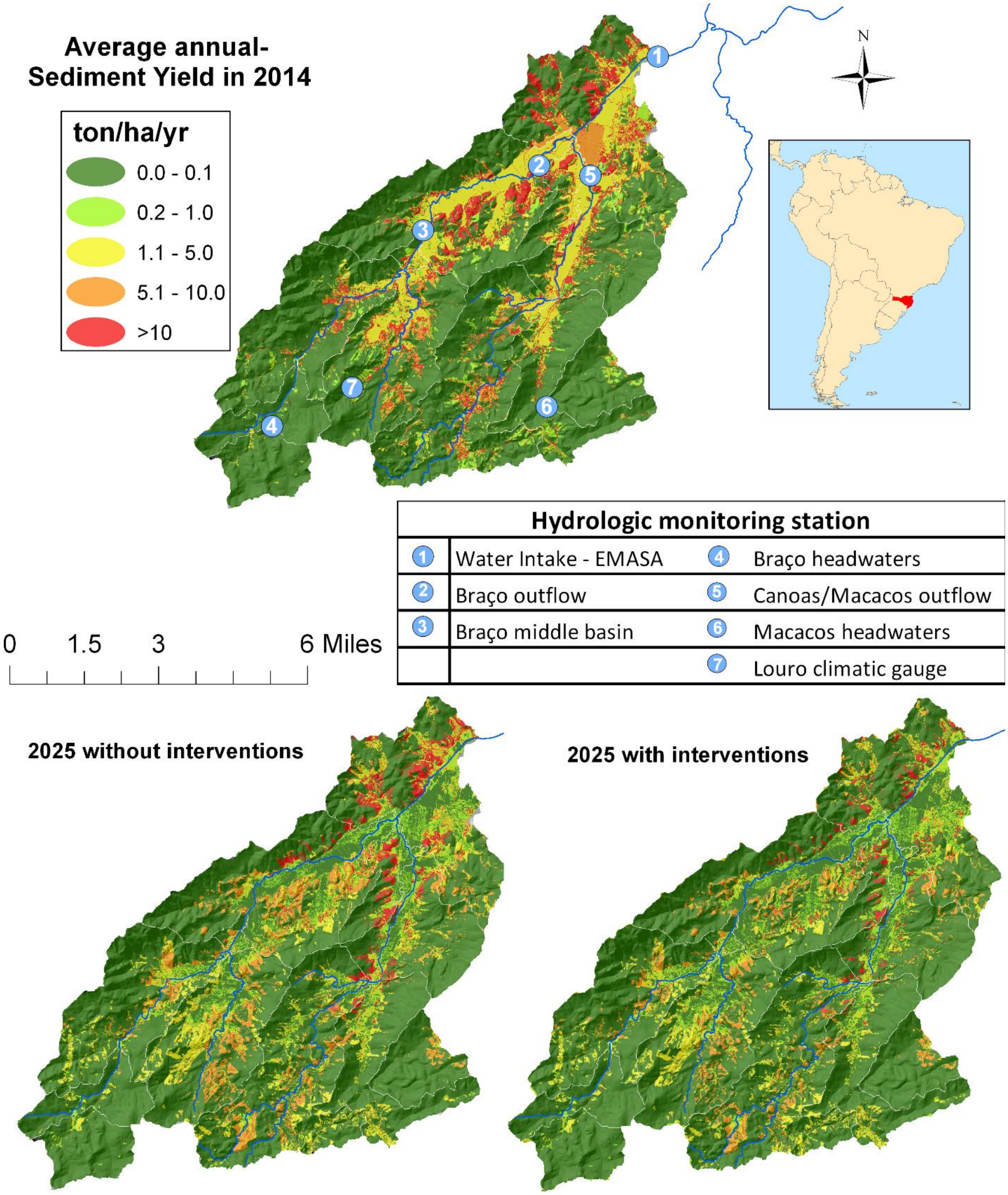

Figure 6: SWAT-modeled annual sediment yield in the Camboriú watershed in 2014 (using 2012 land use and 2014 climate) and 2025 *[2-column fitting image]*

Table 1: Sediment-related water treatment plant unit costs and quantities

	Monetary value	Unit	Quantity
Pumping: from intake channel to treatment plant	0.08	USD/kWh	0.245 kWh/m^3 [a]
Pumping: within treatment plant	0.08	USD/kWh	0.345 kWh/m^3 [a]
Coagulate (polyaluminum chloride)	0.38	USD/kg	25 mg/l
Flocculent (polymer)	3.71	USD/kg	0.03 mg/l
Water lost in filter back-flushing	1.90 [b]	USD/m^3	350 m^3/flushing/filter
Water lost in sludge	1.58 [b, c]	USD/m^3	992.8 g/l sludge
Treatment plant sludge disposal	18.75	USD/ton	9.24 t/day
Intake channel dredging [d]	4.70	USD/m^3	1,250 m^3/yr

Notes: All data from EMASA (B) for 2014 or 2009-2014 average, respectively. Monetary value converted from Brazilian Real (BRL) to US Dollar (USD) using the average 2014 BRL-USD exchange rate of 3.2 (www.xe.com). [a] At normal (design) operating rate of 0.64 $m^3 \cdot s^{-1}$ (2014 year-on-year operating rate was 0.69 $m^3 \cdot s^{-1}$). [b] Foregone marginal revenue from sale of water of BRL 6.08 m^{-3} (Appendix D); applies only in peak season. [c] Marginal water price reduced for high-season processing water losses of 17%; applies only in peak season. [d] Collected for free by third party.

Table 2: Current and modelled 2025 TSS concentration at EMASA raw water intake in counterfactual and intervention scenarios

Scenario	annual avg. TSS concentration ($mg \cdot l^{-1}$)
2012 [a]	149.6
2025 Counterfactual	125.3
2025 with interventions	107.5

Notes: [a] Based on 2012 land use and 2014 climate data.

Table 3: Estimated average annual sediment-related benefits of Camboriú PWS program, 2015-2045

Benefit	Average annual impact, 2015-2045	
	Quantity	Present Value (2014USD)
Avoided peak season water loss	77,400 m^3	71,400
Avoided PACI use	73,400 kg	13,520
Avoided polymer use	150 kg	270
Avoided off-peak water pumping	77,600 kWh	2,990
Avoided dredging	110 m^3	500
Reduction in dry sludge landfilling	640 t	5,820
Reduced treatment plant expansion [a]	345,000 $m^3 \cdot yr^{-1}$	7,760
Total		94,500 (102,300 [b])

Notes: [a] Applies to hypothetical avoided capital cost case only. [b] Including hypothetical avoided capital costs. Present values calculated using 3.85% discount rate. Totals may not add up due to rounding.

