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Assessing ecological infrastructure investments

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Conventional markets can underprovide ecosystem services. Deliberate creation of a market for ecosystem services [e.g., a payments for ecosystem services (PES) scheme] can close the gap. The new ecosystem service market alters behaviors and quantities of ecosystem service provided and reveals prices for the ecosystems service: a market-clearing equilibrium. Assessing the potential for PES programs, which often act as ecological infrastructure investment mechanisms, requires forecasting the market-clearing equilibrium. Forecasting the equilibrium is complicated, especially at relevant social and ecological scales. It requires greater disciplinary integration than valuing ecosystem services or computing the marginal cost of making a land-use change to produce a service. We conduct an *ex ante* benefit–cost assessment and forecast market-clearing prices and quantities for ecological infrastructure investment contracts in the Panama Canal Watershed. The Panama Canal Authority could offer contracts to private farmers to change land use to increase dry-season water flow and reduce sedimentation. A feasible voluntary contracting system yields a small program of about 1,840 ha of land conversion in a 279,000-ha watershed and generates a 4.9 benefit–cost ratio. Physical and social constraints limit market supply and scalability. Service delays, caused by lags between the time payments must be made and the time services stemming from ecosystem change are realized, hinder program feasibility. Targeting opportunities raise the benefit–cost ratio but reduce the hectares likely to be converted. We compare and contrast our results with prior state-of-the-art assessments on this system.

hydrology | ecosystem services | incentives | reforestation | natural capital

Conventional thinking suggests that society converts natural capital into produced capital like engineered infrastructure (1). However, as produced capital stocks have grown, and natural capital is scarcer, there is growing interest in reinvesting in natural capital in the form of ecological infrastructure (2), with the intent that ecological infrastructure substitutes for engineered infrastructure (3). Durable ecological infrastructure can provide a flow of "ecosystem service" outputs. In this way, nature stores wealth and passes it through time, acting as capital (4, 5). Reinvesting in ecological infrastructure requires institutions and mechanisms for investing in ecological infrastructure.

Feasible ecological investment depends on (*i*) the value and demand for the services that ecological infrastructure provides; (*ii*) the ability to change an ecosystem to produce a greater level of service; and (*iii*) the ability, right, and willingness for an entity to make changes to an ecosystem, that is, to supply ecological infrastructure (6–8). In many contexts, the beneficiaries of ecosystem services lack the ability to implement changes to supply, necessitating a voluntary contracting system that enables transfers from demander to supplier (9, 10). Such systems are called "payments for ecosystem services" (PES). In practice, the vast majority of PES contracts are "practice-based" and require one party to provide ecological inputs, ecological infrastructure, rather than idealized "performance-based" contracts where one party is obligated to ensure the delivery of a service or outputs (2). Irrespective of which type of contract is used, passing a benefit–cost test is a precursor for successful contracting programs (11). *Ex ante* benefit–cost assessment of a PES program needs to be conducted within the context of the likely market, at appropriate and feasible ecological and social scales, and conditional on data about that system to forecast the market-clearing equilibrium. Such empirical estimates capture market imperfections and barriers to participation likely to persist following PES implementation. This contrasts with current state-of-the-art assessments, which are conducted as if a central planner could fully internalize ecosystem services from ecological infrastructure investments and assuming away of other market imperfections or barriers (e.g., ref. 12).

A gap remains between concept and practice. Naeem et al. (13) lament the state of the biophysical data used to inform the ecological production functions that connect ecological infrastructure with ecosystem services. Ferraro et al. (14) argue that, "there have been few efforts to compare ecosystem service benefits with costs of service delivery." A small but important literature addresses this concern by estimating and projecting supply curves for land uses that provide ecosystem services, and forecasting the land-use market-clearing equilibria when landowners receive ecosystem service incentives (e.g., ref. 15). Such analyses are only benefit–cost analyses if incentive payments reflect empirical estimates of demand for ecosystem services based on a clearly defined buyer with a willingness to pay for the marginal ecosystem service flows. Benefit–cost analyses remain rare. We believe that the nuance between cost-effectiveness analyses or simulations that identify land-use market-clearing equilibria and benefit–cost analysis that uses ecosystem service market-clearing equilibria leads Scheufele and Bennett (16) to report that they fail to find a study that comparably estimates "demand and supply to determine the quantity and price of ecosystem services provision."

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One reason why studies forecasting market-clearing equilibria are rare is that it remains challenging to assemble the necessary expertise to integrate the multiple dimensions of PES programs to forecast the prices and quantities in equilibrium. Teams must go beyond valuing ecosystem services or computing the marginal costs of production. Teams must also estimate the biophysical and social extent of the "market"; account for the delay between ecological infrastructure investment and service provision; and measure the opportunity costs of providers, which may differ greatly from marginal production costs (2).

We forecast the market-clearing prices and quantities of an ecological infrastructure investment project in the Panama Canal Watershed (PCW), which informs the feasibility of the market-based PES mechanisms for this system. Doing so required assembling a multidisciplinary team capable of linking best practices in hydrology and economics. We proceed by first focusing on a well-defined service where the ecological infrastructure aims to enhance the ability of ships to transit the canal in the dry season by increasing dry-season water availability and reducing sedimentation. Second, we clarify that the Autoridad del Canal de Panamá (ACP; Panama Canal Authority) can capitalize the "end-of-pipe" services from ecological infrastructure investment and is expected to bear their cost. We draw on the economic valuation literature to identify the marginal value of a unit of ecosystem change. Using data from within the system, we determine the ACP's likely willingness to pay for such changes. We avoid benefits transfer, addressing another of the concerns of Ferraro et al. (14) about the state of economics within the ecosystem services literature. Third, we build on the literature of ecological production functions (17, 18), the primary focus of the ecosystem services literature. We use hydrological data from within the system to nonparametrically estimate the ecological production function. We avoid structural identifying assumptions based on model form and parameters from outside our study system, addressing other common critiques of the ecosystem service literature (13, 19). Fourth, using the best practices in preference elicitation (20), we survey land-use decision makers who make decisions affecting approximately a fifth of the unforested land in the watershed. We collect "landowner" responses to contract offers to establish an ecological infrastructure supply curve based on landowner opportunity costs. We use the term landowner as shorthand for land-use decision maker, and do not mean to imply clear title. In this sense, landowners are people able to exclude others and make decisions about a plot of land.

