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Ecosystem complementarities: Evidence from over 700 U.S. watersheds

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ABSTRACT

This paper demonstrates how the slope of a production possibilities frontier (PPF) can be used to empirically identify the presence of an ecosystem externality. Complementarity between non-market ecosystem services implies the PPF between these services may be upward sloping. In contrast, private landowners that ignore these complementarities will treat non-market ecosystem services as competing products thereby ensuring a downward sloping PPF. Using data from 738 U.S. watersheds, we compare the slope of PPFs on predominately private and predominately public watersheds. We find evidence that private markets are failing to internalize complementarities between forest habitat and two other non-market ecosystem services: instream flows and grazing. In addition, we show how three-stage least squares regression can be used to reduce endogeneity bias associated with more common approaches to estimating ecosystem service relationships.

1. Introduction

Ecosystem externalities arise when a firm's production impacts another firm's production via a series of adjustments that alter the shape of the production possibilities frontier between ecosystem services (Crocker and Tschirhart, 1992). When ecosystem externalities are quantified, it is typically for a specific, small-scale ecological system making the prevalence and relevance of the externality unclear. A point of departure for many policy analyses is that these externalities exist and incentives are needed to correct the resulting under-provision of non-market ecosystem services (NMES). Based on an analysis of NMES relationships in 738 U.S. watersheds, this paper illustrates how ecological-economic production possibilities frontiers (PPFs) can be used to identify the existence of ecosystem externalities.

Relationships between marketed ecosystem services (i.e., commodities such as livestock and timber) and NMES (e.g., grazing, forest habitat) can be captured through joint production technologies in which the production of a marketed commodity is enhanced by a NMES (Nalle et al., 2004; Fisher et al., 2009; Kragt and Robertson, 2014; Sims et al., 2014). With these joint production technologies, firms have an incentive to provide NMES even though there is no market for them (Wossink and Swinton, 2007). Similar joint production technologies may also exist between NMES (e.g., Kandziora et al., 2013; Obiang Ndong et al., 2020; Spake et al., 2017). These joint production technologies imply an ecological-economic PPF (White et al., 2012; Polasky et al., 2005; Cavender-Bares et al., 2015; Bekele et al., 2013) that delineates the set of non-dominated ecosystem service that can be produced from a fixed set of inputs (here land). Much like a traditional PPF, an ecological-economic PPF is determined by the relevant joint production technologies. However, while a traditional PPF shows the tradeoffs and complementarities between market goods and services, an ecological-economic PPF shows tradeoffs and complementarities that involve non-market ecosystem services – either relationships between marketed ecosystem services and NMES or between different NMES. If NMES are substitutes, managing for one would force difficult tradeoffs. On the other hand, if NMES are complementary to each other, managing for one would produce win-win outcomes. Keeping open fields for pollinator habitat, for example, increases output of agricultural commodities but is contrary to natural regeneration of forests and ecosystem services like timber and flood prevention.

In a centralized market where commodities and NMES are jointly produced by a single firm, the ecological-economic PPF can exhibit non-convexities (Brown et al., 2011). In a decentralized market where commodities are produced by different landowners and firms, an ecosystem externality can arise that changes the shape of this ecological-economic PPF (Crocker and Tschirhart, 1992). Because firms have no

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incentive to incorporate complementarities between their jointly-produced NMES and NMES produced by other firms, traditional assumptions (i.e., limited factor inputs and decreasing returns to scale) dominate and create familiar tradeoffs between NMES. The ecosystem externality results in under-provision of the NMES and NMES are competing products.

We exploit these differences in the shape of the ecological-economic PPF, to provide empirical evidence that two ecosystem externalities associated with the supply of forest habitat is pervasive in the U.S. Using three-stage least squares (3SLS) regression to address endogeneity bias, we estimate pairwise relationships between the levels of three NMES in 738 U.S. watersheds: instream flows, forest habitat, and grazing. We compare results from watersheds with predominantly private ownership to those with predominantly public ownership where management for multiple NMES is typically mandated. We find complementary relationships exist in public watersheds for two of the three pairwise relationships. In contrast, all pairwise ecosystem service relationships are negative on private watersheds. We take these differences as evidence of an ecosystem externality that is effectively internalized by public management but persists in the private market.

This paper contributes to the general body of work estimating the relationships between ecosystem services (Lee and Lautenbach, 2016; Obiang Ndong et al., 2020; Cord et al., 2017 provide reviews). Existing studies generally take one of three approaches to characterize pairwise relationships: correlation coefficients, multivariate statistics (e.g., factor or cluster analysis), or OLS regression, and results tend to be sensitive to the chosen methodology (Vallet et al., 2018). One reason may be the assumptions and/or subjectivity inherent in each approach. Correlation coefficients assume linearity and ignore the role of potentially important drivers. Multivariate statistics often involve a large degree of researcher subjectivity (e.g., choosing the number of clusters). Finally, because ecosystem service levels are codetermined by complex ecosystem interactions (i.e., they are both independent and dependent variables), OLS estimates likely suffer from endogeneity bias (Cohen et al., 2019). Our study addresses these shortcomings by estimating the relationships as a set of reduced form simultaneous equations using three-stage least squares (3SLS). We note discrepancies between our 3SLS estimates and estimates obtained from correlation coefficients and OLS, calling into question the latter two methods.

In addition, our analysis is at a broader spatial scale than the majority of studies assessing ecosystem service relationships (Lee and Lautenbach, 2016). Typical studies are at landscape (e.g., Alemu et al., 2021) or regional (e.g., Lorilla et al., 2018) scale. In contrast, our dataset contains watersheds throughout the continental U.S., which offers two advantages. First, there is sufficient variation across study site characteristics that our conclusions are reasonably generalizable. Prior attempts to identify consistent relationships between ecosystem services relied on meta-analysis (Howe et al., 2014; Lee and Lautenbach, 2016), and are confounded by methodological differences as well as unobserved study site heterogeneity. Another advantage to our data is that we have enough observations to split the data according to the factor of interest (private versus public land ownership), which is a unique approach among studies of social and governance drivers of ecosystem service relationships (Dade et al., 2019; Dittrich et al., 2017; Manners and Varela-Ortega, 2018; Raudsepp-Hearne et al., 2010).

Our study also contributes to the literature estimating the degree of substitutability between natural capital and other forms of capital (for a review see Cohen et al., 2019). Much of this existing literature focuses on static definitions of substitution and complementarity and does not distinguish between flows of services and stocks of natural capital.[1] Two notable exceptions are Yun et al. (2017) and Rouhi Rad et al. (2021) which take a dynamic approach in order to distinguish between service flows and natural capital stocks. These dynamic studies show that complementarities between ecosystem service flows may imply the underlying natural capital are substitutes. However, the focus of these studies is on assessing the potential for sustainability (Arrow et al., 2004; Hartwick, 1977) and not as a tool for identifying ecosystem externalities.

2. Theoretical model

A fixed quantity of land, X, is used in N joint production processes involving marketed commodities (i.e., provisioning ecosystem services) and NMES. Each joint production process is indexed by $i = 1, \ldots, N$ and the quantity of market commodities and NMES associated with process i is given by y_i and z_i, respectively. The joint production function $f_i(x_i, y_i, z_i)$ describes how the marketed commodity y_i and NMES z_i are jointly produced using a single land input x_i that contributes to the output of both y_i and $z_i : \frac{\partial f_i}{\partial y_i}\frac{\partial y_i}{\partial x_i} \geq 0$ and $\frac{\partial f_i}{\partial z_i}\frac{\partial z_i}{\partial x_i} \geq 0$.[2] For example, forest habitat (z_1) and timber (y_1) are both produced from land devoted to timber production (x_1). Grazing (z_2) and livestock (y_2) are both produced from land devoted to livestock production (x_2). NMES are enhanced by the conservation of other NMES: $z_i = g_i(\mathbf{z})$ where $\mathbf{z}$ is a vector of the levels of all NMES except i and $\partial g_i / \partial z_{-i} > 0$. For example, forest habitat and grazing produced from land devoted to timber and livestock production may also improve the quality and quantity of municipal drinking water supplies by limiting siltation and controlling instream flows.