Table 4: Estimated average annual costs of Camboriú PWS program during 2015-2045, present values

Cost type	EMASA	Overall
	(2014USD)	
Organization and outreach (design phase)	5,150	8,500
Technical planning (design phase) [a]	890	2,850
Hydrologic monitoring	11,560	14,180
Landowner engagement	4,040	4,040
Intervention design, implementation, initial maintenance	65,020	65,020
Payments to landowners	16,760	16,760
Program management	23,520	56,180
Total	126,940	167,540

Note: [a] Cartographic, legal and hydrologic studies. Present values calculated using 3.85% discount rate. Totals may not add up due to rounding. Pre-2015 costs assigned to 2015-2045.

Table 5: Estimated present value ROI metrics of the Camboriú PWS program for sediment control in municipal water supply, for 30yr and 50yr time horizons

ROI metric:		Cost-effectiveness, TSS concentration reduction (mg $TSS \cdot l^{-1}$ per million USD) [a]		Cost-effectiveness, TSS mass removal (kg TSS per USD)		Benefit-cost ratio	
ROI for:	Avoided capital cost	30yr	50yr	30yr	50yr	30yr	50yr
Program overall	No	2.1	2.0	2.91	5.97	0.59	0.82
	Yes	2.2	2.0	3.05	6.23	0.63	0.86
EMASA	No	2.8	2.6	3.84	7.85	0.77	1.08
	Yes	3.0	2.7	4.09	8.32	0.83	1.14

Notes: [a] Average concentration reduction during full period. All dollar values in 2014USD present values using a 3.85% discount rate. 30yr, 2015-2045; 50yr, 2015-2065.

Appendix

A. ROI analytical framework for watershed conservation programs

Reliable ROI assessment of any natural infrastructure project requires application of an integrated framework that links the biophysical and economic spheres (Daily et al., 2009; Keeler et al., 2012; National Research Council, 2005). Such a framework must meet seven conditions:

1) *Focus on ecosystem services and clearly distinguish among ecosystem functions, services, benefits and values.* Ecosystem services are the outputs or aspects of nature that support human uses (Brown et al., 2007; Tallis and Polasky, 2009), such as clean freshwater flows used for municipal water supply. *Ecosystem functions* are the processes performed by ecosystem structure (Odum, 1962), such as soil retention. Distinguishing between functions and services is crucial because not all changes in ecosystem functions translate into changes in services, due to absence of beneficiaries, attenuation of impacts between intervention and beneficiary sites, or temporal mismatch between affected functions and service demand. *Benefits* in turn are the specific uses people make of ecosystem services, such as municipal drinking water supply. These benefits have *economic value*, which is the change in human wellbeing they produce (Boyd and Banzhaf, 2007; Brown et al., 2007), such as avoided cost of municipal water treatment, development of alternative drinking water sources, or water-related health effects;

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), to avoid double-counting the value of intermediate services (Boyd and Banzhaf, 2007; Johnston and Russell, 2011);

3) *Define services in benefit-specific terms* using metrics that reflect the service characteristics crucial to benefit generation (Boyd and Banzhaf, 2007; Keeler et al., 2012; Landers and Nahlik, 2013), because the value of one service unit (e.g., 1 cubic meter of water with reduced TSS concentration) often varies widely among different uses (e.g., municipal water supply *vs* crop irrigation *vs* hydropower *vs* swimming) and locations or over time (e.g., TSS concentrations in municipal intake water during high flows or high water demand);

4) *Use calibrated ecosystem service production functions* (National Research Council, 2005) that relate interventions (e.g., riparian revegetation) to target services flows and incorporate spatial and temporal attenuation;

5) *Use "counterfactual", without-the-project service flow baselines* to allow proper attribution of observed or modeled changes in service flows to the project (Blackman, 2013; Ferraro, 2009; Ferraro and Pattanayak, 2006);

6) *Use empirically-based benefit functions* for key service beneficiaries that quantitatively relate service flow level (e.g., TSS concentrations at municipal water intakes) to specific, actual benefits (e.g., avoided TSS removal [Valade et al., 2009] and disposal at the treatment plant); and

7) For monetized ROI analysis, *use appropriate valuation approaches* to quantify changes in human wellbeing associated with those benefits (Brown et al., 2007; Griffiths et al., 2012; Wilson and Carpenter, 1999).

Supplementary Information B: Water treatment plant processes affected by sediment and associated costs

An analysis of treatment operational processes identified five discrete processes impacted by sediment concentrations (Figure B.1).