We find that land-use changes that act as ecological infrastructure to supply water to the Panama Canal are justified, a qualitatively different result than reported by Simonit and Perrings (12). However, the projected market-clearing equilibrium implies that only a fraction of the watershed will participate in the PES. The scope for investing in ecological infrastructure with PES is narrow.

Benefit–Cost Considerations for the *ex Ante* Assessment of Ecological Infrastructure in the PCW

Ex ante benefit–cost analysis requires measuring costs incurred for inputs and valuing outputs. Connecting inputs and outputs requires an empirical understanding of the ecological production function. We combined these three pieces of information to examine if a benefit–cost criterion could be satisfied. We pay close attention to the institutional context. First, we measure the marginal benefits of the service. Then, we determine how the ecosystem needs to change to produce the service. Third, we estimate the cost of that change. Fourth, we determine the market-clearing equilibrium. Finally, we consider if alternative, lower cost, means can produce the same services.

Benefit–cost assessment for ecological infrastructure is challenging in practice despite being straightforward in concept. An application helps illustrate the process. Incentivizing agroforestry programs within the PCW to increase dry-season water supply and reduce sedimentation provides a case study. The PCW has been extensively studied with respect to ecosystem services (12, 19, 21). Simonit and Perrings (12) conducted a state-of-the-art assessment of ecosystem services in the PCW, but their study is subject to the criticisms of Naeem et al. (13) and Ferraro et al. (14). We show how addressing the criticisms of Naeem et al. (13) and Ferraro et al. (14) through multidisciplinary collaboration, leveraging the comparative advantage of each discipline, can lead to qualitatively different results than standard practice (e.g., ref. 12).

Panama Canal ship traffic accounts for ∼3% of annual world maritime commerce (22). Ships transiting the canal pay tolls to

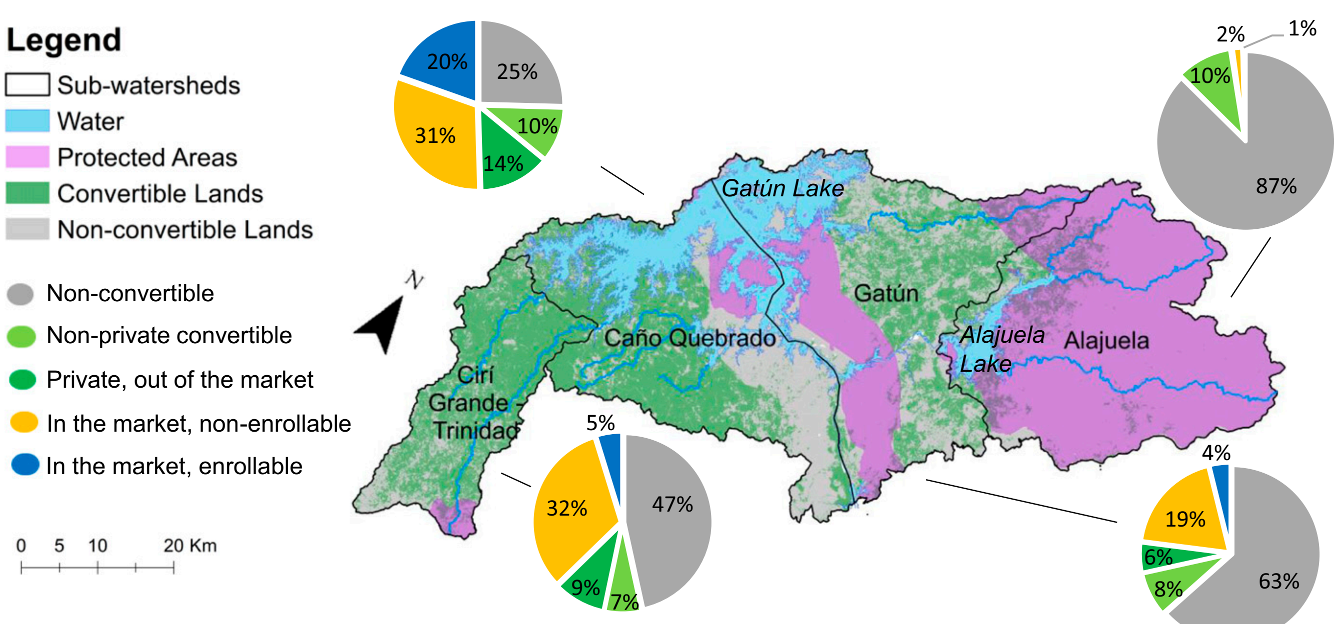

Fig. 1. Map of the PCW showing the four subwatersheds considered in our analysis, as well as the physical and social extents of the market. The middle black line is on the Panama Canal.

the ACP. The ACP contends with the prospect of low water levels in the 30,000-ha Gatun Lake (Fig. 1). Historically, the ACP formally restricted draft when the lake fell below 24.0 m (*SI Appendix*, Fig. S1), but it has often required weight reductions during dry months, linking lake height to revenue. New locks, which allow larger NeoPanamax ships, exacerbate the potential losses from low lake levels. Fully loaded NeoPanamax ships require 25.9 m (*SI Appendix*, Fig. S1). The ACP pays to dredge the lake to remove sediment (23).

Working agroforestry land uses [i.e., production forests, agroforestry, silvopastoral lands (collectively referred to as agroforestry)] can serve as ecological infrastructure by enhancing dry-season water supply relative to traditional pasture, which lacks trees (24). Agroforestry enhances bioturbation, resulting in increased wet-season infiltration that recharges groundwater; this water increases streamflow in the dry season. This is called the forest sponge effect (25). Agroforestry enhancement of the sponge effect can occur within 20 y in cloud forests (26, 27). An "enhanced sponge effect" (ESE) process serves as an ecological production function connecting land-use change to a valued service. Agroforestry projects can also stabilize soil, reducing sedimentation (28).