The value of commodity i is p_i. There is a shadow value for each NMES derived from demand for the commodity. For simplicity we assume that society values the NMES only for their contribution to production of the commodity (i.e., the societal value of the NMES is its shadow value). Allowing for additional values for NMES such as existence, bequest, or option values adds notational complexity but doesn't change our fundamental results.

2.1. Optimal supply of NMES under joint production

A social planner will choose to allocate land to maximize the value of all commodities

$$\max_{x_1, x_2, \ldots x_N} \sum_{i=1}^{N} p_i y_i \tag{1}$$

subject to production technologies $f_i(x_i, y_i, z_i)$ and $z_i = g_i(\mathbf{z})$ and a limited amount of land $\sum_{i=1}^{N} x_i \leq X$. The Lagrangian for the social planner's problem is

$$\mathscr{L} = \sum_{i=1}^{N} p_i f_i(x_i, y_i, g_i(\mathbf{z})) + \lambda\left(X - \sum_{i=1}^{N} x_i\right) \tag{2}$$

The social planner's optimal land allocation, x_i^*, ensures that the value marginal product of land used to produce the commodity is equal to the shadow value of land which serves as a constant marginal cost of

[1] For an example of a study that focuses on substitution and complementarity of ecosystem services see (Reynaud, 2003). For an example of a study that focuses on substitution or complementarity of natural capital instead of ecosystem service flows see Markandya and Pedroso-Galinato (2007). These studies also suffer from the endogeneity bias prevalent in the ecosystem services literature (Cohen et al., 2019).

[2] Joint production arises from either technical interdependencies or the presence of non-allocatable inputs that cannot be managed between products.

land:

$$p_i \frac{\partial f_i}{\partial y_i}\frac{\partial y_i}{\partial x_i} + p_i \frac{\partial f_i}{\partial y_i}\frac{\partial y_i}{\partial z_i}\frac{\partial z_i}{\partial x_i} + \sum_{-i} p_{-i} \frac{\partial f_{-i}}{\partial y_{-i}}\frac{\partial y_{-i}}{\partial g_{-i}}\frac{\partial g_{-i}}{\partial z_i}\frac{\partial z_i}{\partial x_i} = \lambda \text{ for all } i \tag{3}$$

The left-hand side of Eq. (3) reflects the joint production associated with NMES. The first term is the direct marginal effect of land on commodity production. The second term is the indirect marginal effect of land on commodity production by way of the NMES. Due to this technical relationship, the social planner optimally produces NMES ($x_i^* > 0$) based on the shadow value for each NMES derived from demand for the commodity. The third term arises because NMES i enhances the production of NMES $-i \neq i$ which also enhances the production of commodity $-i \neq i$.

The joint production technology implies that the PPF for commodity production may not be concave (Brown et al., 2011). To illustrate, consider the two-good, two-stock case. In the absence of joint production, $\frac{dy_1}{dy_2} < 0$ and the PPF is downward sloping. Joint-production makes the slope of the PPF ambiguous. However, the optimal commodity mix will equate the ratio of commodity prices to the marginal rate of transformation between commodities, $\frac{p_1}{p_2} = -\frac{dy_1}{dy_2}$, which ensures that the socially optimal land allocation will find commodities competing for land (occurs on a downward sloping portion of the PPF) since the values for both commodities are positive.

A downward sloping PPF for the production of NMES also cannot be guaranteed (see Fig. 1A and B). The NMES becomes an inferior input for commodity production once a specific level $\bar{z}_i$ is reached. For $z_i < \bar{z}_i$, complementary products can be produced in increasing quantities: $\frac{dy_i}{dz_i} > 0$. For $z_i > \bar{z}_i$, NMES becomes an inferior input $\left(\frac{\partial f_i}{\partial y_i}\frac{\partial y_i}{\partial z_i} < 0\right)$ and the commodity and NMES are competing products: $\frac{dy_i}{dz_i} < 0$.

Since NMES is enhanced by the production of other NMES, shifting land from production of one NMES to the other can increase the production of the NMES and the associated commodity. Because NMES is subject to a similar joint production technology as commodity production, the shape of the PPF between two NMES stocks has a similar shape with upward sloped portions characterizing complementary production $\left(\frac{dz_1}{dz_2} > 0\right)$ and downward sloped portions characterizing competing production technologies $\left(\frac{dz_1}{dz_2} < 0\right)$ (Fig. 1C). Shifting land from production of NMES i to the production of NMES $-i \neq i$ increases NMES i until $z_{-i} \geq \bar{\bar{z}}_{-i}$ at which point the limited availability of land implies the two NMES are competing.

However, there is no guarantee that the social optimum falls on the downward-sloping portion of the ecological-economic PPF. Every point along the PPFs in Fig. 1A, B, and C are potentially optimal. If the NMES does not enhance the production of the other NMES ($\partial g_{-i}/\partial z_i = 0$), the social planner will never produce more than $\bar{z}_i$ since this would lower commodity production in exchange from an NMES that has no market. Thus, the social optimum may be defined by complementarities between commodities and NMES. However, if NMES enhance one another ($\partial g_{-i}/\partial z_i > 0$), the socially optimal amount of NMES i may exceed $\bar{z}_i$ provided the value marginal product of the other commodities is sufficiently high: $p_i\left[\frac{\partial f_i}{\partial y_i}\frac{\partial y_i}{\partial x_i} + \frac{\partial f_i}{\partial y_i}\frac{\partial y_i}{\partial z_i}\frac{\partial z_i}{\partial x_i}\right] < \sum_{-i} p_{-i}\frac{\partial f_{-i}}{\partial y_{-i}}\frac{\partial y_{-i}}{\partial z_{-i}}\frac{\partial z_{-i}}{\partial z_i}\frac{\partial z_i}{\partial x_i}$. In this case, socially optimal land allocations may be characterized by complementarities or tradeoffs between commodities and NMES and between NMES.

2.2. NMES relationships and ecosystem externalities

The interdependencies between NMES are often ignored in practice (Crocker and Tschirhart, 1992). In a seminal paper, Crocker and Tschirhart use a rodent infestation in Kern County, California to motivate the notion of an ecosystem externality. Residents of Kern County engaged in predator removal to reduce livestock depredation. However, the predator removal led to an increase in mice that decimated crops for many years. More generally, a timber producer may not account for the effect forests have on the water supply of a downstream city. A rancher may not account for the effect forage has on crop production of a neighboring farm.

In a competitive market where each commodity is produced by a separate firm, land allocation decisions will not account for the interdependencies between NMES and will allocate land as if $\partial g_{-i}/\partial z_i = 0$.[3] Consider a firm that produces y_i subject to the same production technologies as the social planer. The firm will choose the amount of land allocated to produce commodity i to maximize profits

$$\max_{x_i} p_i y_i - c x_i \tag{4}$$

subject to $f_i(x_i, y_i, z_i)$ where c is the market price of land. Firms continue to provide NMES based on the shadow value created by the joint production relationship between commodity i and NMES i. The firm assumes other NMES will be supplied by other firms and takes $\mathbf{z}$ as given.