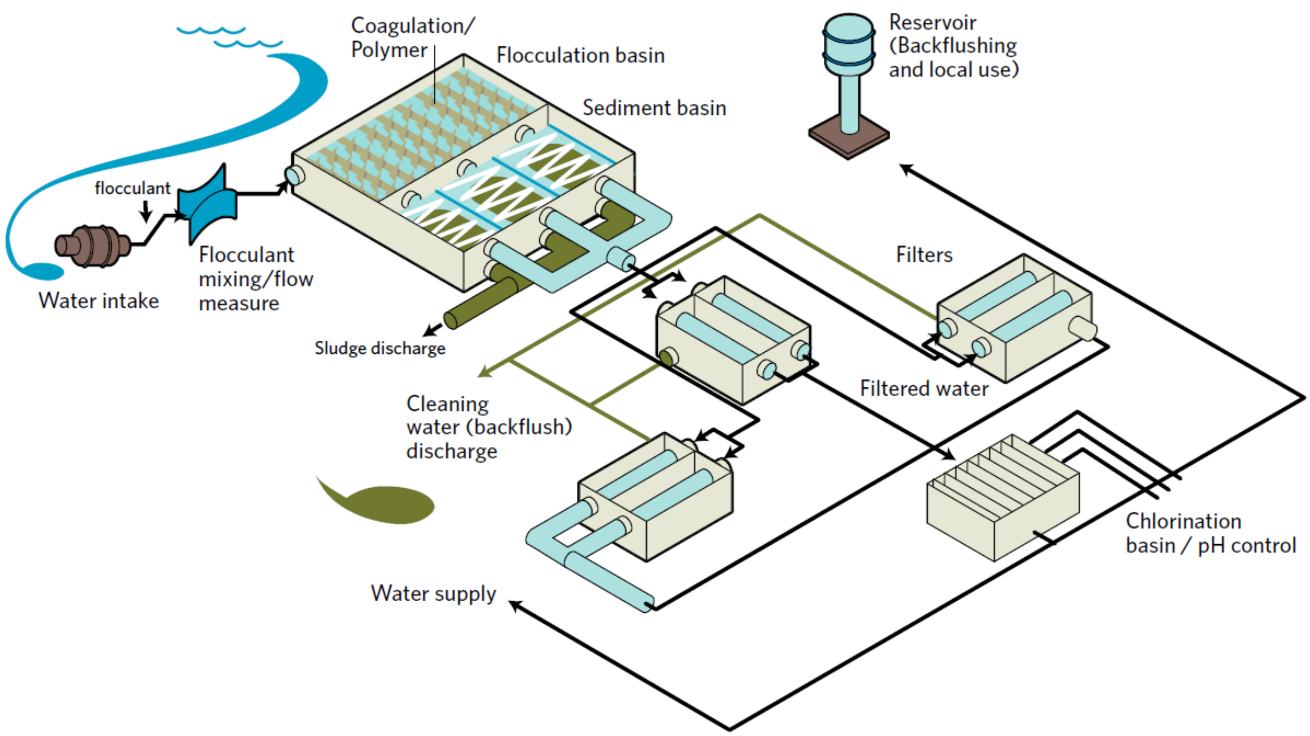

Figure B.1: Schematic representation of water treatment plant processes affected by sediment

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 2,000-2,500 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 (PACl), 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). PACl and polymer application are highly correlated with TSS concentrations in intake water (Figure B.2). However, our analysis uses average TSS loads in intake water to estimate annual impacts. We therefore assume both PACl and polymer use to change proportionally to TSS loads.

Figure B.2 shows a plot of total daily TSS loads entering the EMASA plant and total daily application of PACI for the year 2011.

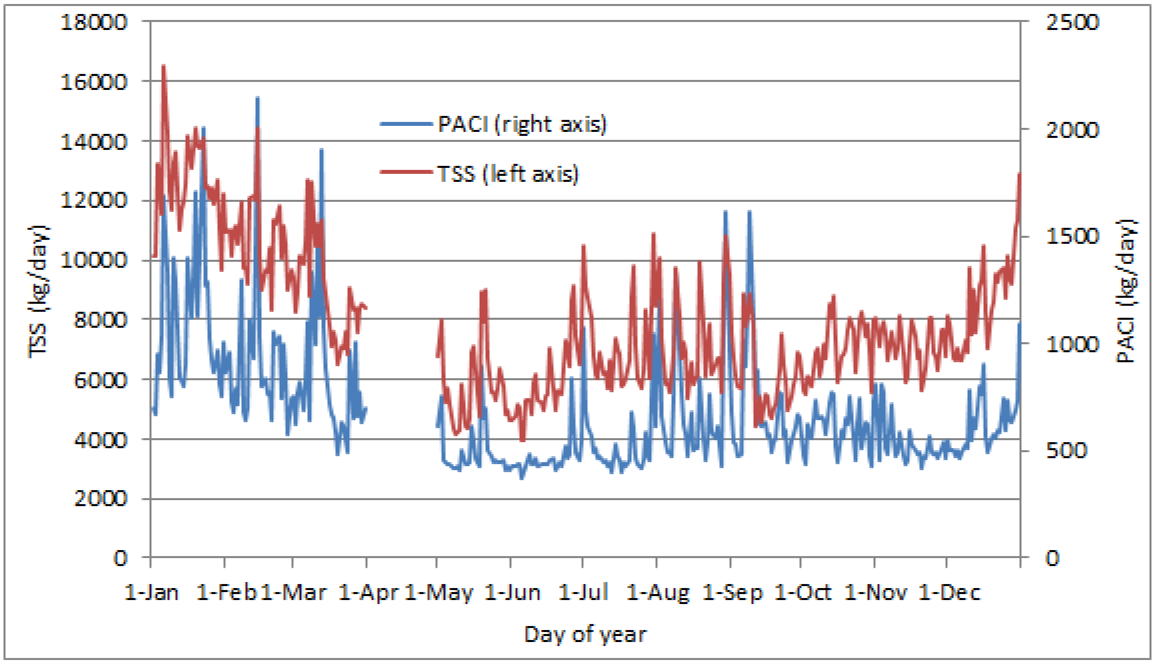

Figure B.2: Daily TTS load and PACI application in EMASA plant in 2011

Statistical analysis shows that TSS is a highly significant predictor of PACI use, with approximately 0.21 kg of PACI applied for each kg of TSS in intake water (adjusted R^2=0.945; $p<0.0001$; 95% CI=0.2050–0.2155).

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^{-1}). 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%. 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% of inflow. Using May 2014-August 2015 turbidity monitoring data at the water intake, the turbidity-TSS rating curve developed for the EMASA intake (Fisher et al. 2017) 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 % 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.

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. Given the fast-rising water demand, we assume that each final TSS filter will be backwashed on average 3 times per day using 350 m^3 per event, resulting in a total estimated annual water loss for filter backwashing of 766,500 m^3, or 4.2 % of total average annual 2008-2014 water intake.

Total sediment removal-related water losses thus sum to 5.9 % of intake water. Treatment plant data for 2008-2014 indicate that total measured water outflow is 15.5 % 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.

Appendix C: Land cover change analysis

Teixeira et al. (2009) found that change in Atlantic Forest cover in an area west of the city of Sao Paulo was affected by proximity to roads (higher forest regrowth far from dirt and main roads), urban centers (higher forest loss near urban cores), rivers (higher forest regrowth near rivers; higher deforestation far from rivers), slope (higher forest regrowth on steep slopes; higher loss on gentle slopes) and elevation (higher forest regrowth at higher elevations). The authors conclude that in recent decades, urban expansion into cities' hinterlands, in the form of both sprawl along the urban periphery and establishment of country homes, has become a major driver of net forest loss in the Sao Paulo area. This is also true for many other parts of the Atlantic Forest (Joly et al., 2014). Urbanization driven by rural depopulation and interurban migration from northern metropolitan areas also is a strong driver of LULCC in Santa Catarina state, which may have been experiencing a slight net increase in natural forest cover by the 1990s coupled with declines in pasture, crops and fallow, as well as an increase in plantations near coastal metropolitan areas (Baptista, 2008; Baptista and Rudel, 2006). With strong observational and anecdotal evidence of urban expansion, rural population decline, declines in pasture and increases in forest plantations in the Camboriú watershed, we expected that the drivers of LULCC there may be the same as those observed in other coastal Atlantic Forest regions.