The ACP does not directly control most land in the watershed. However, the ACP can contract with landowners to change pasture to agroforestry. Contracts require the ACP to provide payments and in-kind services to the landowners. Feasibly, the ACP can pay for land-use change, an ecological infrastructure input, but not water supplied or sediment avoided, ecosystem service outputs. The key question is if landowners will supply land-use changes under contracts that the ACP is likely to offer. This is a first-order test of feasibility of ecological infrastructure, but the ACP may have other ways of increasing water in Gatun Lake. Importantly, the ACP only capitalizes on a fraction of the total environmental and social benefits potentially generated by shifting traditional pasture to agroforestry. Including other environmental and social benefits or mechanisms for increasing water supply alters the benefit–cost analysis.

The ACP's Demand for Ecosystem Services

The ACP capitalizes water quantity, in terms of lake height, and reduced sedimentation through shipping tolls. Both are private benefits to the ACP. The ACP manages water in the canal for shipping, drinking water, and electricity generation. However, ship transits are the major source of revenue and the process for capitalizing the marginal unit of water. Canal tolls consist of a fixed cost per vessel and a weight-dependent variable cost. Low lake levels can require the ACP to decrease the number of transits and tonnage of ships crossing the canal.

There are two ways to think about the ACP's desire for water. First, an economic approach focuses on the marginal contribution of a centimeter of lake level to revenue. This provides a demand function for water. It is comparable to the costs of acquiring water: a benefit–cost analysis. Second, an engineering approach asks how much water is needed to hit a physical target (e.g., fully avoid draft restrictions). The engineering approach does not lead to a benefit–cost question. If the cost of avoiding draft restrictions is sufficiently high, then it may not be in the ACP's interest to avoid the draft restrictions.

The ACP requires water to maintain the depth of Gatun Lake and to fill the locks, which uses water from the lake. Lake depth constrains the draft of ships that can pass through the canal. Regressing revenue on lake depth may lead to biased and inconsistent estimates of the marginal effect of lake level on toll revenue because lake depth may be endogenously determined. This form of endogeneity is a major concern in the economics literature (29). Rainfall is plausibly exogenous and is used in the first stage of an instrumental variables regression to purge the endogeneity (*Methods* and *SI Appendix*). This enables the second stage to consistently estimate that a 1-cm increase in Gatun Lake levels above the average dry-season level results in a \$164,700 (US dollars) per month revenue increase ($P < 0.01$; *SI Appendix*, Table S1). Robustness checks of the linear specification (*SI Appendix*) fail to find evidence of nonlinear marginal effects. Simonit and Perrings (12) avoid benefits transfer and suggest the same value concept of water. However, they calculate the value of water by dividing a year's toll revenue by the cubic meters of water that flow through the locks. This is not the marginal value of water because there is always enough water to fill the locks. The scarce, and therefore valuable, resource is water for the ships to transit Gatun Lake. The estimate of Simonit and Perrings (12) implies that the value of an additional cubic meter of water is an order of magnitude greater than our estimate.

Sedimentation imposes dredging costs on the ACP. Dredging costs average \$8.90 per cubic meter of sediment dredged (30). To construct the ACP's private demand curve for land-use change, we add the avoided dredging costs per hectare to the willingness to pay for water supply for the mean hectare by subwatershed (Fig. 2).

The Ecological Production Function That Links Ecological Infrastructure to Ecosystem Service

Feasible ecological infrastructure investment requires altering ecosystems in ways that change biophysical process to provide services. Dry-season water is the limiting factor. Gatun Lake cannot be filled to capacity during the wet season because capacity must be maintained to handle flood risk resulting from storms occurring late in the wet season. Excess water arriving in the lake in the wet season is released to the ocean. A forest sponge effect means that the groundwater system acts as an additional reservoir. An ESE is the percent change in dry-season streamflow in a catchment with forest land cover and an established bioturbation layer relative to a catchment without these features. Land uses generating an ESE deliver more water during the dry season relative to land uses not generating an ESE (Figs. 3 and 4 and *SI Appendix*, Fig. S2).

There are three ways to measure the ecological production. The first is a highly structural model grounded in physical theory (31). This approach requires a large number of system-specific parameters. Such models and parameters are not yet available for the PCW. Second, it is possible to make nonparametric estimates using the best available data from the system and not to rely on structural assumptions. We take this second approach

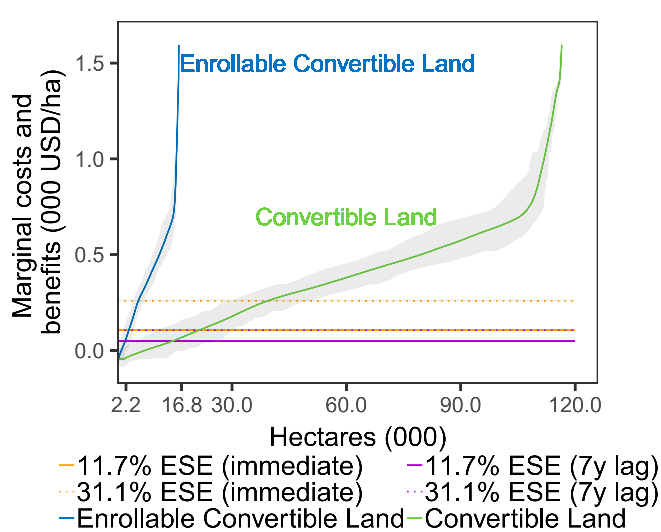

Fig. 2. Marginal cost for convertible land (physical extent of the market) and enrollable convertible land (social and physical extent of the market) with 95% confidence intervals, and marginal revenue under an average 11.7% ESE and an average 31.1% ESE (dashed and dotted lines, respectively) for the entire PCW. Marginal revenues are shown assuming an ESE is realized immediately (orange) and after a 7-y lag (purple).