The firm's optimal land allocation, x_i', ensures that the value marginal product of land devoted to commodity production is equal to the shadow value of land which serves as a constant marginal cost of land:

$$p_i \frac{\partial f_i}{\partial y_i}\frac{\partial y_i}{\partial x_i} + p_i \frac{\partial f_i}{\partial y_i}\frac{\partial y_i}{\partial z_i}\frac{\partial z_i}{\partial x_i} = c \tag{5}$$

The competitive market equilibrium will not supply the socially optimal amount of NMES. By comparing Eq. (5) to Eq. (3) we find that the firm that specializes in the production of a single commodity will not account for the impact of the NMES it produces on the production of other commodities. As is well-known, a positive externality will cause the competitive market equilibrium to underprovide NMES relative to the social optimum $\left(x_i' < x_i^*\right)$ while a negative externality will lead to overprovision of the NMES $\left(x_i' > x_i^*\right)$.

However, the ecosystem externality has implications for the relationships between ecosystem services that are often overlooked. A competitive market equilibrium must lie on the dashed portion of the PPF in Fig. 2A and B (i.e., a complementary relationship between commodities and NMES ($dy_i/dz_i > 0$) is now guaranteed). The firm will not produce NMES in excess of $\bar{z}_i$ because it lowers production of commodity i in exchange for an NMES with no market and a commodity the firm does not own. This guarantees a tradeoff between NMES under a competitive market ($dz_1/dz_2 < 0$). Fig. 2C shows the relationship between NMES that emerges from a competitive market that does not account for the externality between NMES. NMES are viewed as a secondary input when firms ignore technical interdependencies between NMES and, as a result, any increase in one NMES must come at the cost of another NMES due to the land constraint.

The NMES remain competing products when internalizing a negative externality (i.e., the competitive equilibrium and social optimum both yield $dz_1/dz_2 < 0$). However, internalizing a positive externality makes possible complementary relationships between NMES ($dz_1/dz_2 > 0$). In the following section, we show how this difference in the private and social PPF can be used to test whether positive ecosystem externalities are incorporated in production decision.

3. Empirical analysis

We use the results of our theoretical model to empirically test whether private markets in the United States are successfully internalizing externalities associated with drinking water; an externality that has been the focus of USDA's "Forests to Faucets" initiative. Table 1

[3] General results still hold provided a single firm doesn't produce all commodities.

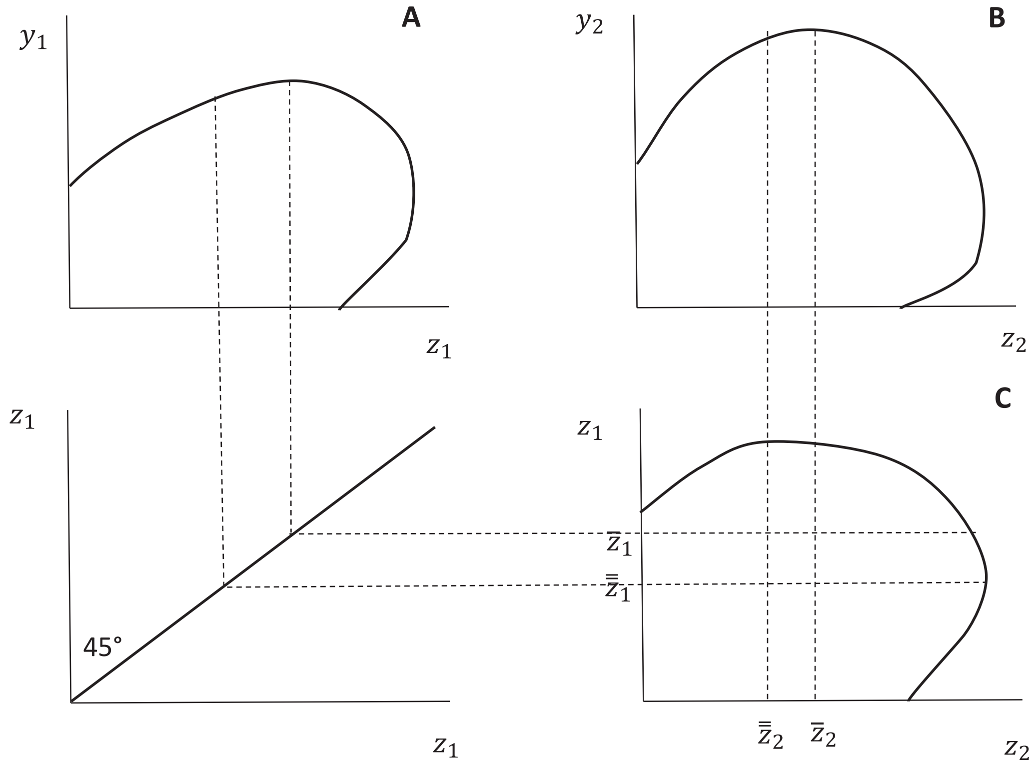

Fig. 1. Production possibility frontiers under a social optimum where interdependencies between ecosystem services are internalized.

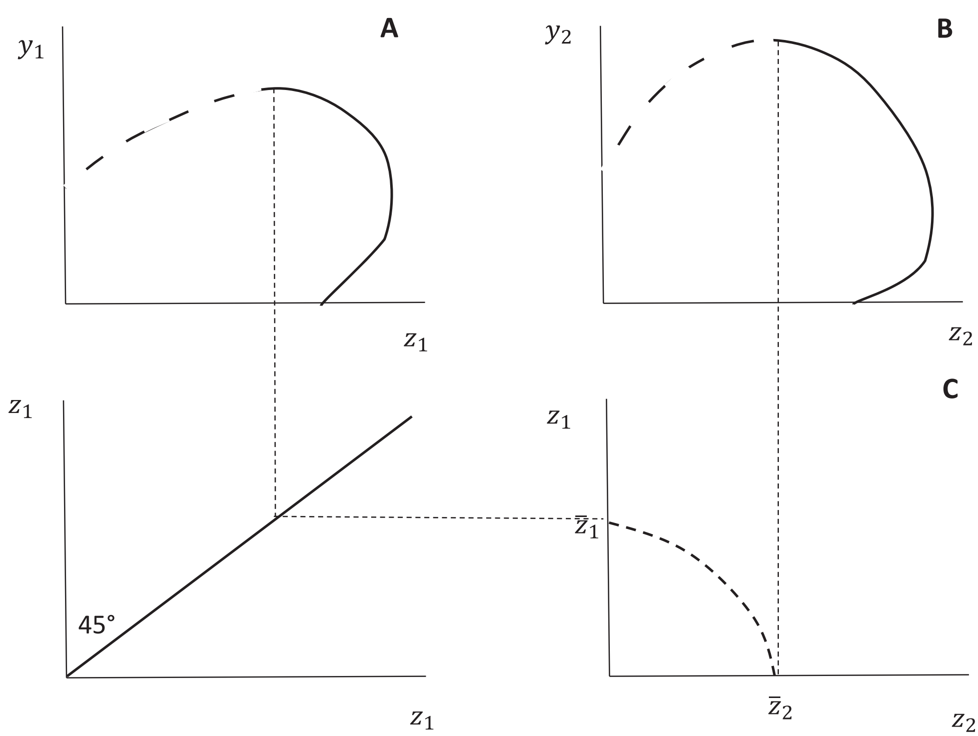

Fig. 2. Production possibility frontiers under a competitive market where interdependencies between ecosystem services are not accounted for in production decisions.

indicates the commodities included in our study y_i, the jointly produced NMES z_i, and the hypothesized sign of the externality associated with each NMES.

In areas that have predominately privately-owned land, we estimate pairwise relationships between three NMES that are jointly produced with a commodity (see Table 1): instream flows and municipal drinking water, W; forest habitat and timber, F; and grazing and livestock production, G. Our theoretical model predicts that the pairwise relationships between NMES on privately owned lands should be negative ($dz_i/dz_{-i} < 0$). Positive relationships between any two NMES

Table 1
Jointly produced NMES-commodity pairs considered in our analysis.