Table C.1: Land cover transitions (2003 to 2012) included in LCM modeling

Area (ha)	Transition	% of total study area	% of total 2003-2012 change
266.8	Pasture to Forest	2.0%	24.7%
212.3	Forest to Pasture	1.6%	19.7%
135.7	Pasture to Plantation	1.0%	12.6%
97.2	Pasture to Rice	0.7%	9.0%
94.8	Pasture to Bare	0.7%	8.8%
93.7	Rice to Pasture	0.7%	8.7%
41.8	Pasture to Impervious	0.3%	3.9%
35.1	Bare to Pasture	0.3%	3.3%

Table C.2: Variables used to predict land cover change for all transitions

Variable	Data based on	Notes
Distance to bare	2004 land cover	Euclidean distance to 2004 bare land cover pixels
Distance to impervious	2004 land cover	Euclidean distance to 2004 impervious land cover pixels
Distance to pasture	2004 land cover	Euclidean distance to 2004 pasture land cover pixels
Distance to rice	2004 land cover	Euclidean distance to 2004 rice land cover pixels
Distance to roads	2003 roads data	Euclidean distance to small and large roads
Distance to urban	City boundaries manually traced from 2004 land cover	Allows distinguishing actual urban area from scattered impervious pixels
Elevation	Elevation data	Elevation data from DEM
Evidence likelihood of change	2004 & 2012 land cover	Generated via LCM

Table C.3: Net land cover change observed, 2003 to 2012

Land Cover	2004 (m^2)	2012 (m^2)	Net change (ha)	Net change (% of land cover class)	Net change (% of study area)
Water	399,168	456,430	5.7	14.35%	0.04%
Bare	1,547,718	2,324,451	77.7	50.19%	0.58%
Pasture	24,675,973	21,876,744	-279.9	-11.34%	-2.08%
Rice	9,055,034	9,044,171	-1.1	-0.12%	-0.01%
Impervious	106,069	610,447	50.4	475.52%	0.37%
Plantation	9,650,345	10,659,607	100.9	10.46%	0.75%
Forest	91,247,201	91,709,658	46.2	0.51%	0.34%

Table C.4: Predicted net land cover change, 2012 to 2025

Land Cover	2012 (m^2)	2025 (m^2)	Net change (ha)	Net change (% of land cover class)	Net change (% of study area)
Water	456,430	456,430	0.0	0.00%	0.00%
Bare	2,324,451	2,758,356	43.4	18.67%	0.32%
Pasture	21,876,744	19,083,305	-279.3	-12.77%	-2.07%
Rice	9,044,171	8,926,136	-11.8	-1.31%	-0.09%
Impervious	610,447	1,091,046	48.1	78.73%	0.36%
Plantation	10,659,607	12,417,658	175.8	16.49%	1.31%
Forest	91,709,658	91,947,977	23.8	0.26%	0.18%

Table C.5: Gross land cover change from 2012 to 2025, by transition

Area (ha)	Transition	% of total study area	% of 2012-2025 change
336.9	Pasture to Forest	2.5%	25.5%
313.0	Forest to Pasture	2.3%	23.7%
175.8	Pasture to Plantation	1.3%	13.3%
135.5	Rice to Pasture	1.0%	10.3%
123.7	Pasture to Rice	0.9%	9.4%
115.6	Pasture to Bare	0.8%	8.8%
72.1	Bare to Pasture	0.5%	5.5%
48.1	Pasture to Impervious	0.4%	3.6%

