Global Environmental Change 58 (2019) 101938

Contents lists available at ScienceDirect

Global Environmental Change

journal homepage: www.elsevier.com/locate/gloenvcha

Insights from watershed simulations around the world: Watershed service-based restoration does not significantly enhance streamflow

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ARTICLE INFO

Keywords:
Watershed services
Hydrological ecosystem services
Forest restoration
Forest conservation
Water yield
Dry-season flow

ABSTRACT

Increased water yield and baseflow and decreased peak flow are common goals of watershed service programs. However, is the forest management often used in such programs likely to provide these beneficial watershed services? Many watershed service investments such as water funds typically change less than 10% of watershed land cover. We simulate the effects of 10% forest-cover change on water yield, low flow, and high flow in hydrologic models of 29 watersheds around the world. The forest-cover changes considered are: forest restoration from degraded natural lands or agriculture, forest conversion to agriculture, and forest conversion to urban cover. We do not consider grassland restoration by removal of alien tree species from riparian zones, which does increase water yield and low flow. Forest restoration from locally-predominant agricultural land resulted in median loss in annual water yield of 1.4%. Forest restoration reduced low flow and high flow by ~3%. After forest restoration, low flow increased in ~25% of cases while high flow and water yield declined in nearly all cases. Development of forest to agriculture or urban cover resulted in a 1–2% median increase in water yield, a 0.25–1% increase in low flow, and a 5–7% increase in high flow. We show that hydrologic responses to forest cover changes are not linearly related to climate, physiography, initial land cover, nor a multitude of watershed characteristics in most cases. These results suggest that enhanced streamflow watershed services anticipated from forest restoration or conservation of 10% or less of a watershed are generally modest.

1. Introduction

Watershed services are a key driver of ecosystem service investments, and globally $24 billion was spent to protect, restore, and sustainably manage watersheds in 2016 (Bennett and Ruef, 2016; Brauman, 2015; Bremer et al., 2016). Water funds and similar programs have grown rapidly. In Latin America, there are now more than 40 water funds–payments for watershed service programs in which downstream stakeholders invest in upstream watershed management and hope to receive an improved water supply and other watershed services in return (Bremer et al., 2016). Watershed service programs in the U.S., China, Africa, and elsewhere (Bennett and Ruef, 2016; Melanson, 2014; The Nature Conservancy, 2019; Vogl et al., 2016) often include increased dry-season flow and water yield as primary objectives (Bennett and Carroll, 2014; Bremer et al., 2016; Zheng et al., 2016; Kang, personal communication). Afforestation and reforestation are key activities for 20% of programs, while sustainable forest management and landscape protection, including forest, comprise an additional 30% of program activities (Bennett and Carroll, 2014). Many programs target less than 10% of the watershed for restoration because of limited funds (e.g., Kroeger et al., 2017; Podolak et al., 2017; Vogl et al., 2016).

We note two large counterexamples. South Africa's Working for Water program has removed alien tree species from more than 25,000 km^2 in select watersheds to enhance water quantity by maintaining natural grasslands. These programs have been well-studied and have beneficial effects (Le Maitre et al., 2002; Preston et al., 2018; van Wilgen and Wannenburgh, 2016). China's Sloping Land Conversion Program and Natural Forest Conservation Program conserve forest and reforest agricultural land to maintain water quality and reduce flooding, and have also been extensively studied (Liu et al., 2008; Ouyang et al., 2016).

However, few studies have focused on the effects of smaller watershed service programs on streamflow (e.g., Kareiva et al., 2011; Podolak et al., 2017) rather than water quality (Kroeger et al., 2017; McDonald and Shemie, 2014; Melanson, 2014; Podolak et al., 2017; Vogl et al., 2016). Success for such programs may require realizing promised streamflow changes even when other objectives include carbon, biodiversity, and poverty alleviation (Bremer et al., 2016, 2014a, 2014b; Brouwer et al., 2011; Goldman-Benner et al., 2012; Suyanto et al., 2007). Without those streamflow enhancements, such programs may incur significant reputational risk. Here we investigate

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https://doi.org/10.1016/j.gloenvcha.2019.101938
Received 4 July 2018; Received in revised form 15 May 2019; Accepted 18 June 2019
0959-3780/

whether there are significant impacts of such forest change on the desired outcomes of increased water yield and dry-season flow under realistic forest restoration or conservation to inform watershed service investments.