Commodity (y)	Jointly produced NMES (z)	Hypothesized externality ($\partial g/\partial z$)
Municipal drinking water	Instream flows (W)	Positive
Timber	Forest habitat (F)	Positive
Livestock	Grazing (G)	Positive

($dz_i/dz_{-i} > 0$) would suggest that private markets are at least partially internalizing the positive externality between these NMES.[4] A negative relationship, however, does not necessarily signal that markets are failing to internalize an externality. Negative relationships are also possible when the positive externality is internalized.

To provide a benchmark in these cases, we also estimate the NMES relationships in areas that have predominately publicly-owned land. Since public land managers are charged with managing for multiple uses, observed relationships on public lands should be more similar to the social planner case considered in the theoretical model. Evidence of NMES externalities only occur if we find positive pairwise relationships on public lands that flip to negative relationships on private lands.

3.1. Data

NMES levels are drawn primarily from the Risk of Watershed Impairment (RWI) workbook published by the USDA Forest Service (Brown and Froemke, 2012). The RWI workbook contains information on physical characteristics and stressors for the 3672 10-digit hydrologic unit code (HUC10) watersheds containing National Forest System (NFS) lands in the contiguous United States.[5] From this database, we construct measures of three NMES in each HUC10 watershed:

1. Instream flows (W): We measure the instream flows as a simple inverse function of evapotranspiration ($W = 1/ET$), which is the main avenue through which land use impacts instream flows (Burt, 1979; Fang et al., 2022).[6] ET is a function of forest cover, derived using Zhang curves (Zhang et al., 2001) with the following functional form:

$$\frac{ET}{P} = f\frac{1 + w\frac{E_Z}{P}}{1 + w\frac{E_Z}{P} + \left(\frac{E_Z}{P}\right)^{-1}} + (1-f)\frac{1 + w\frac{E_Z}{P}}{1 + w\frac{E_Z}{P} + \left(\frac{E_Z}{P}\right)^{-1}} \tag{6}$$

where P is precipitation, E_z and w are, respectively, evaporation and a plant-available water coefficient specific to the type of vegetation, f is the percent of area in forests, and (1-f) is the percent of area in grasses. Zhang et al. estimate $E_z = 1410$ and $w = 2.0$ for trees, and $E_z = 1100$ and $w = 0.5$ for grasses. Utilizing those parameter values and solving for ET yields

$$ET = 0.996369\,P + 0.003038\,P \times f. \tag{7}$$

2. Forest habitat (F): The amount of forest habitat provided by a watershed is measured as the percentage of the watershed that is forested and conducive to timber harvesting (i.e., <45-degree slope). We take total forest area in each watershed and subtract area that is >45-degree slope, then convert to a density measure by dividing by the area of each watershed. Forest area is based on 300-m resolution data from the 2001 National Land Cover Database (NLCD).
3. Grazing (G): The amount of grazing provided by a watershed is measured by the livestock grazing density (animal units per km^2). Animal units (AUs) per km^2 in year 2007 are based on USDA Census of Agriculture county data, released by the National Agricultural Statistics Service. Included livestock categories and their respective factors for converting them into AUMs are: cattle (1), horses (1.25); sheep (0.2); goats (0.2); llamas (0.2), alpacas (0.2), bison (1.25), deer (0.2), elk (0.5) and mules (0.8).

NMES levels within a watershed depend on technical interrelationships as well as a host of exogenous factors. We attempt to control for as many of these factors as possible. Housing density and developed landcover account for some differences in land use that affect NMES, while land value and population growth rate are proxies for development pressure. Annual precipitation directly impacts the biophysical productivity of an ecosystem, and we include a measure of wildfire risk, since managing for NMES is less likely when there is a high probability of total loss. Finally, the complexity of ecosystems suggests that the pairwise relationships we aim to identify may differ based on the presence or absence of other NMES. For this reason, we include a dummy variable indicating whether all ecosystem services are present in the watershed.

To focus on potential positive externalities associated with drinking water, we limit our analysis to the 738 watersheds in the RWI Workbook that draw drinking water from surface water or groundwater under the influence of surface water (Fig. 3).[7] We classify the watersheds as private or public according to the dominant (>50%) land ownership status and compare the summary statistics (Table 2). The 468 predominantly private watersheds and 270 public watersheds contain, on average, 78% and 25% privately-owned land, respectively.

Significant differences exist between private and public watersheds across the variables. Generally, private lands are used more intensively, are more developed, are more likely to produce all three NMES simultaneously, and are valued more highly. Public lands, on the other hand, are experiencing more rapid population growth but have more land subject to potential loss from a devastating wildfire. The results are predictable, given that public lands are likely chosen for retirement because of their relatively low productivity and high development costs (e.g., due to rougher terrain). Since our goal is to identify differences in NMES relationships arising from the presence or absence of private markets, these systematic differences between private and public lands could confound the results.

We preprocess the data in two steps to improve the comparison of private and public watersheds. First, we drop 69 watersheds (<10% of the total) with extremely high withdrawals of surface water (>15,000

[4] The inability to find a statistically significant relationship between NMES may indicate NMES levels are near the flat portion of the PPF where the technology switches between complementary and competing (i.e., when z_i is in the neighborhood of $\bar{z}_i$). Note that even if the production technologies for two NMES are independent in that $\partial g_i/\partial z_{-i} = 0$, we should still find a negative relationship in the PPF due to the limited amount of land. Thus, finding no statistically significant relationship indicates that z_i is in the neighborhood of $\bar{z}_i$ or land constraints are not binding.

[5] Data is from https://www.fs.fed.us/rm/value/watershedcondition/index.html.

[6] We also calculate an alternative measure, withdrawals of drinking water, to be used in robustness checks (see Appendix). The EPA collects spatially explicit data on the population served by individual public water system intakes, which are of three types: ground water, surface water, and ground water under the influence of surface water. We sum the population served by intakes drawing surface water and groundwater under the influence of surface water within each watershed, assuming that underground aquifers are unaffected by other NMES. We multiply the population served totals by per capita water usage to obtain an estimate of watershed-level withdrawals. Water usage statistics are reported at the county level, so we utilize the mean when a watershed intersects multiple counties. Finally, we divide by the area of each watershed, making the alternative water measure an estimate of gallons withdrawn per day per km^2.

[7] EPA intake data is from https://www.epa.gov/ground-water-and-drinking-water/safe-drinking-water-information-system-sdwis-federal-reporting.