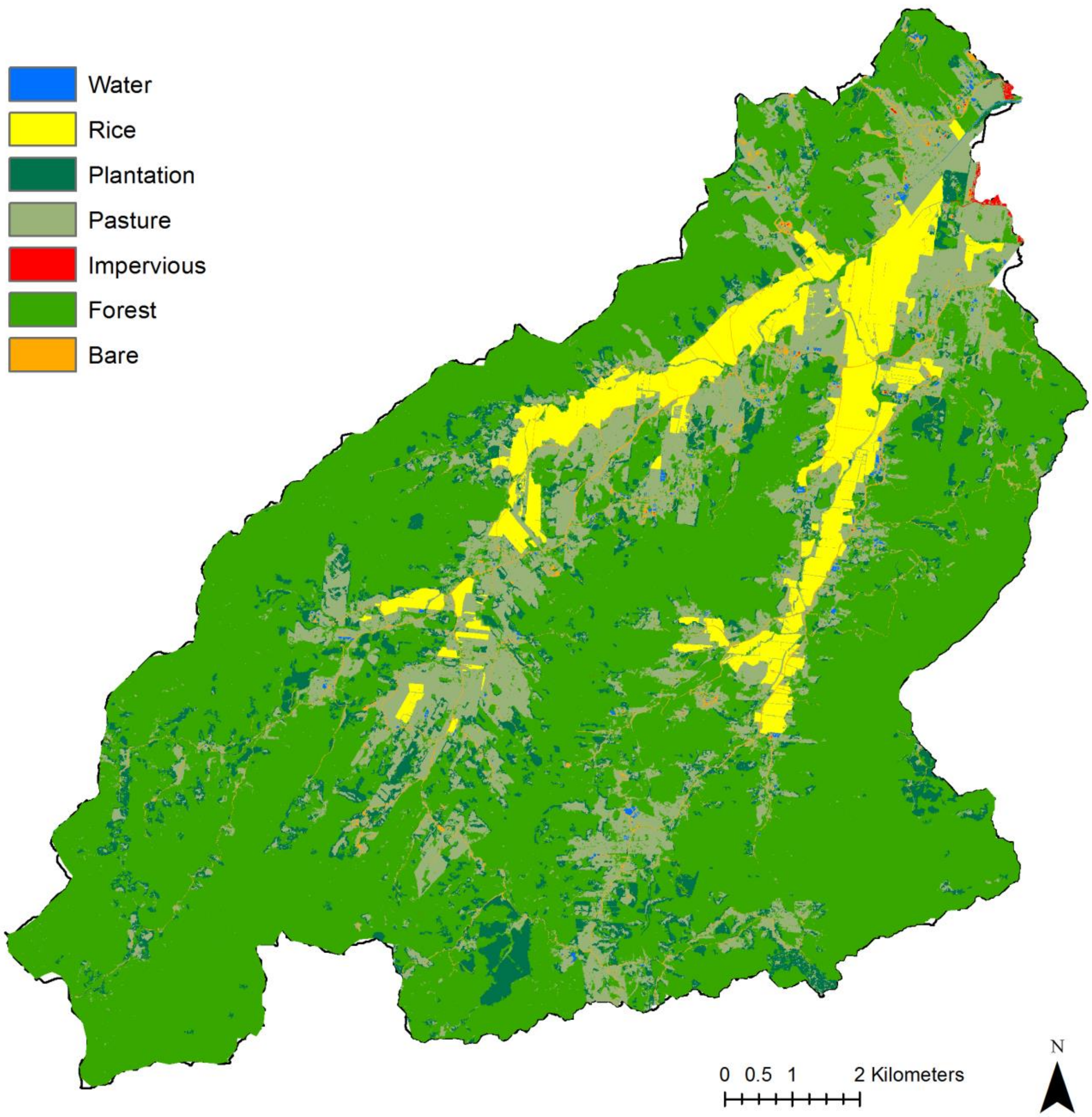

Figure C.1: Year 2003 land use classification in the Camboriú watershed upstream of the municipal treatment plant water intake

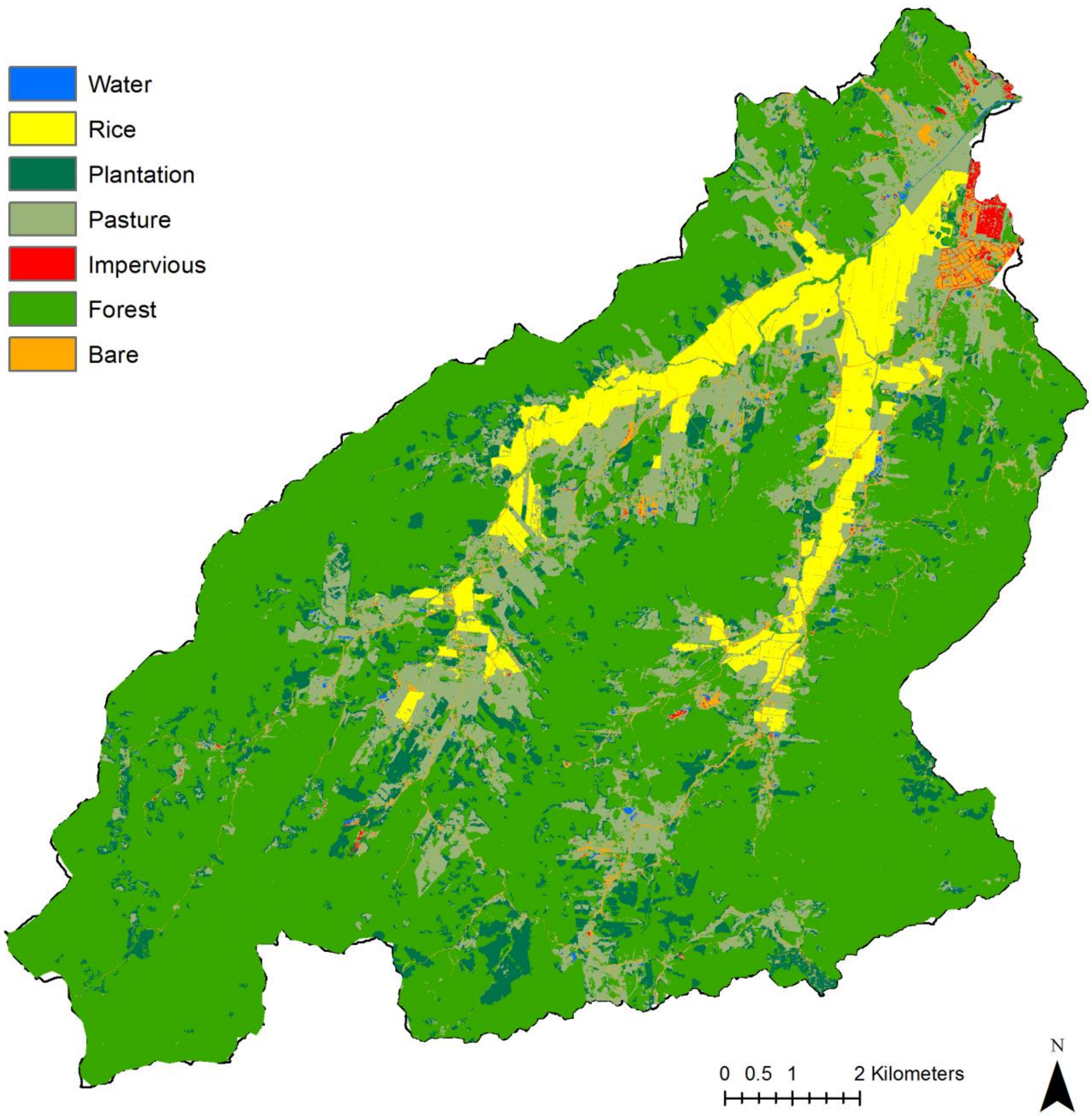

Figure C.2: Year 2012 land use classification in the Camboriú watershed upstream of the municipal treatment plant water intake

Appendix D: Marginal Water Price

To calculate the weighted mean marginal price of water sold by EMASA, we first calculate the mean marginal consumption-weighted price paid by each specific user category. This price is calculated from August 2015 data obtained from EMASA that shows total water consumption and water price by consumption level. We then calculate the consumption and user type-weighted overall marginal price (BRL 3.73 m^{-3}) of water as the consumption-weighted average of the weighted mean marginal prices paid by the different user categories. Table D.1 shows the data used in these calculations. Finally, we multiply the resulting weighted marginal water supply price by the ratio of August 2015 the average water supply (BRL 2.69 m^{-3}) to average sewer charges (BRL 1.70 m^{-3} water supply; automatically billed at 80 % of water supply by volume) to calculate the combined marginal weighted price per m^3 water supply and sewer. The latter is BRL 6.08, or USD 1.90 (at the average 2014 BRL:USD exchange rate of 3.2).