1.1. Experimental evidence of hydrologic response to forest change

Many studies have used a paired-catchment approach to investigate the annual hydrologic response to afforestation, increasing native tree cover or introducing plantation monocultures of alien trees, or deforestation, removing alien trees or destroying natural forest. Such studies found that an increase in forest cover reduces annual water yield and a decrease in forest cover increases annual water yield (Bosch and Hewlett, 1982; Brown et al., 2005; Filoso et al., 2017; Sahin and Hall, 1996; Stednick, 1996; Whitehead and Robinson, 1993). However, the effects varied strongly across sites–water yield changes varied by more than 500 mm within the same vegetation type or for similar precipitation (Brown et al., 2005). The differences also vary over time, with deforestation sites trending back to the pre-treatment baseline as trees regrow (e.g., Baker, 1986), but the direction of change varying interanually (e.g., Brown, 1971; Clary et al., 1974). Forest changes in the riparian area have larger effects than those in the uplands in many cases (Everson et al., 2007; Scott, 1999; Scott et al., 2004). Studies suggest that we cannot statistically discriminate changes in water yield after forest change of less than 20% of the watershed area (Bosch and Hewlett, 1982; Stednick, 1996), but it may be feasible to predict the change in water yield for smaller land-cover change (Brown et al., 2005).

A few paired catchment studies investigate the highly variable changes in low flow after land-cover change. Brown et al. (2005) and Scott and Lesch (1997) found that afforestation decreased low flow by 10–90%. Ogden et al. (2013) and Price et al. (2011) found that forested watersheds had higher low flows than similar watersheds with land cover such as pasture and secondary forest. Biederman et al. (2015) found minimal changes in low flow after tree die-off. Agricultural and urban development have similarly complex effects on streamflow, with both water yield and low flow increasing or decreasing depending on conditions (Bhaskar et al., 2016; Grogan et al., 2017; Koelliker, 1998; McGrane, 2016; Mukherjee et al., 2018).

Instead of studying change in individual watersheds, catchment classification investigates the relationship between forest cover and hydrologic behavior by considering a larger number of watersheds relatively unaffected by humans. Globally, on average, tropical and temperate forested catchments have lower water yield than non-forested catchments in the same climate (Peel et al., 2010; Zhang et al., 2001). However, both Peel et al. (2010) and Zhang et al. (2001) show significant overlap in ET values for forests and non-forests in most conditions.

1.2. Hydrologic simulation under land-cover change has typically been site- or region-specific

Hydrologic simulation is the primary alternative to experimental studies for assessing the effects of land-cover change on flow. These models allow controlled land-cover changes while holding other factors constant, but require significant simplifications, including to groundwater, riparian areas, and soil structure. Continuing work in hydrology seeks to address these shortcomings while remaining tractable (Bailey et al., 2016; Easton et al., 2008; Strauch and Volk, 2013; White et al., 2011). Despite these simplifications, hydrologic simulation with many models and across many sites has shown great promise and strong results for building understanding about the mechanisms and specific hydrologic responses to land-cover change (Khoi and Suetsugi, 2014a; Liang et al., 1994; Logsdon and Chaubey, 2013; Schoups et al., 2006; Somura et al., 2012; Strauch et al., 2013; Volk et al., 2010; Wada et al., 2011; Warburton et al., 2012; Zhang et al., 2015). However, no previous work has built a global understanding of the hydrologic response to forest change across the broad variety of climates, land covers, and geology from which people obtain important watershed services from a consistent modeling framework. This is key to answering the question addressed in this paper: Are there significant impacts of forest restoration and conservation as practiced by many watershed service programs on water quantity watershed services?