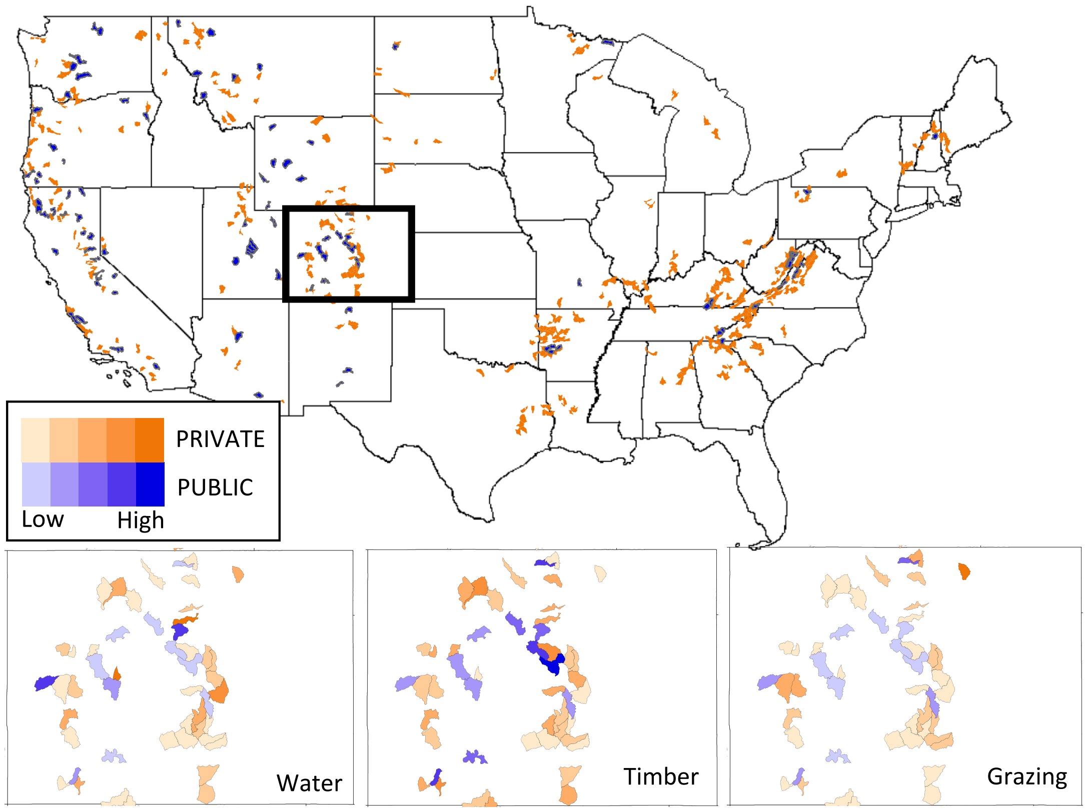

Fig. 3. Watersheds in the final dataset by predominant ownership type with ecosystem service levels detailed for the state of Colorado.

Table 2
Summary statistics.

	Raw data			Preprocessed data		
	Private	Public	t-stat	Private	Public	t-stat
Sample size	468	270		417	417	
% privately owned	77.74	24.50		77.74	33.15	
Jointly-produced NMES						
Instream flows (*W*)	1.25	1.48	3.98	1.17	1.70	9.28
Forest habitat (*F*)	43.42	36.16	3.98	46.65	27.63	10.76
Grazing (*G*)	9.38	4.14	8.43	9.84	5.02	9.28
Controls						
Mining (active mines/km^2, *M*)	1.78	0.54	5.89	1.49	0.57	6.76
Housing density (units/km^2)	34.16	19.22	2.55	17.98	15.91	1.21
% developed landcover	8.14	2.46	8.74	6.13	6.09	0.10
Land value ($/ha)	29,883	13,817	2.20	14,468	14,044	0.37
% change in population	1.54	2.16	5.55	1.48	1.68	2.34
Annual precipitation (mm)	1034	944	2.26	1072	909	4.13
% high wildfire risk	15.38	20.70	3.42	16.18	17.68	0.97
All four services present	0.37	0.13	7.40	0.39	0.20	6.15

gal per day per km^2). The mean housing density for these observations is 134.6 units per km^2 and the mean land value is $121,323 per hectare, both several orders of magnitude above the respective overall means. At the same time, these watersheds tend to be more arid than our overall sample, with average annual precipitation of 746.6 mm. We drop these outliers because there are no comparable public watersheds.

Second, we perform matching to reduce model dependence (Ho et al., 2007). We use the control variables to calculate a propensity score using the scoring function that optimizes covariate balance between private and public watersheds using the R package CBPS (Fong et al., 2017), diminishing potential bias introduced by misspecifying the propensity score model (Imai and Ratkovic, 2014). Then we perform 1:1 nearest-neighbor matching with replacement, effectively dropping some public watersheds and weighting those with observed characteristics most similar to private watersheds.

Dropping outliers and matching on propensity scores greatly improves the comparability of the two types of watersheds (Table 2). With three exceptions, statistically significant differences across the control variables disappear. In terms of NMES levels, public watersheds remain less productive than their private counterparts in terms of forest habitat and grazing. Instream flows, however, are now higher in public watersheds relative to private. While it would be unrealistic to assume that the controls capture all differences between public and private lands, the jump in instream flows in public watersheds suggests the existence of a positive externality that goes uncaptured on privately-owned watersheds. However, a comparison of means is insufficient to identify definitive relationships within a complex ecosystem. We turn next to careful identification of pairwise NMES relationships.

3.2. Empirical model

Given the potential for technical interdependencies between NMES, we specify a system of equations with each NMES $i = 1, 2, 3$ as a dependent variable and all the other NMES entering as independent variables:

$$lnW_j = \beta_{12}lnF_j + \beta_{13}lnG_j + \theta_{1M}lnM_j + \mathbf{x}_j'\boldsymbol{\alpha}_1 + \mathbf{w}_{1j}'\boldsymbol{\gamma}_1 + \delta_k + \varepsilon_1$$

$$lnF_j = \beta_{21}lnW_j + \beta_{23}lnG_j + \theta_{2M}lnM_j + \mathbf{x}_j'\boldsymbol{\alpha}_2 + \mathbf{w}_{2j}'\boldsymbol{\gamma}_2 + \delta_k + \varepsilon_2$$

$$lnG_j = \beta_{31}lnW_j + \beta_{32}lnF_j + \theta_{3M}lnM_j + \mathbf{x}_j'\boldsymbol{\alpha}_3 + \mathbf{w}_{3j}'\boldsymbol{\gamma}_3 + \delta_k + \varepsilon_3 \quad (8)$$

where W_j is the amount of instream flows provided for municipal use in watershed j, F_j is the amount of forest habitat jointly produced with timber in watershed j, G_j is the amount of grazing provided for livestock production in watershed j, and ε_i is an error term. We include three types of control variables. Since ecosystem services are jointly determined, the first is a marketed ecosystem service (mineral resource extraction, M_j) that coexists with and likely influences the NMES but does not have an obvious jointly produced NMES.[8] The second, $\boldsymbol{x_j}$, is a vector of other controls common to all three equations and contains the control variables. The third, $\boldsymbol{w_{ij}}$, is unique to each equation, containing variables that impact the level of a single ecosystem service, the dependent variable in equation i. To control for unobserved regional variation in landcover, ownership, management, industrial activity, and climate, we include regional fixed effects δ_k, where k is the HUC 2 region containing watershed j.

Since ecosystem services are jointly determined, they are endogenous when they enter the system on the right-hand side. Performing OLS on each equation separately, therefore, will produce inconsistent estimates for our coefficients of interest $\beta_{i,-i}$. We address this endogeneity concern in three ways. First, we estimate a similar equation for the marketed ecosystem service:

$$lnM_j = \theta_{M1}lnW_j + \theta_{M2}lnF_j + \theta_{M3}lnG_j + \boldsymbol{x}_j^{'}\boldsymbol{\alpha_M} + \boldsymbol{w}_{Mj}^{'}\boldsymbol{\gamma_M} + \delta_k + \varepsilon_M \tag{9}$$

Second, we utilize $\boldsymbol{w_{ij}}$ as instruments for each NMES i. We carefully select a pair of instruments for each NMES: density of water supply dams and the presence of a sole-source aquifer (W), presence of logging employment and timber tract operations (F), and presence of ranching employment and goat or sheep farm employment (G).[9] The success of this approach requires choosing elements of $\boldsymbol{w_{ij}}$ that are plausibly exogenous to the -i NMES (i.e., $\boldsymbol{w_{ij}}$ only impacts the other NMES through its effect on i). For example, the instrumental variable "presence of logging employment" only impacts instream flows through its effect on forest habitat. We believe this exclusion restriction likely holds for each of our instrumental variables. The other requirement for a valid instrumental variable is that the instrument is correlated with the variable it is instrumenting. For example, the presence of logging employment is correlated with the percentage of a watershed that is forested (i.e., forest habitat). The OLS results provided in the appendix indicate that are least one instrumental variable is correlated with each dependent variable.