Table D.1: August 2015 consumption and price data by user category

User type	Use volume category ($m^3 \cdot month^{-1}$)	Actual use ($m^3 \cdot month^{-1}$)	Rate ($BRL \cdot m^{-3}$)	Weighted rate by user type ($BRL \cdot m^{-3}$)	($USD \cdot m^{-3}$) [a]
Commercial (normal)	0-10	18,380	2.87		
	11 to 20	8,196	3.9		
	21-999,999	62,480	4.5		
				4.11	1.28
Industrial (normal)	0-10	2,960	2.87		
	11 to 20	1,353	3.9		
	21-999,999	7,103	4.5		
				4.01	1.25
Residential (normal)					
	0-10	193,060	1.97		
	11 to 25	117,926	3.43		
	26-40	54,931	4.05		
	41-999,999	330,105	4.69		
				3.67	1.15

Notes: [a] All BRL:USD conversions based on average 2014 exchange rate of 3.2:1. All data provided by EMASA.

Appendix E: Hypothetical avoided capital cost from reduced treatment capacity expansion

In our hypothetical avoided capitol cost case, we estimate the cost of the recent expansion of treatment plant intake design capacity from 0.67 $m^3 \cdot s^{-1}$ to 1 $m^3 \cdot s^{-1}$ based on US construction cost data for this plant type, adjusted to the 2013 Brazilian price level using the 2013 ratio of Brazil's PPP conversion factor to the market exchange rate (World Bank, 2015) (Table E.1).[1] We use 2013 as the base year for the construction cost estimates since the construction of the expansion capacity began in that year. We then scale those costs proportionally to the reduction in the size of the treatment expansion that is equivalent to the avoided process water losses the PWS achieves during the peak season when demand is highest.

Table E.1: Estimated construction costs of EMASA treatment plant expansion by 0.33 $m^3 \cdot s^{-1}$

Cost item	2013 USD at Brazil price level
Alum feed system (Coagulation)	119,903
Polymer feed system	84,178
Rapid mix	79,295
Flocculation	354,075
Rectangular clarifiers	1,588,287
Gravity filtration	1,375,608
Surface wash	191,436
Backwash pumping	257,126
Wash water surge basin	766,077
Clearwell-below ground	815,282
Sand drying beds	n/a [a]
Subtotal	5,631,268
Site work, interface piping, roads (at 2.5%) [b]	140,782
Total construction cost	5,772,049
General contractor overhead and profit at 12%	692,646
Subtotal	6,464,695
Engineering at 10%	646,470
Total	7,111,165

Notes: Cost from table 16 in U.S. EPA (1979) for 5 million gallons per day (0.21 $m^3 \cdot s^{-1}$) drinking water treatment plant using conventional technology, scaled proportionally to 0.33 $m^3 \cdot s^{-1}$. Prices updated from 1978 to 2013 values using 1978-2013 US CPI inflation, and adjusted for US-Brazil price level differences in 2013 using the ratio of Brazil's PPP conversion factor to the market exchange rate in that year (0.745; World Bank, 2015). [a] Assumed additional sludge handled in existing drying beds. [b] Source assumes 5%, but we exclude roads since the EMASA plant already has road access and assume roads account for 50% of this cost item.

[1] EMASA records indicate that in 2012, the company allocated a total of BRL 20,000,000 (USD 10,000,000 at mid-2013 exchange rate) to the treatment plant upgrades. Half of this amount was obtained in the form of a Federal government loan at 6 % interest. The balance was made up by EMASA's reserves. According the company managers, most of this cost is caused by the expansion of treatment capacity.

Appendix F: Total intervention area and functionality over time

Table F.1: Intervention footprint and proportion of total final functionality of interventions achieved in each year

Year	2015	2016	2017	2018	2019	2020	2021	2022	2023	2024	2025	2026	2027	2028	2029	2030	2031	2032
Aggregate intervention area (ha)																		
Conservation	39	78	117	157	196	235	274	313	313	313	313	313	313	313	313	313	313	313
Restoration	41	82	122	163	204	245	285	326	326	326	326	326	326	326	326	326	326	326
Proportion of total final functionality attained (%)																		
Conservation	13	25	38	50	63	75	88	100	100	100	100	100	100	100	100	100	100	100
Restoration	0	1	4	8	13	19	26	35	45	55	65	74	81	88	93	96	99	100

Notes: Total final functionality refers to the full TSS reduction each intervention achieves once it is implemented across the full target area and has attained full functionality.

Appendix G: Intervention impacts on sediment and flow

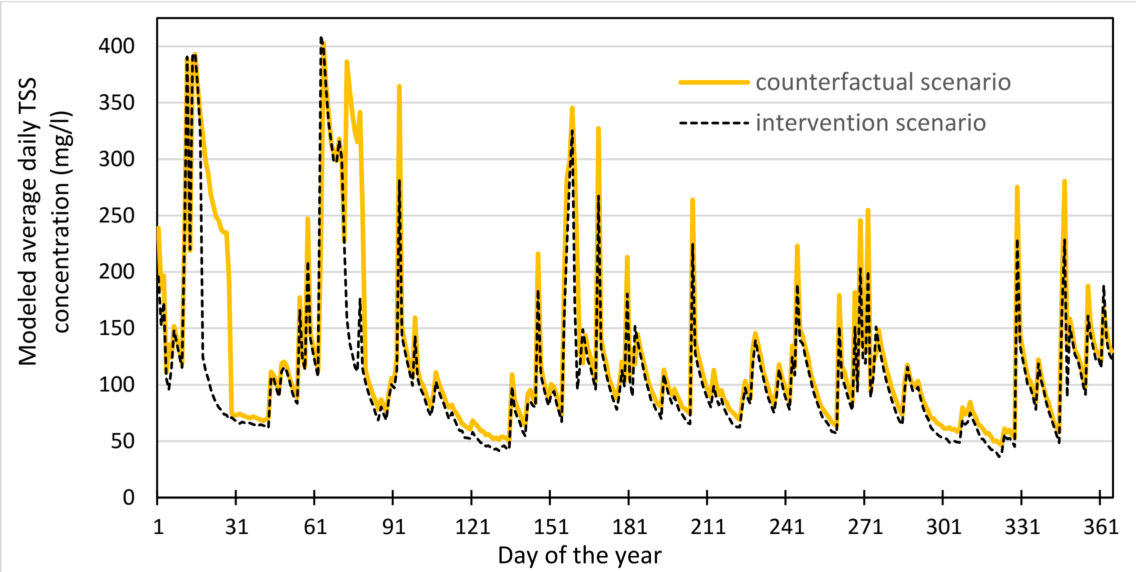

Figure G.1: Modeled average daily TSS concentration at the EMASA intake in the counterfactual and intervention scenarios using 2014 climate data, with full functionality of interventions