(*Methods* and *SI Appendix*). The third approach is a highly parametric approach that relies on approximation rather than physical theory, such as curve number models. Simonit and Perrings (12) take the third approach, which can potentially provide high-fidelity phenomenological descriptions of a system. However, in the case of the Panama Canal, a curve number model requires a large number of parametric assumptions because parameters are transferred from outside the system. This is the essence of Naeem et al.'s (13) general criticism of the ecosystem service literature. Ogden and Stallard (19) provide specific criticism of Simonit and Perrings' (12) nonstandard application of a parametric curve number approach to the PCW.

The nonparametric ESE estimate compares streamflow data between matched catchments, that is, an old secondary forest and a traditional mosaic of swidden agriculture and cattle grazing on pastures cleared from forest (*SI Appendix*). Ogden et al. (25) provide 3 y of matched catchment data between forest and pasture in the PCW and find an ESE of 15.4%. Six years of data from catchments covering an older secondary forest and a mosaic land cover suggest an ESE of 11.7% (Fig. 3). We focus on this more conservative measure, but conduct extensive sensitivity analysis. These data suggest that over the course of the year, forests produce less water than mosaic or pasture, but the implication of agroforestry on the ESE and water supply only matter in the dry season. The curve number model of Simonit and Perrings (12) implies an ESE of <0% and that agroforestry cannot be thought of as ecological infrastructure for water supply in the dry season.

The level of Gatun Lake depends on water inputs as a function of land use, the size of the ESE, and canal management. We multiply per hectare normalized dry-season river discharge by the ESE (*SI Appendix*, Tables S2 and S3), scaled to the hectares converted to compute the change in dry-season water volume provided to Gatun Lake. We use hypsometric measurements to determine the increases in dry-season lake height (*SI Appendix*, Fig. S3). The ACP could leave all of the additional water in the lake (in situ use), allowing larger ships to pass; use all water for

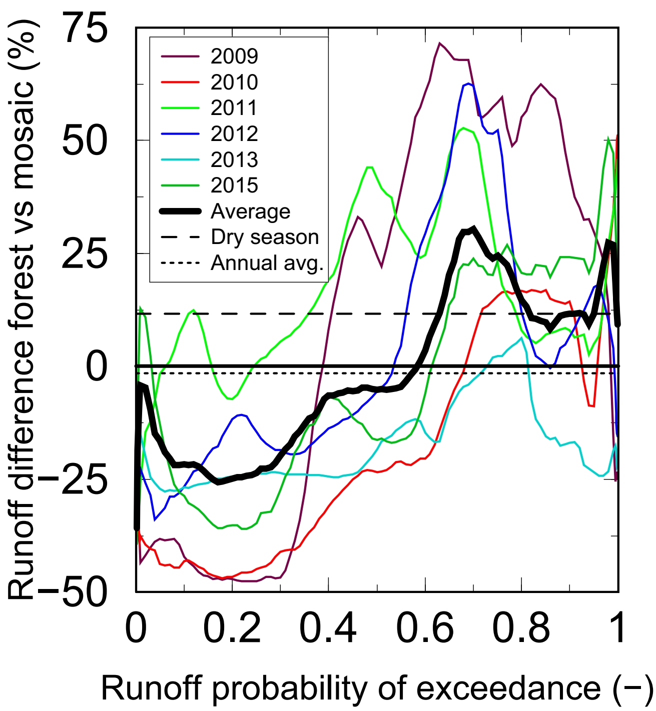

Fig. 3. Percent increase in runoff between forest and swidden mosaic by year, which provides the estimate of the ESE. The probability of exceedance is greatest in the dry season.

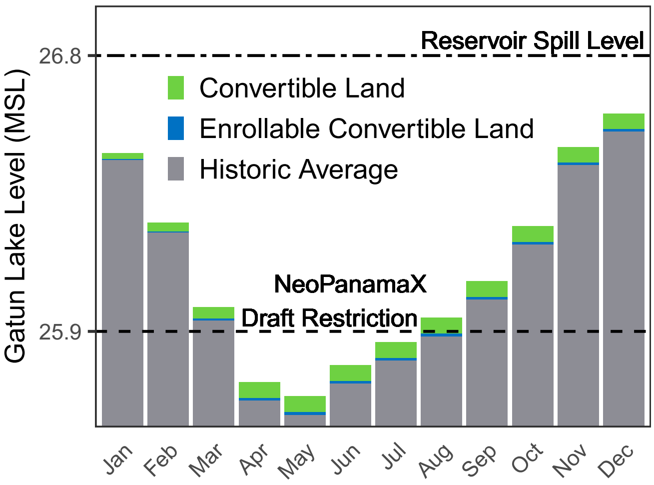

Fig. 4. Average monthly Gatun Lake levels (meters) by market extent assuming in situ water management and an ESE of 11.7%. The upper (double-dashed) and lower (dashed) lines are the reservoir spill level and NeoPanamax draft restriction level in meters above sea level (MSL), respectively.

transits as it arrives (extractive use), allowing more ships to pass; or a mix of these two strategies. We focus on in situ management, which mimics the operation of the NeoPanamax locks. Furthermore, following distance restricts the number of ships passing through the canal. Extractive water management is substantially less profitable than in situ management (*SI Appendix*). Under in situ management, water accumulates during the entire January–May dry season (Fig. 4) and acts as infrastructure, so contributions that increase lake level in January continue to provide services in February and so on.

The rate at which an ESE develops, following land-use change, is uncertain. Plant roots and soil fauna must reestablish. This creates a lag between ecological infrastructure investment and service flow. The best available data suggest that it takes 7 y to establish the ESE (32).

The ecological production function for avoided sedimentation uses Stallard's (33) estimates of equilibrium sediment yields for the six major river basins in the PCW (*SI Appendix*, Table S4). Comparing observed sediment yields with river basin land cover (34) provides subwatershed-specific estimates of reduced sedimentation from reforestation ranging between 0.7 and 66.0 $m^3 \cdot ha \cdot y^{-1}$ (*SI Appendix*, Table S5). Combining this information with the dredging costs suggests that the ACP's willingness to pay to avoid sedimentation from the Alajuela subwatershed is \$59 $ha \cdot y^{-1}$, falling to \$6 $ha \cdot y^{-1}$ in the Cirí Grande-Trinidad and Caño Quebrado subwatersheds (*SI Appendix*, Table S5).