Third, to allow correlation between the errors, we create a system of equations by adding Eq. (9) to the system of equations in (8) and utilize three-stage least squares (3SLS) to estimate the resulting system of equations jointly (Zellner and Theil, 1962). Performing two-stage least squares on each equation individually would assume independence of the error terms, ε_i. This is an unreasonable assumption for ecological systems where stochastic weather shocks may influence multiple ecosystem service flows simultaneously, or there is a common driver such as land use change influencing multiple ecosystem services.

We interpret the coefficients as indicative of the production relationship between a pair of NMES. For example, a positive and statistically significant estimate for β_{12} suggests that instream flows (W) and forest habitat (F) are complements-in-production. The directional relationship should hold regardless of which service is the dependent variable (i.e., $\beta_{12} > 0 \Rightarrow \beta_{21} > 0$). This system of equations approach is a reduced form representation of the complex ecological and human-built system in which NMES influence and are influenced by the provision of other NMES. For example, instream flows encourage forest growth that contributes to the provision of timber. However, timber harvesting activities will also alter forest habitat and the amount of water that reaches municipal water intakes.

4. Results and discussion

Results from 3SLS (Table 3) are consistent with predictions from our theoretical model. All coefficients of interest are statistically significant, and the signs are consistent regardless of the direction by which the relationship is estimated (i.e., the pairwise relationships we identify are the same regardless of whether NMES i enters as a dependent or independent variable). The coefficients can be interpreted as elasticities, but we focus only on the signs to identify directional relationships.

In predominantly private watersheds, uniformly negative coefficients indicate that tradeoffs between NMES dominate. Each NMES is only increased at the expense of others, which is consistent with the theoretical result precluding the possibility of complementary relationships in competitive markets. The negative relationships may result from biophysical relationships or an externality.

In contrast, two of the three pairwise relationships are positive in predominantly public watersheds, providing evidence that complementary relationships between certain NMES are possible. When we compare these relationships to those found in private watersheds, we find that two pairwise relationships, (instream flow, forest habitat) and (grazing, forest habitat), flip from negative relationships on private land to positive relationships on public lands. To the extent that systematic differences between the quality of public and private lands are captured by our controls, our theoretical model suggests that the negative relationships between these services on private lands reflect a failure to internalize a positive ecosystem externality. Private market participants are failing to account for the complementarities between forest habitat, instream flows, and grazing because they are not incentivized to do so, resulting in inefficient provision of these services.

We perform several robustness checks. First, we estimate an alternative specification of Eq. (8) utilizing fewer and different controls. Specifically, the land use, development pressure, and fire hazard controls are replaced by two coarse measures: (1) percentage of the watershed with slope >45 degrees, and (2) percentage of the watershed that is privately owned. Second, we return to the original controls but utilize surface water withdrawals as an alternative measure of instream flows, W. Finally, we re-estimate the original specification using the raw (unprocessed) data, which is unmatched and includes outliers. Full results are in the Appendix. In terms of each directional relationship, the robustness checks are consistent with our main results, though slight differences are apparent in the level of statistical significance. Overall, the robustness checks strengthen our findings that private markets erode the possibility of complementary relationships produced from the multi-use management on public lands.

Finally, since OLS and correlations are commonly employed in the literature exploring ecosystem service relationships, we compare our main 3SLS results to results obtained using these techniques with the preprocessed data. Results are summarized in Table 4 (see Appendix for full OLS results). 3SLS acts as a benchmark to which we can compare correlations and OLS results, because 3SLS addresses shortcomings of the other two methods. The validity of correlation coefficients to

[8] We measure mineral resource extraction by the number of active mines per 1000 km^2. We sum the active mines within a watershed and divide by the area. Active mine data is from https://mrdata.usgs.gov/mineplant/geo-inventory.php.

[9] We use BLS NAICS codes to construct dummy variables indicating the presence of the following employment categories within a watershed: Logging (1133); Timber tract operations (1131); Mining (212); Support activities for mining (213); Cattle ranching and farming (1121); Sheep and goat farming (1124). Reverse causation would invalidate employment as an instrument if the NMES causes employment. Concerns about reverse causation are addressed by the joint production relationship which implies employment is a non-allocatable input that cannot be managed between commodity and NMES production. We utilize a similar dummy variable for employment presence as an instrumental variable for mineral resource extraction in Eq. (9) (NAICS codes 212 and 213). We tried several other instrumental variables with limited success including housing density, range and agricultural land density, a dummy variable for cattle ranching activity, and forest biomass density.

Table 3
3SLS results.

	Private			Public		
	ln(*W*)	ln(*F*)	ln(*G*)	ln(*W*)	ln(*F*)	ln(*G*)
ln(*W*)		−19.475*** (6.271)	−15.791*** (5.123)		15.382*** (2.366)	−13.546*** (1.683)
ln(*F*)	−0.028*** (0.008)		−0.581*** (0.227)	0.058*** (0.007)		0.725*** (0.069)
ln(*G*)	−0.043*** (0.011)	−1.380*** (0.307)		−0.071*** (0.010)	1.180*** (0.135)	

Note: Regressions include all variables in Table 2, plus instruments and HUC 2 fixed effects. *,**,*** indicate significance at the 90%, 95% and 99% levels, respectively.

Table 4
Summary of nonmarket ecosystem service relationships.

	Private		Public	
	W	*F*	*W*	*F*
F	− − −		− + +	
G	− − −	**0 0** −	− − −	+ **0** +

Note: First, second and third entries represent results from correlation coefficients, OLS, and 3SLS, respectively. 0 indicates failure to identify a statistically significant relationship.

correctly identify pairwise relationships rests on assumptions of a linear relationship and the absence of confounders. OLS allows us to control for confounding variables and introduce nonlinearities, but endogeneity persists because ecosystem services are jointly determined (i.e., they appear as both independent and dependent variables). Given these shortcomings, deviation from 3SLS results are expected. Correlation coefficients fail to identify one significant relationship (*F-G* on private watersheds), and they misidentify directional relationships in one case (*W-F* on public watersheds). OLS does not misidentify the direction of any relationships, but it disagrees with 3SLS regarding statistical significance in one third of cases. The three methods produce consistent results for only half of the possible relationships, indicating a severe degree of bias associated with OLS and correlation coefficients. In particular, studies based on correlations and OLS may be under counting the number of significant relationships and misrepresenting the direction of others.

5. Conclusions

Using a theoretical model of jointly produced ecosystem service provision and data from over 700 watersheds, we find evidence of an ecosystem externality involving two NMES relationships: 1) instream flows and forest habitat and 2) grazing and forest habitat. We identify complementary relationships between these services on publicly-owned lands, while the same services become competing on privately-owned lands. This finding suggests that land markets are overlooking win-win opportunities to take advantage of synergistic biophysical relationships between nonmarket ecosystem services. For instance, private forest landowners do not appear to recognize that their forests increase drinking water for cities and towns downstream. This finding is consistent with other research that suggests watersheds on public lands are healthier not because of management actions specifically targeting water quality, but rather, public lands improve water quality by keeping other activities out (Abildtrup et al., 2013). Regulations or transfers could potentially be designed to internalize the ecosystem externality on private lands.

Broader-scale studies (i.e., non-case studies) tend to focus on trade-offs and synergies between broad categories of ecosystem services (provisioning vs regulating, for example). Evidence of certain externalities can be inferred from other literatures. For example, the small group of studies estimating the value of forests as avoided cost of drinking water treatment demonstrate a positive externality between forest biomass and water quality (Warziniack et al., 2017; Abildtrup et al., 2013). Using a different approach, we find related evidence for a positive externality involving the presence of forest and water quantity. Managed grasslands produce higher quality forage compared to abandoned grasslands that are encroached by shrubs and forest suggesting a possible negative relationship between grazing and forests (Johansen et al., 2019). However, our evidence suggests that public management of grasslands can reverse this relationship and exploit complementarities between grazing and forest habitat.