In this work we explore the streamflow effects of forest restoration and conservation based on hydrologic simulation of 29 site models from around the globe and different climate zones. For each site model, we analyze the effect of 10% land-cover change, an amount which 1) allows controlled comparison across sites, and 2) is in the upper range of land-cover change for which watershed service investments are typically made (Kroeger et al., 2017; Ma et al., 2016; Podolak et al., 2017; Vogl et al., 2016). We explore three land-cover change scenarios: “forest restoration”, in which degraded natural lands and agriculture are replaced by natural forest forest; “agricultural development”, in which forest is converted to agriculture; and “urban development” in which forest is converted to urban cover. The latter two scenarios may represent the inverse of forest conservation scenarios in many cases. We investigate whether hydrologic response to forest restoration and conservation is related to climate, physiography, initial or changed land cover, and other watershed characteristics.

2. Methods

We use a novel dataset of 29 watershed site models across global gradients in climate and land cover (Fig. 1), perform daily timestep hydrologic simulations before and after land-cover changes for each site, and explore the response to the scenarios across these gradients. Details on the watersheds are available in Appendix A.1 and the sites are described in Appendix A.2. The site models were collected from the published literature by contacting site-model authors and requesting their model data, including the calibrated SWAT models, model input, and calibration data. We systematically assess these hydrologic simulations under forest cover changes to investigate the effects on desired water quantity watershed services, here represented by greater annual water yield, greater low flow, and reduced high flow.

2.1. Model implementation

2.1.1. Soil & Water Assessment Tool

The 29 watershed site models were rebuilt in SWAT2012 revision 637 to enable consistent modeling of the hydrologic response to land-cover change. The Soil & Water Assessment Tool, SWAT (Arnold et al., 1998; Gassman et al., 2007), simulates rainfall-runoff and other hydrologic processes including plant growth and in-channel processes to represent watershed behavior as well as human water applications. SWAT’s vertical water balance is based on Hydrologic Response Units, HRUs, that represent a distinct combination of land cover, soil and slope in a subbasin, itself a unique area of the watershed. Stream channels are connected across subbasins. HRUs are not necessarily spatially contiguous, and each is connected directly to the stream channel. The site model is calibrated to observed discharge, and spatial feedbacks are incorporated into the calibration parameters. SWAT has been widely used for water resources assessment (Francesconi et al., 2016; Gassman et al., 2007), with more than 3500 publications using the model. The watershed sites modeled here typically include between 100 and 1500 HRUs and 15 and 150 subbasins (Table 1).

SWAT partitions precipitation that reaches the surface based on the curve number method (Rallison and Miller, 1982), which is dynamically updated based on the soil moisture. Soil water percolation into groundwater through layers is represented by linear and non-linear reservoirs. Soil water is transpired based on energy demand from potential evapotranspiration and a root density function that affects plant root access depth. Total runoff includes overland flow from water that does not