Landowners' Willingness to Supply Ecological Infrastructure

Landowners, not the ACP, bear the costs of changing pasture to agroforestry. The ACP can secure land-use changes by offering landowners voluntary contracts that include payments, extension services, and other benefits. The cost of securing ecological infrastructure is the ACP's cost of fulfilling these contracts.

Simonit and Perrings (12) used a different cost concept. They compared the net benefits with and without the ecological infrastructure program. Their standard approach assumed an institutional framing with a social planner who only contended with markets failing to provide the ecosystem service. If there are other missing markets (e.g., banking services) or if the distribution of a landowner's preferences differs from the broader population for nonmarket goods, then the social planner's and landowner's opportunity costs differ. The latter determines the cost the ACP faces, because it is not a social planner. Importantly, willingness to accept (WTA) estimates incorporate the behavior of the landowners in responding to voluntary contracts, and landowners are likely to know their opportunity costs and behavioral responses better than analysts or the ACP. Ordering the costs to ACP from lowest to greatest creates the supply curve (Fig. 2).

The supply curve shows that the supply of ecological infrastructure has an upper bound. Increasing payments for ecological infrastructure entices new landowners to accept a contract, but there are constraints that determine the upper bound of available ecological infrastructure. This is the extent of the market. Smith (35) argues that "Definitions of the extent of the market are probably more important to the values attributed to the environmental resources as assets than any changes that might arise from refining estimates of per-unit values." Increasing the contract offer will not necessarily lead to market entrants or innovation. Market entrants are limited by physical condition and potentially by cultural or social barriers.

The physical extent of the market, or convertible lands, includes lands in the watershed that are not forested or in urban uses. Thirty-eight percent of the watershed (116,520 ha) is convertible land (*SI Appendix*, Table S6) in the physical market (Fig. 1). Privately owned convertible lands are targetable through contracting. Public convertible lands are targetable by regulation and policing existing laws. Privately owned convertible land makes up 30% of the watershed (89,830 ha). In total, 14,341 ha (about 5% of the watershed) of the convertible lands are in protected areas; over 65% (9,669 ha) is in national parks draining into Alajuela Lake, which feeds the Gatun Lake through the Chagres River.

The social extent of the market, or enrollable convertible lands, includes lands enrollable through voluntary contracting. The land must have an owner who can engage in a contract, excluding nonprivately held lands. We also excluded lands owned by educational or religious organizations because focus groups suggested the transaction costs with these groups are substantially greater than with private landowners. We identified the social extent of the market by surveying 711 landowner households. These in-person surveys covered 19% of the private convertible lands in the watershed (survey description and land types are discussed in *Methods* and *SI Appendix*). Twenty-four percent of the watershed (71,750 ha) was privately owned by people willing to engage in a contract. Most respondents were willing to convert land. On average, survey respondents who were willing to participate in a program were willing to enroll 25% of their convertible land (*SI Appendix*, Table S7). Five percent of the watershed, 16,061 ha (the asymptote of the blue supply curve in Fig. 2), was enrollable convertible land, which is the intersection of the social and physical extents of the market (*SI Appendix*, Tables S6 and S7). Some participants were unwilling to participate irrespective of contract offer. Scope tests and follow-up questions revealed that 22% of respondents were uninterested in contracts at any price. Reasons offered included that they were "too elderly to participate," "did not want to change their land use," and "do not trust organizations." Seventy respondents rejected contracts offered but were considered within the social extent of the market because they rejected contracts for specific contract features, such as a noncompliance penalty or loan structure, or stated that the "contract would not cover the cost." The total annual cost of contracting all enrollable convertible lands in an agroforestry program is $6.3 million, or $391 per hectare, on average.

People with lexicographic preferences against participation and the limited amount of land that participants were willing to enroll bound the social extent of the market. The social extent of the market can be substantially more restrictive than the physical extent of the market. Applying the supply schedule to the physical extent of the market suggests a program with nearly 7.5-fold as many hectares (Fig. 2, green curve).

The Market-Clearing Equilibrium and Benefit–Cost Test for Ecological Infrastructure in the PCW

A feasible ecological infrastructure program requires that the demand curve be above the supply curve for the first unit of ecological infrastructure. Market principles imply that the program will expand until the demand curve crosses the supply curve from above (Figs. 2 and 5). This is the market-clearing equilibrium, and it defines the feasible size of the program.

The size of the feasible ecological infrastructure investment, in the number of hectares enrolled, depends on the location and ownership of hectares. Assuming an 11.7% ESE, the gross benefits of converting all 16,061 ha of enrollable convertible lands to agroforestry is $551,000 per year or $34 per hectare per year, on average. However, cost matters. Under voluntary contracting, the ACP would offer and landowners would accept contracts that enroll 1,970 ha (12% of the potential market) at a cost of $48 per hectare. This assumes only enrollable convertible lands can be contracted, a 7-y lag from the time of investment until ESE effects are realized, that benefits and costs are uniformly distributed across the watershed, and that the ACP discounts benefits at its financing rate of 2.34%. We relax the uniformity assumption in the following paragraphs. This program is expected to add 0.03 cm of water, on average, to the lake per dry-season month, ultimately raising the 30,000-ha lake 0.12 cm at the end of the dry season (Fig. 4 and Table 1; extractive use is detailed in *SI Appendix*, Fig. S2 and Table S8). If the 11.7% ESE emerged instantaneously, then the ACP would offer compensation of $107 per hectare and enroll 2,940 ha (18%