Our results also provide an additional motivation for addressing endogeneity arising from joint-production technologies involving complex ecosystem interactions. As Cohen et al. (2019) discuss, the growing number of estimates of relationships between ecosystem services should be interpreted with caution due to endogeneity bias. Our paper shows how correcting for endogeneity bias can also be used to uncover warning signs of ecosystem externalities at large spatial scales. While our method provides one possible way to address endogeneity concerns, it also highlights the importance of ensuring the conditions for a valid instrumental variable are met. The growing number of landscape-level datasets documenting land use change, land use intensity, economic returns, and resource use will ease the well-known challenges of identifying valid instrumental variables for ecosystem services. However, additional empirical work with these datasets will be needed to determine valid instruments for different sets of jointly produced ecosystem services that can be used in both predominately private and predominately public watersheds. We leave these empirical analyses for future work.

CRediT authorship contribution statement

Ben Blachly: Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Software, Validation, Visualization, Writing – original draft, Writing – review & editing. **Charles Sims:** Conceptualization, Data curation, Formal analysis, Funding acquisition, Investigation, Methodology, Project administration, Resources, Supervision, Validation, Writing – original draft, Writing – review & editing. **Travis Warziniack:** Conceptualization, Data curation, Funding acquisition, Investigation, Methodology, Writing – original draft, Writing – review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

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are those of the author(s) and should not be construed to represent any official USDA or U.S. Government determination or policy.

Appendix A. Appendix

Table A1
Ordinary least squares (OLS) results.

	Private			Public		
	ln *W*	ln *F*	ln *G*	ln *W*	ln *F*	ln *G*
ln *W*		−6.781***	−4.607***		0.897	−2.393***
		(1.161)	(1.218)		(1.364)	(0.822)
ln *F*	−0.012***		−0.047	0.005**		−0.011
	(0.002)		(0.052)	(0.002)		(0.031)
ln *G*	−0.007***	−0.057		−0.006**	0.073	
	(0.002)	(0.050)		(0.003)	(0.073)	
ln *M*	0.001	−0.216***	−0.197**	−0.012	−1.459***	−0.174
	(0.003)	(0.083)	−0.085	(0.008)	(0.191)	(0.132)
Housing density	−0.010***	0.056	−0.247***	−0.010***	0.162**	−0.259***
	−0.003	(0.062)	(0.062)	(0.003)	(0.079)	(0.050)
% developed	−0.003	0.164	0.234	−0.190***	3.791***	0.473
	(0.014)	(0.347)	(0.357)	(0.029)	(0.718)	(0.508)
Land value	0.001	0.131**	−0.128**	−0.010***	−0.292***	−0.478***
	(0.003)	(0.060)	(0.062)	(0.004)	(0.094)	(0.055)
Population change	−0.002	1.260***	0.231	−0.169***	3.469***	0.284
	(0.015)	(0.366)	(0.379)	(0.023)	(0.577)	(0.402)
Precipitation	−0.521***	−2.444***	−2.391***	−0.496***	1.616***	−1.465***
	(0.004)	(0.630)	(0.647)	(0.005)	(0.684)	(0.416)
Wildfire risk	−0.023***	0.470**	−0.683***	−0.063***	0.750***	0.235
	(0.009)	(0.215)	(0.217)	(0.009)	(0.256)	(0.166)
All services	−0.004	0.408***	0.350***	0.004	2.406***	0.327**
	(0.005)	(0.127)	(0.130)	(0.010)	(0.213)	(0.159)
Pop chg * % dev	0.009	−0.408**	0.160	0.136***	−2.682***	0.150
	(0.008)	(0.198)	(0.204)	(0.018)	(0.437)	(0.312)
Dam density	−0.012***			0.008		
	(0.004)			(0.006)		
Withdrawals	−0.004***			0.000		
	(0.001)			(0.002)		
Dam*Withdrawals	0.002***			−0.001		
	(0.001)			(0.001)		
Logging employment		0.262***			0.875***	
		(0.075)			(0.120)	
Timber operations		−0.032			−0.113	
		(0.081)			(0.125)	
Ranch employment			0.108			0.636***
			(0.078)			(0.077)
Goat/sheep farming			0.105			−0.055
			(0.109)			(0.078)

Table A2
Robustness checks for 3SLS results.

Alternative Controls						
	Private (*N* = 417)			Public (N = 417)		
	ln(*W*)	ln(*F*)	ln(*G*)	ln(*W*)	ln(*F*)	ln(*G*)
ln(*W*)		−13.930***	−11.136***		11.792***	−9.651***
		(5.495)	(4.642)		(3.160)	(1.412)
ln(*F*)	−0.046***		−0.805***	0.029***		0.347***
	(0.011)		(0.158)	(0.006)		(0.053)
ln(*G*)	−0.051***	−1.254***		−0.084***	1.927***	
	(0.013)	(0.218)		(0.011)	(0.180)	

Alternative Water Availability Measure						
	Private (N = 417)			Public (N = 417)		
	ln(*W*)	ln(*F*)	ln(*G*)	ln(*W*)	ln(*F*)	ln(*G*)
ln(*W*)		−0.258*	−0.192**		0.213***	−0.075**
		(0.136)	(0.099)		(0.045)	(0.031)
ln(*F*)	−1.533***		−0.478**	0.469**		0.231***
	(0.494)		(0.223)	(0.218)		(0.086)
ln(*G*)	−3.045***	−1.333***		−1.481***	1.221***	

(*continued on next page*)

Table A2 (*continued*)

Alternative Controls	Private ($N = 417$)			Public (N = 417)		
	ln(W)	ln(F)	ln(G)	ln(W)	ln(F)	ln(G)
	(0.716)	(0.313)		(0.334)	(0.164)	

Raw Data (Original Model)	Private ($N = 468$)			Public ($N = 270$)		
	ln(W)	ln(F)	ln(G)	ln(W)	ln(F)	ln(G)
ln(W)		−24.027***	−16.991***		15.097	−6.869
		(5.273)	(4.720)		(32.993)	(4.383)
ln(F)	−0.040***		−0.794***	0.048***		0.554***
	(0.010)		(0.209)	(0.019)		(0.160)
ln(G)	−0.055***	−1.362***		−0.049*	1.037**	
	(0.012)	(0.226)		(0.026)	(0.473)	

References

Abildtrup, J., Garcia, S., Stenger, A., 2013. The effect of forest land use on the cost of drinking water supply: A spatial econometric analysis. Ecol. Econ. 92, 126–136.

Alemu, I., Richards, D.R., Gaw, L.Y.-F., Masoudi, M., Nathan, Y., Friess, D.A., 2021. Identifying spatial patterns and interactions among multiple ecosystem services in an urban mangrove landscape. Ecol. Indic. 121, 107042.

Arrow, K., Dasgupta, P., Goulder, L., Daily, G., Ehrlich, P., Heal, G., Levin, S., Mäler, K.-G., Schneider, S., Starrett, D., 2004. Are we consuming too much? J. Econ. Perspect. 18, 147–172.

Bekele, E.G., Lant, C.L., Soman, S., Misgna, G., 2013. The evolution and empirical estimation of ecological-economic production possibilities frontiers. Ecol. Econ. 90, 1–9.

Brown, T.C., Froemke, P., 2012. Nationwide assessment of nonpoint source threats to water quality. BioScience. 62 (2), 136–146.

Brown, G., Patterson, T., Cain, N., 2011. The devil in the details: non-convexities in ecosystem service provision. Resour. Energy Econ. 33 (2), 355–365.

Burt, T., 1979. Diurnal variations in stream discharge and throughflow during a period of low flow. J. Hydrol. 41, 291–301.