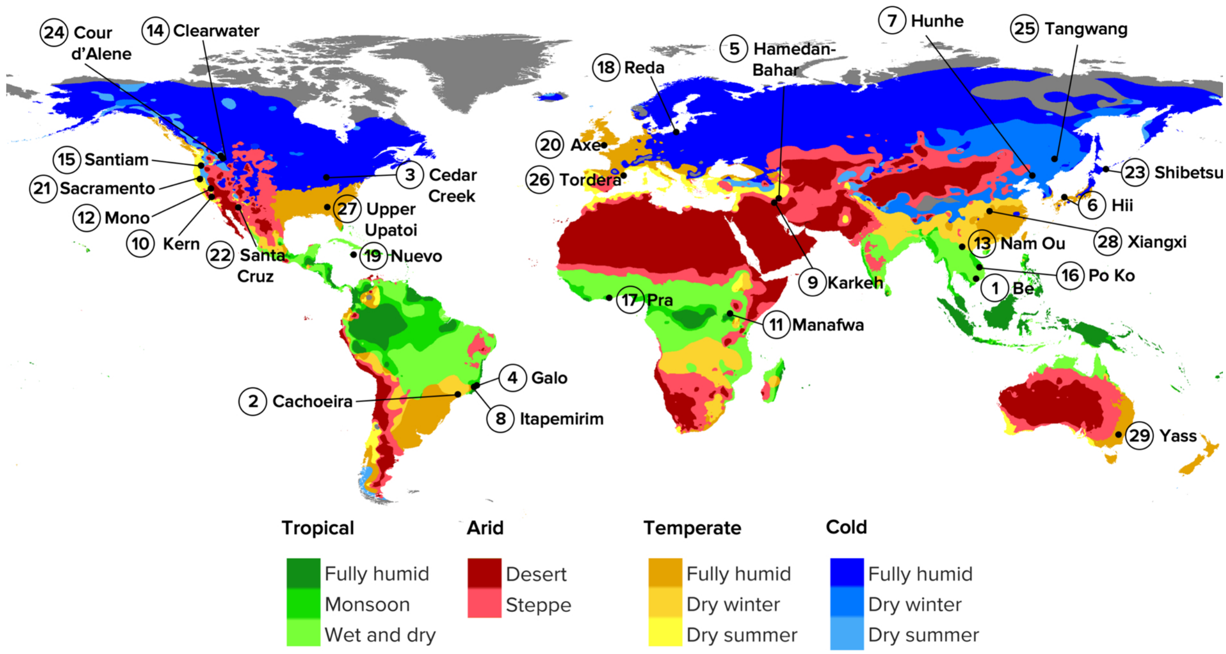

Fig. 1. 29 Hydrologic site models on 6 continents in 11 second-order Köppen-Geiger Climate Zones. The sites are mapped here against the Köppen-Geiger Climate Zone map from Peel et al. (2007), with names and numbers shown in Table 1.

infiltrate, soil lateral flow, and discharge from a groundwater layer represented as a linear reservoir. SWAT estimates evapotranspiration using a leaf area index-modified crop factor, available water, and potential evapotranspiration. SWAT's sophisticated vegetation model responds to both climatic and nutrient stresses (Williams, 1995)

2.1.2. Site model selection

Watershed selection was stratified using the Köppen-Geiger Climate Zone map (Peel et al., 2007) and distributed across different geographies. The selected watersheds are on 6 continents and in 11 second-order Köppen-Geiger Climate Zone non-polar classes. Watersheds covered a median area of 1900 km^2, with a range of 50 - 20,000 km^2. These watersheds are larger than experimental catchments but smaller than large sub-continental watersheds. They cover the area range often considered for watershed service programs. Each watershed has at least five years of calibration data, and most significantly more, with a median of 15 years. Only three sites have solely monthly data, though 12 were initially calibrated using monthly data. For the other cases with initial monthly calibration, we calibrated them on daily data available from the authors or government sources, described in Appendix A.3.

Selected cases have a Nash-Sutcliffe Coefficient (Nash and Sutcliffe, 1970) of 0.5 for daily calibration data and 0.7 for monthly calibration data in the original work to assure satisfactory model performance. Simulated water balance closure was checked during calibration, and all results had an acceptable total volumetric bias within 20% accounting for known point sources and groundwater abstraction (Moriasi et al., 2007). The calibrated models had Nash-Sutcliffe efficiencies between 0.5 and 0.77 and R^2 between 0.54 and 0.92 at the daily scale, where such data was available. In addition to the summary statistics, model plant growth and ET were tested by checking watershed crop type, irrigation amounts and sources, and average monthly LAI against global datasets to assure reasonable land cover representations (Monfreda et al., 2008; Myneni et al., 2015; Ramankutty et al., 2008; Siebert et al., 2013). Post-calibration curve number values were checked against standard value ranges for each soil-cover complex (Soil Conservation Service, 2004) and had a median difference of ~ -2% and a median absolute difference of ~4%, considered reasonable for models calibrated to observed data. These tests are discussed further in Appendix A.3.

These site models represent watersheds with a variety of land covers, geologic and pedologic conditions, and different degrees of management. Table 1 provides details of the sites, their climate, area, simulation period, calibration statistics, and primary citations. More information on the sites, including their climates, land cover distributions, soil characteristics, and existing infrastructure and irrigation schemes are available in Appendix A.1, as is further information about SWAT. Further detail regarding model development is in Appendix A.3.