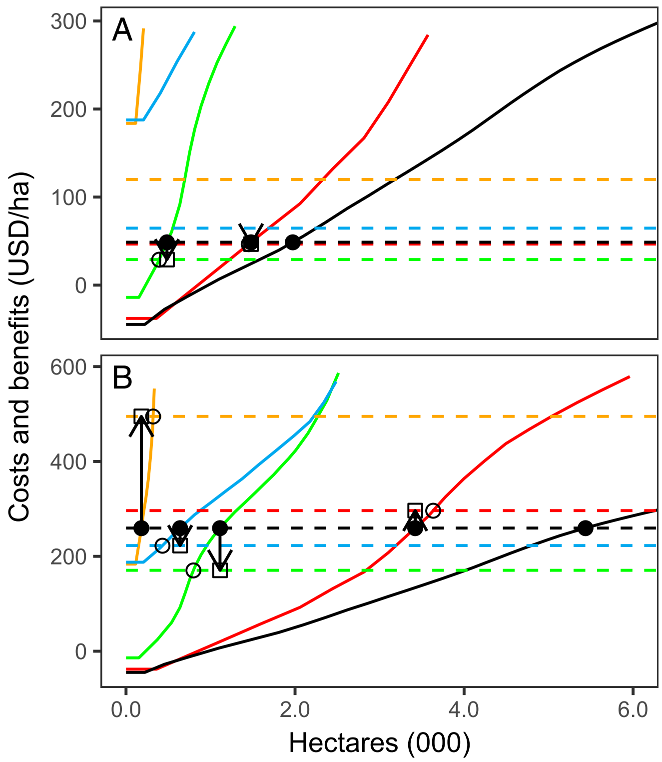

Fig. 5. Marginal benefit (dashed) and marginal cost (solid) curves for the entire PCW and each subwatershed for enrollable convertible lands with a 7-y lagged ESE of 11.7% (*A*) and 0-y lagged ESE of 31.1% (*B*). Voluntary exchange levels for subwatershed-specific programs (○) and the quantity of hectares provided under an aggregate program (●) are shown. The arrows connect the closed circles to the open squares, which illustrate the actual marginal benefit that would be provided by an aggregate program. Black correspond to the aggregate program. Colors correspond to subwatersheds: yellow is Alajuela, green is Caño-Quebrado, red is Cirí Grande-Trinidad, and blue is Gatun.

of the potential market), raising the lake 0.48 cm by the end of the dry season. A 31.1% ESE would be required to realize the 2,940-ha program if the ESE were delayed 7 y. A 31.1% ESE exceeds the largest ESE observed in the data (27.2%; *SI Appendix*, Table S2) and exceeds a reasonable upper bound estimate. This suggests that uncertainty about the development rate of the ESE may be as important as, or more important than, the size of the ESE. If the ESE were zero, then benefits would be restricted to avoided sedimentation, which justifies a program of 1,226 ha.

Extrapolating the supply curve to all convertible lands implies a program that would expect to enroll 14,960 ha and raise the lake level 0.92 cm by the end of the dry season at a price of $54 per hectare. Such projections would not be realized and illustrate the importance of the market extent for contracting programs.

It is possible to differentiate marginal benefits and costs by subwatershed. The eastern part of the watershed provides greater hydrological services than the western part (*SI Appendix*, Table S3). This shifts the marginal benefits curve for these subwatersheds up relative to the PCW average (Fig. 5). Landowners in the eastern watershed have a greater opportunity cost, which shifts the region-specific supply curves up relative to the aggregate supply curve (Fig. 5). Ignoring these differences leads to overestimating the net benefits from the agroforestry program. If contract offers aim to equalize the average marginal benefits (Fig. 5, dashed black line) with the aggregate marginal cost curve, where demand equals supply, then a voluntary program will not realize the expected benefits. The reason is that landowners in the eastern Alajuela and Gatun subwatersheds produce the greatest benefits per hectare, pulling up the average across the entire watershed. However, these landowners would not voluntarily participate if offered a contract priced at the average marginal benefit. Indeed, a program focusing on only these two subwatersheds would have a negative benefit–cost ratio. All hectares voluntarily enrolled would come from the western Caño Quebrado and Cirí Grande-Trinidad subwatersheds, which offer hectares where their respective marginal cost curves cross the black dashed marginal benefit curve (Fig. 5, black squares). The result is fewer benefits than expected. Benefits produced in each subwatershed must be read off their respective marginal benefits curve rather than the aggregate curve. Failing to account for heterogeneity in a voluntary contracting program leads to a 14% or $10,000 overestimate of net benefits, with a 7-y delayed ESE of 11.7% (Fig. 5*A* and Table 1). Spatial heterogeneity creates opportunities to target by offering different incentives by subwatershed. Equating subwatershed marginal benefits and marginal cost curves ($47 per hectare for Cirí Grande-Trinidad and $29 per hectare for Caño Quebrado; Fig. 5*A*, circles) increases net benefits but reduces the number of hectares enrolled in the program. Targeting increases the net benefits by $1,000 and increases the benefit–cost ratio from 4.0 to 4.9. Enrolling 120 fewer hectares improves the benefit–cost ratio, while achieving the same lake level increase. The benefits of targeting increase if the ESE is greater or develops more quickly (Fig. 5*B*).

Considerations Beyond Benefit–Cost Analysis for Contracting

Not all decisions involve benefit–cost analysis. If the focus is simply avoiding draft restrictions, then the ecological production function for enhancing the sponge effect does not offer a feasible path to fully avoiding draft restrictions (Fig. 4 and *SI Appendix*, Fig. S2). Policing existing land-use restrictions could secure ecological infrastructure. The Alajuela subwatershed is unique and provides substantial hydrological services. The Chagres River, in the Alajuela subwatershed, contributes ~60% of annual water volume to the lake. Only 20% of the convertible lands in the Alajuela subwatershed are owned privately (*SI Appendix*, Table S6). The remaining 80% are located within Chagres National Park, where agriculture operates without title. Converting all nearly 10,000 ha within the Chagres National Park to provide an ESE would provide enough water to raise the level of Gatun Lake more than converting all 16,061 ha of private enrollable land (Table 1). However, policing is also costly, and we have no estimate of policing costs.