Cavender-Bares, J., Polasky, S., King, E., Balvanera, P., 2015. A sustainability framework for assessing trade-offs in ecosystem services. Ecol. Soc. 20 (1). Available at: http://www.jstor.org/stable/26269756 [Accessed July 19, 2018].

Cohen, F., Hepburn, C.J., Teytelboym, A., 2019. Is natural capital really substitutable? Annu. Rev. Environ. Resour. 44, 425–448.

Cord, A.F., Bartkowski, B., Beckmann, M., Dittrich, A., Hermans-Neumann, K., Kaim, A., Lienhoop, N., Locher-Krause, K., Priess, J., Schröter-Schlaack, C., Schwarz, N., Seppelt, R., Strauch, M., Václavík, T., Volk, M., 2017. Towards systematic analyses of ecosystem service trade-offs and synergies: Main concepts, methods and the road ahead. Ecosyst. Serv. 28, 264–272.

Crocker, T.D., Tschirhart, J., 1992. Ecosystems, externalities, and economies. Environ. Resour. Econ. 2 (6), 551–567.

Dade, M.C., Mitchell, M.G.E., McAlpine, C.A., Rhodes, J.R., 2019. Assessing ecosystem service trade-offs and synergies: the need for a more mechanistic approach. Ambio 48 (10), 1116–1128.

Dittrich, A., Seppelt, R., Václavík, T., Cord, A.F., 2017. Integrating ecosystem service bundles and socio-environmental conditions – A national scale analysis from Germany. Ecosyst. Serv. 28, 273–282.

Fang, Q., Xin, X., Guan, T., Wang, G., Zhang, S., Ma, M., 2022. Vegetation patterns governing the competitive relationship between runoff and evapotranspiration using a novel water balance model at a semi-arid watershed. Environ. Res. 214, 113976.

Fisher, B., Turner, R.K., Morling, P., 2009. Defining and classifying ecosystem services for decision making. Ecol. Econ. 68 (3), 643–653.

Fong, C., Ratkovic, M., Hazlett, C., Imai, K., 2017. CBPS: R Package for Covariate Balancing Propensity Score. Comprehensive R Archive Network (CRAN). Available at the. https://CRAN.R-project.org/package=CBPS.

Hartwick, J.M., 1977. Intergenerational equity and the investing of rents from exhaustible resources. Am. Econ. Rev. 67, 972–974.

Ho, D.E., Imai, K., King, G., Stuart, E.A., 2007. Matching as nonparametric preprocessing for reducing model dependence in parametric causal inference. Polit. Anal. 15 (03), 199–236.

Howe, C., Suich, H., Vira, B., Mace, G.M., 2014. Creating win-wins from trade-offs? Ecosystem services for human well-being: A meta-analysis of ecosystem service trade-offs and synergies in the real world. Glob. Environ. Chang. 28, 263–275.

Imai, K., Ratkovic, M., 2014. Covariate balancing propensity score. J. R. Stat. Soc. Ser. B Stat Methodol. 76 (1), 243–263.

Johansen, L., Taugourdeau, S., Hovstad, K.A., Wehn, S., 2019. Ceased grazing management changes the ecosystem services of semi-natural grasslands. Ecosyst. People 15 (1), 192–203.

Kandziora, M., Burkhard, B., Müller, F., 2013. Interactions of ecosystem properties, ecosystem integrity and ecosystem service indicators—A theoretical matrix exercise. Ecol. Indic. 28, 54–78.

Kragt, M.E., Robertson, M.J., 2014. Quantifying ecosystem services trade-offs from agricultural practices. Ecol. Econ. 102, 147–157.

Lee, H., Lautenbach, S., 2016. A quantitative review of relationships between ecosystem services. Ecol. Indic. 66, 340–351.

Lorilla, R.S., Poirazidis, K., Kalogirou, S., Detsis, V., Martinis, A., 2018. Assessment of the spatial dynamics and interactions among multiple ecosystem services to promote effective policy making across Mediterranean Island landscapes. Sustainability 10 (9), 3285.

Manners, R., Varela-Ortega, C., 2018. The role of decision-making in ecosystem service trade-offs in lowland Bolivia's Amazonian agricultural systems. Ecol. Econ. 153, 31–42.

Markandya, A., Pedroso-Galinato, S., 2007. How substitutable is natural capital? Environ. Resour. Econ. 37, 297–312.

Nalle, D.J., Montgomery, C.A., Arthur, J.L., Polasky, S., Schumaker, N.H., 2004. Modeling joint production of wildlife and timber. J. Environ. Econ. Manag. 48 (3), 997–1017.

Obiang Ndong, G., Therond, O., Cousin, I., 2020. Analysis of relationships between ecosystem services: A generic classification and review of the literature. Ecosyst. Serv. 43, 101120.

Polasky, S., Nelson, E., Lonsdorf, E., Fackler, P., Starfield, A., 2005. Conserving species in a working landscape: land use with biological and economic objectives. Ecol. Appl. 15 (4), 1387–1401.

Raudsepp-Hearne, C., Peterson, G.D., Bennett, E.M., 2010. Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. Proc. Natl. Acad. Sci. 107 (11), 5242–5247.

Reynaud, A., 2003. An econometric estimation of industrial water demand in France. Environ. Resour. Econ. 25, 213–232.

Rouhi Rad, M., Adamowicz, W., Entem, A., Fenichel, E.P., Lloyd-Smith, P., 2021. Complementarity (not substitution) between natural and produced capital: evidence from the Panama canal expansion. J. Assoc. Environ. Resour. Econ. 8, 1115–1146.

Sims, C., Aadland, D., Powell, J., Finnoff, D.C., Crabb, B., 2014. Complementarity in the provision of ecosystem services reduces the cost of mitigating amplified natural disturbance events. Proc. Natl. Acad. Sci. USA 111 (47), 16718–16723.

Spake, R., Lasseur, R., Crouzat, E., Bullock, J.M., Lavorel, S., Parks, K.E., Schaafsma, M., Bennett, E.M., Maes, J., Mulligan, M., Mouchet, M., Peterson, G.D., Schulp, C.J.E., Thuiller, W., Turner, M.G., Verburg, P.H., Eigenbrod, F., 2017. Unpacking ecosystem service bundles: towards predictive mapping of synergies and trade-offs between ecosystem services. Glob. Environ. Chang. 47, 37–50.

Vallet, A., Locatelli, B., Levrel, H., Wunder, S., Seppelt, R., Scholes, R.J., Oszwald, J., 2018. Relationships between ecosystem services: comparing methods for assessing tradeoffs and synergies. Ecol. Econ. 150, 96–106.

Warziniack, T., Sham, C.H., Morgan, R., Feferholtz, Y., 2017. Effect of Forest cover on water treatment costs. Water Econom. Policy 03 (04), 1750006.

White, C., Halpern, B.S., Kappel, C.V., 2012. Ecosystem service tradeoff analysis reveals the value of marine spatial planning for multiple ocean uses. Proc. Natl. Acad. Sci. 109 (12), 4696–4701.

Wossink, A., Swinton, S.M., 2007. Jointness in production and farmers' willingness to supply non-marketed ecosystem services. Ecol. Econ. 64 (2), 297–304.

Yun, S.D., Hutniczak, B., Abbott, J.K., Fenichel, E.P., 2017. Ecosystem-based management and the wealth of ecosystems. Proc. Natl. Acad. Sci. 114, 6539–6544.

Zellner, A., Theil, H., 1962. Three-stage least squares: simultaneous estimation of simultaneous equations. Econometrica 30 (1), 54–78.

Zhang, L., Dawes, W.R., Walker, G.R., 2001. Response of mean annual evapotranspiration to vegetation changes at catchment scale. Water Resour. Res. https://doi.org/10.1029/2000wr900325 [Accessed February 25, 2021].