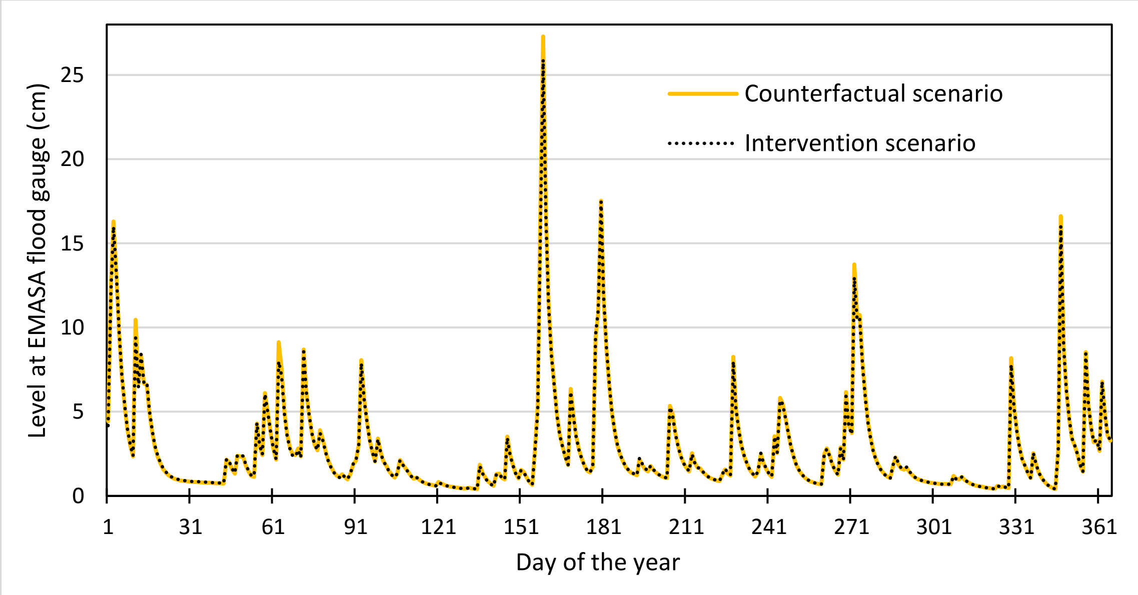

Figure G.2: Modeled river level at the EMASA gauge in the counterfactual and intervention scenarios using 2014 climate data, with full functionality of interventions

Appendix H: Benefits and Costs

Table H.1: Stated preference-based literature estimates of household willingness-to-pay for improved water quality, supply certainty or reduced flood risk in Brazil and Ecuador

Study	Machado et al. (2014)	Zapata et al. (2012)	Fuks and Chatterjee (2008)
Benefit valued	Avoid further loss of forest cover and associated ecosystem services	Enhance water supply (quality and reliability)	Reduce flood risk
Intervention	Watershed protection	Watershed protection	Flood control project
Average household WTP (2014USD/month)	2.10	7.13	10.63
Year of survey	2010	2005	1995
Study site	São Carlos, São Paulo, Brazil	Loja, Ecuador	Baixada Fluminense, Rio de Janeiro, Brazil
Mean household income (2014USD/month)	12%: <524; 51%: 761 to 3,281; other ranges not reported	1,059	343 [a]

Notes: [a] Study reports mean household income of respondents as 2.2 times the then-current minimum wage, BRL 100/month (https://tradingeconomics.com/brazil/minimum-wages). Converted to 2014USD using the 1995 BRL-USD exchange rate of 1 (https://tradingeconomics.com/brazil/currency) and 1995-2014 US Consumer Price Index inflator of 1.56.

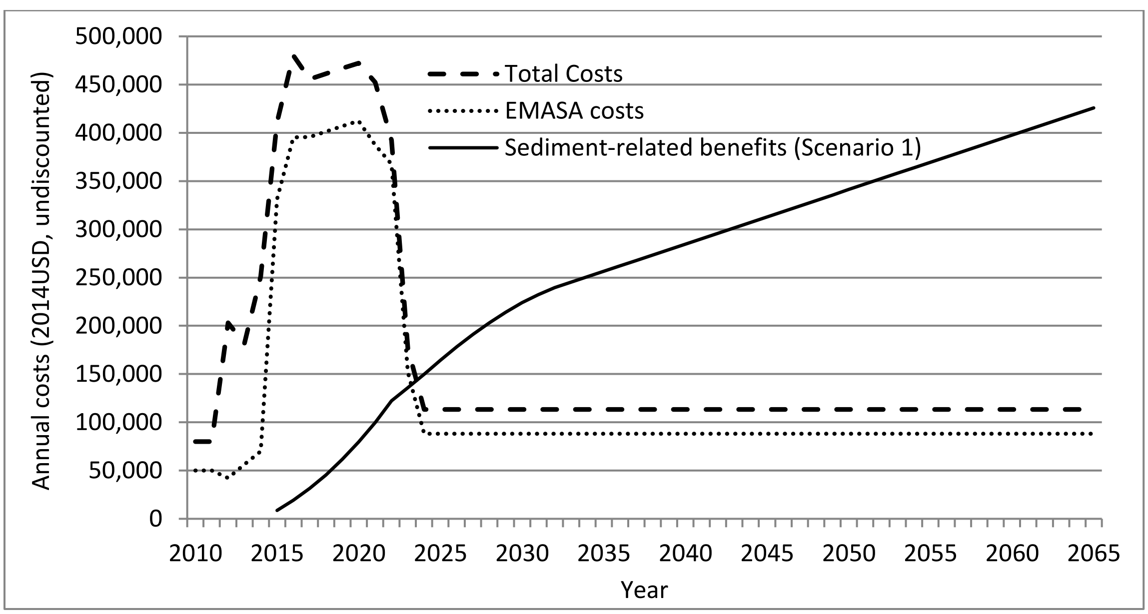

Figure H.1: Estimated annual costs and benefits of the Camboriú PWS program

Table H.2: Estimated average annual benefits of the Camboriú PWS program, 2015-2045 (undiscounted)

Benefit	Average annual impact, 2015-2045	
	Quantity	Value (2014USD)
Avoided peak season water loss (m^3)	77,400	147,000
Avoided use of PACI (kg)	73,400	27,800
Avoided use of polymer (kg)	150	560
Avoided off-peak water pumping (kWh)	77,600	6,100
Avoided dredging (m^3)	110	1,050
Reduction in dry sludge landfilling (t)	640	12,000
Reduced treatment plant expansion ($m^3 \cdot yr^{-1}$) [a]	345,000	7,800

Notes: [a] Applies to hypothetical avoided capital cost case only.