The ACP has considered alternative ways to increase lake level (*SI Appendix*, Table S9). The proposed Rio Indio Dam could provide an additional 300 cm of lake depth at a direct cost of $610 million (36). Maintaining Gatun Lake at a maximum safe height suggests the dam's benefit–cost ratio is 3.1. This ignores political costs and restrictions that may make the dam infeasible. If the dam were feasible, it might be preferred to ecological infrastructure if the ACP only considers private criteria, because the dam passes the benefit–cost test and can operate at a much greater scale. However, the dam does not outperform ecological

Table 1. Area, increase in lake levels, net benefits, and benefit–cost ratio for the physical and social extent of the market with an 11.7% (31.1%) ESE with a 7-y lag in development for in situ water management

Market extent assumption	Area (1,000 ha)		Increase in cumulative dry-season lake levels, cm		Total net benefit ($1,000)		Benefit–cost ratio	
Convertible land								
Full market extent	116.52		6.04	(16.00)	−41,737	(−35,588)	0.1	(0.2)
Voluntary participation assuming uniform marginal benefits across subwatersheds	14.96	(12.41)	0.92	(3.48)	594	(1,554)	3.9	(3.1)
Voluntary participation assuming uniform marginal benefits but differentiated marginal costs by subwatershed	11.56	(15.82)	0.58	(2.14)	289	(971)	2.9	(3.0)
Enrollable convertible land								
Full market extent	16.06		0.87	(2.31)	−5,730	(−4,816)	0.1	(0.2)
Voluntary participation assuming uniform marginal benefits across subwatersheds and undifferentiated marginal costs	1.97	(2.94)	0.12	(0.48)	72	(212)	4.0	(3.2)
Voluntary participation assuming uniform marginal benefits but differentiated marginal costs by subwatershed	1.96	(2.85)	0.11	(0.44)	62	(206)	3.9	(3.3)
Marginal benefits equal marginal cost by subwatershed	1.84	(2.98)	0.11	(0.48)	63	(213)	4.9	(2.8)
Convertible land in Chagres National Park								
All convertible land in Chagres National Park	9.67		0.93	(2.47)				

infrastructure on the benefit–cost ratio criteria. Building the dam might restrict private ecosystem service benefits to avoided sedimentation.

Agroforestry programs may provide cobenefits that the ACP cannot directly capitalize, but may indirectly capitalize through reputation and public goodwill. For example, more complex landscapes can provide improvements in threatened species habitats, income redistribution, and enhanced livelihoods, and can reduce human mortality and property damage by reducing flood risks (37). If the bulk of the value were in cobenefits, then programs that directly target ecological infrastructure provision to generate these cobenefits, making them primary benefits, would be more effective. Carbon sequestration, biodiversity, and many other cobenefits have fundamentally different market extents, and there is no reason to restrict investments to provide these services to the PCW. Where general markets for these ecosystem services exist, supply tends to greatly exceed demand, and the markets are difficult to enter in practice (2). The ACP may choose to expand the program beyond the private market-clearing equilibrium if the ACP perceives the cobenefits to be large relative to the marginal costs, which are partially covered by capitalizable hydrological services. The ACP could expand the program up to 3,540 ha, while maintaining a benefit–cost ratio ≥1. However, cobenefits can only enter into determination of the market-clearing equilibrium if someone is willing to pay for them.

Discussion

Combining expertise from multiple disciplines to identify the market-clearing equilibrium in an *ex ante* benefit–cost assessment brings critical features to the fore. The payers need to be able to privately capitalize services, so finding the market-clearing equilibrium starts with marginal benefit and marginal cost measures determined by the actors in the system, subject to a broader social context. Most PES programs are ecological infrastructure programs, and this puts the onus on the buyer offering the contract to connect benefits and costs with an ecological production function. Investing in ecological infrastructure, relative to paying for ecosystem services outputs, also shifts the risks of getting the ecological production wrong from the supplier to the buyer. Focusing on contracting facilitates innovation in contract design to overcome barriers and reduce transactions costs (10). Moreover, focusing on the market-clearing equilibrium and contracting between actors provides a mechanism for analyzing the scale of the program. Scale is necessary to compare benefits and costs, something most assessments set aside (14, 16, 38, 39).

The scope for ecological infrastructure in the PCW is limited. Land-use change would not be sufficient to avoid draft restrictions completely. Nevertheless, a relatively small ecological infrastructure program provides private net benefits to the ACP. Sharply rising contracting costs, coupled with a small social extent of the market, drives the small size of the feasible program. The fact that payments are for infrastructure and there is a delay between land-use change and service provision drives up program cost.

The prior assessment of Simonit and Perrings (12) found that agroforestry would be a net liability if only hydrological services were considered. Hydrological assumptions, not economic or hydrological data, drive their result. Adopting our ecological production function, but applying Simonit and Perrings' social planner perspective (12), which only accounts for the physical extent of the market, would suggest a very large program. However, under realistic conditions, we find that only a small program is feasible.

Many analyses have illustrated that PES programs can generate environmental services, but these assessments tend to be narrow, identifying the "low-hanging fruit" without measuring how much low-hanging fruit exists. Identifying the extent of the market is imperative for achieving the policy goals because extent of the market may be the most important factor in determining the costs at a realistic scale. In the case of the Panama Canal, ignoring the extent of the market can result in greatly underestimating costs or overestimating expected service levels.

Contracting for ecological infrastructure is a common occurrence within the class of PES programs. Salzman et al. (2) emphasize the need to evaluate the existing programs. However, what should they be evaluated against? In some cases, asking whether the programs do anything at all is reasonable. However, for ecological production to be part of an investment calculus, it is helpful to know if ecological infrastructure can match *ex ante* expectations. If not, then planners need to consider structuring programs differently, adjusting expectations, or abandoning hope that ecosystems as infrastructure can play an important role in planning for the future. Having a sense of the likely market-clearing equilibrium associated with potential ecological infrastructure projects helps adjust expectations.

Methods

Ecological Infrastructure Benefit Estimation. We estimate the value of an increase in lake level by two-stage least squares and regress toll revenue against lake level instrumented by rainfall (details are provided in *SI Appendix*, Table S1). Robustness checks focusing on nonlinear effects are provided in *SI Appendix*.

We assume ESE delivers discharge continuously through the dry season (January–April) proportional to average monthly dry-season flow. During the dry season, river discharge inputs are often lower than lockage outflow, and the level of Gatun Lake drops. We consider two water management extremes: extractive use (presented in *SI Appendix*) and in situ use; both are constrained to lake volumes that never exceed storage capacity. The extractive use strategy employs ESE volumes to increase toll revenues by increasing transits and tonnage during the dry season. Water spills to the ocean via the lock system as it arrives. The in situ use strategy assumes transits remain constant and increased toll revenues come from increased tonnage. In this strategy, inflows are carried over until Gatun Lake reaches levels where spilling is needed to avoid wet-season floods.

We estimate expected benefits by calculating ESE-driven lake level rise conditional on management strategy and then multiplying by the marginal benefits, in toll revenues, of a 1-cm increase in lake level. We assume all land-use changes are successful and all hectares provide equal dry-season discharge within the subwatershed (*SI Appendix*, Table S2). ESE volumes are the product of ESE percentages and average monthly historic dry-season (January 1 to April 30) discharge supplied by enrollable convertible and convertible lands (https://micanaldepanama.com/nosotros/cuenca-hidrografica/anuario-hidrologico/). We convert volumes into increases in lake level using a water depth and lake volume relation for Gatun Lake (details are provided in *SI Appendix*, Fig. S3). The distribution of increases comes from applying the increases to 40 y of historic lake levels. In the analysis, the contribution of each hectare to dry-season discharge is assumed to be heterogeneous by, but not within, subwatershed (details are provided in *SI Appendix*, Table S3).

For each river basin in the PCW, we account for reductions to surficial and deep erosion resulting from agroforestry conversion (33). We divide the difference between Stallard and Kinner's (34) annually measured and equilibrium sediment rates by the basin's nonforested land area (33). The resulting basin-level values are per hectare estimates of the annually avoided sediment yields from surface and landslide erosion (*SI Appendix*, Table S4). These values provide a more accurate measure of avoided sedimentation benefits than a surficial erosion model (33). Subwatershed sediment yield reductions are calculated from area-weighted averages of the river basins located within their boundaries (*SI Appendix*, Table S5). Avoided sediment originating from the river basins of the Alajuela subwatershed are adjusted to reflect the 90% sediment trapping efficiency of Alajuela Lake and Madden Dam (40). We convert sediment mass to volume using a factor of 1.2 $Mg{\cdot}m^{-3}$ (33). We apply the average dredging costs to the expected sediment volume reductions of agroforestry adoption (*SI Appendix*, Table S5). We acknowledge that actual sedimentation contributions vary by localized conditions. However, it is not clear that there is a directional bias, and most convertible lands are within 112 m of water bodies. Nevertheless, greater targeting for sedimentation benefits may be possible.

Extent of the Market. We utilize the most recent land cover data (41) for the PCW and protected area data of the Panama Ministry of Environment (42) to identify the physical extent of the market, which accounts for current lands, public and private, available for conversion (Fig. 1). (Digital data were provided by the Unidad del Sistema de Informacion Geografica de la Autoridad del Canal de Panama. The document has not been verified by the ACP, and is not an official document of that entity.) In 2013, the 303,755-ha land area of the watershed was classified into nonconvertible lands [mature forests (27.6%), secondary forests (27.1%), forest plantations (2.5%), urban areas (3.5%), bare soils (0.5%), agroforestry (0.3%), and mining (0.3%)] and convertible lands [pasture land (25.0%), young regenerating forests that form part of the shifting cultivation cycle (10.1%), invasive canal grass *Saccharum spontaneum* L. (2.2%), and commercial crops, (e.g., pineapple) (1.1%)]. We account for variation in convertible land across four subwatersheds of the PCW: Alajuela, containing the Chagres, Pequeni, and Boquerón Rivers; Gatun, containing the Gatun River; Cirí Grande-Trinidad, containing the Cirí Grande and Trinidad Rivers together; and Caño Quebrado, which includes the Caño Quebrado, Hules, and Tinajones Rivers. Private convertible land outside of protected areas is found using an adjustment factor of 88%, taken from a sample land title coverage (43), and extrapolated to the watershed (*SI Appendix*, Table S6).

An in-person survey of a spatially representative sample of landowners covering approximately a fifth of the nonforested lands ($n = 711$) in the PCW provides the information on participation and enrollment used to identify the social extent (*SI Appendix*, Table S7). The survey was conducted following the principles of Johnston et al. (20) (details are provided in *SI Appendix*). The sample frame was landowners with at least 1 ha of convertible land. Respondents provided their WTA a contract for a land conversion incentive program and the amount of land they would enroll. To determine the amount of convertible land that could be enrolled in a program, we adjust all private convertible lands using the proportion of land owned by landowners accepting contracts and the proportion of each type of land those landowners were willing to supply (*SI Appendix*, Table S7).

Land Conversion Cost Estimation. Costs include landowners' WTA and the cost of technical assistance. We used stochastic payment card questions to estimate land conversion WTA. A landowner's WTA lies in the "switch point" interval defined by $1 greater than the greatest amount the landowner would not accept (lower bound) and the lowest amount the landowner would accept (upper bound) for a conversion program (more details are provided in *SI Appendix*). Technical assistance costs are estimated on a per landowner (farm) basis using data from Yale's Environmental Leadership Training Initiative, which has provided agricultural extension services in the watershed. We use the cost of implementing silvopastoral programs as a lower bound cost estimate.

The supply curves are created by sampling landowners from each subwatershed and then drawing WTA estimates from the switch point interval for each landowner (Figs. 2 and 5). Technical assistance costs were added to yield total land conversion cost. For each sample, random draw, and subwatershed, we order costs from lowest to highest. For each subwatershed, we compute the mean and 95% confidence of costs and mean hectares enrolled to create the supply curves, which are scaled up from our sample to represent the PCW population. Market clearing is found through a recursive process at the intersection of marginal benefits and marginal costs. The aggregate supply curve results from the treatment of the entire watershed as a single sample rather than the horizontal summation of the four subwatersheds. Total cost and total benefits are estimated as the area under the marginal cost and marginal benefit curves.

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