Table H.3: Estimated average annual costs of the Camboriú PWS program, 2015-2045 (undiscounted)

Cost type	EMASA	Overall
Organization and outreach (design phase)	4,589	7,581
Technical planning (design phase) [a]	853	2,708
Hydrologic monitoring	20,000	23,266
Landowner engagement	4,516	4,516
Intervention design, implementation, initial maintenance	77,189	77,189
Payments to landowners	30,560	30,560
Program management	38,387	82,437
Total	176,094	228,258

Note: [a] Cartographic, legal and hydrologic studies. Present values calculated using 3.85% discount rate. Totals may not add up due to rounding. Pre-2015 costs assigned to 2015-2045.

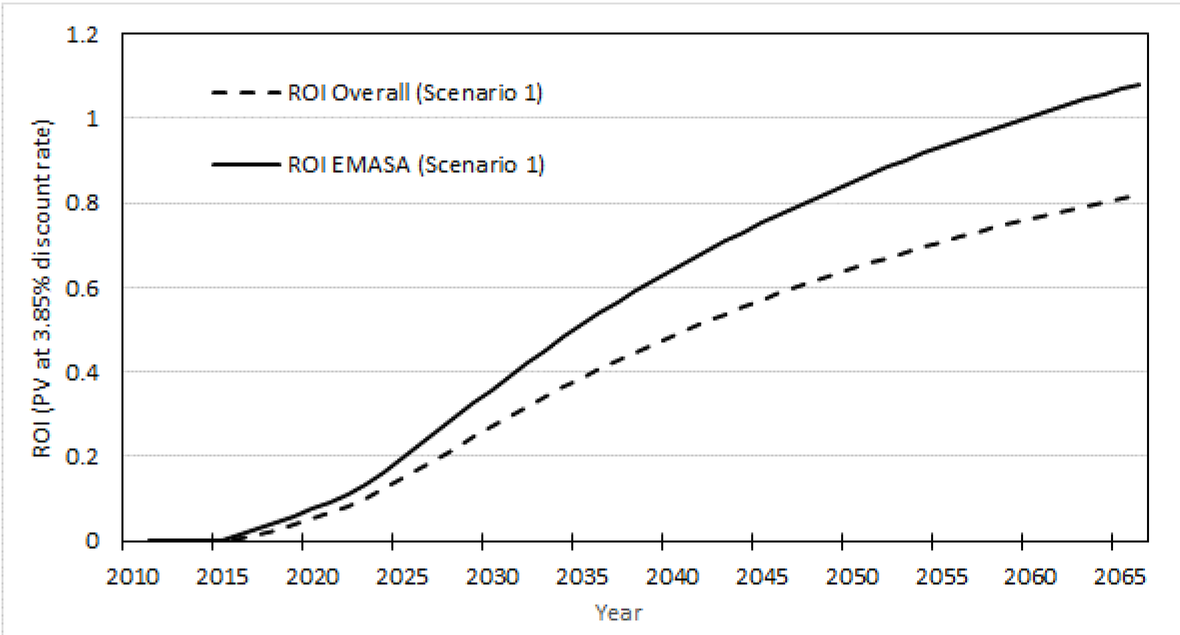

Figure H.2: ROI (PV benefit-cost ratio) of the Camboriú PWS program for EMASA and for the program overall. Only sediment control benefits are included in the analysis

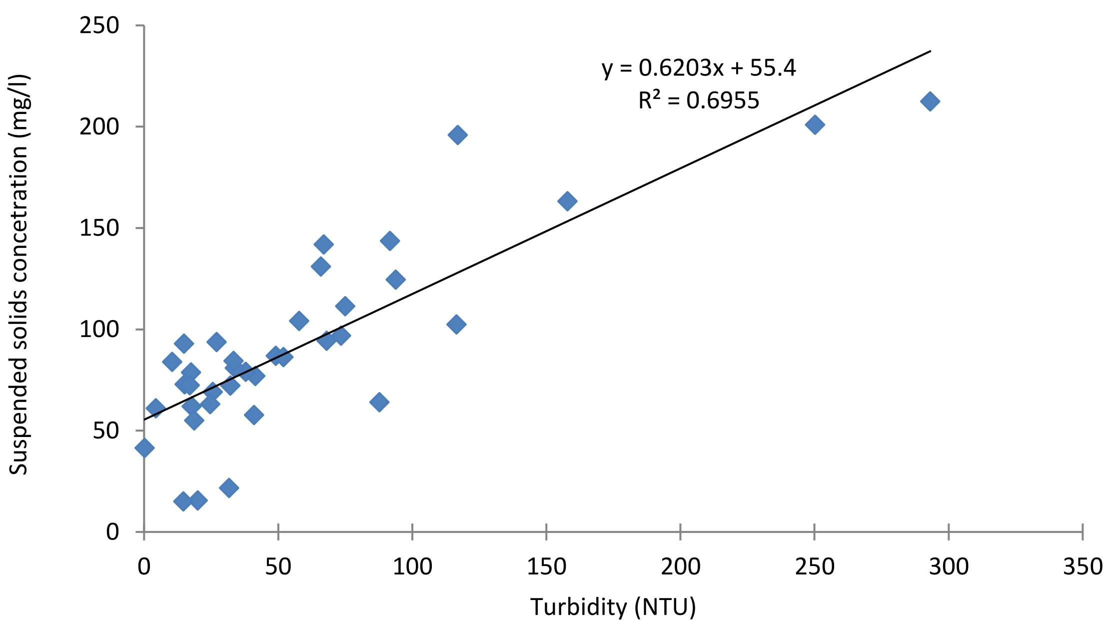

Figure H.3: Relation between turbidity and total suspended solids concentration at the EMASA water intake

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