2.1.3. Land-cover changes

We identify "forest restoration", "agriculture development", and "urban development" land-cover changes for each site. These land-cover changes represent plausible conversions in each watershed depending on whether there is: 1) effort to restore forest for watershed services; 2) conversion of forest to agriculture in the watershed; or 3) development of urban land cover. "Restoration" focuses on reforestation of native-type forest. In contrast, "agriculture" and "urban" land-cover changes expand either agricultural or urban area while reducing the area of forest. Table 2 details the forest cover changes made in each site. When either expanding or reducing agricultural land cover, the irrigation scheme for the crop was maintained. Similarly, withdrawals and wastewater discharges were increased and decreased proportionally with the associated land covers.

Scenarios employing the forest cover changes are applied to calibrated models using SWAT's land-use update feature (Arnold et al., 2010). This feature shrinks and grows HRUs within subbasins to represent land-cover change. We promote the most realistic forest cover change by prioritizing land-cover changes in HRUs with similar slopes and soils. Site models are run with pre- and post- change land cover, without transitional land-cover progression, as our goal is to quantify the long-term effects of forest-cover change.

We impose forest cover changes of 10% of watershed area. In some cases, it was not possible to implement a 10% forest cover change because: 1) the SWAT land-use update feature requires that urban land-cover exist in the watershed before it can be expanded; or 2) there was not 10% of one land cover that could be replaced by another. In those cases, we imposed the maximum possible areal change and extrapolated the resulting changes in streamflow index values (discussed presently) to a 10% land-cover change for comparison across sites. The

Table 1
List of the SWAT site models, their basic characteristics, and simulation period.

Site	Country	Climate Zone[1]	Area [km^2]	Latitude [º]	Longitude [º]	Simulation years	Calibration NSE[2]	Subbasins[3][#]	HRUs[4][#]	Primary reference(s)
1. **Be** River	Vietnam	Aw	7347	11.8	107.1	1981-1989	**0.77**	90	341	Khoi and Suetsugi (2014b, 2014a)
2. **Cachoeira** River Basin	Brazil	Cw	291	−23.0	−46.3	1987-2001	**0.68**	85	1375	Rodrigues et al. (2014)
3. **Cedar Creek** Watershed	U.S.	Df	706	43.0	−90.0	1993-2013	**0.57**	42	295	Kumar and Merwade (2009)
4. **Galo** Creek	Brazil	Am	942	−20.3	−40.7	1997-2003	**0.52**	29	342	Pereira et al. (2014a, 2014b)
5. **Hamedan-Bahar** Watershed	Iran	BS	2387	34.9	48.5	1993-2008	0.59	64	197	Akhavan et al. (2010)
6. **Hii** River Basin	Japan	Cf	914	35.2	132.9	1988-2006	**0.68**	14	385	Somura et al. (2012, 2009)
7. **Hunhe** Catchment	China	Dw	9285	46.8	132.0	1958-1965	0.88 **(0.50)**	30	161	Zhang et al. (2015)
8. **Itapemirim** River	Brazil	Am	2237	−20.5	−41.5	1993-2000	**0.69**	17	126	Fukunaga et al. (2015)
9. **Karkeh** Watershed	Iran	BW	20808	32.5	48.0	1990-2002	0.71	84	882	Vaghefi et al. (2015, 2014)
10. **Kern** River	U.S.	BW	6234	35.6	−118.4	1950-2005	0.85 **(0.57)**	33	184	Ficklin et al. (2012b, 2013a)
11. **Manafwa** River Basin	Uganda	Aw	353	0.9	34.2	1956-1961	**0.50**	19	116	Mutenyo et al. (2011)
12. **Mono** Lake Catchment	U.S.	Cs	400	37.9	−119.1	1950-1992	0.80 **(0.51)**	16	197	Ficklin et al. (2013b)
13. **Nam Ou** Basin	Laos	Cw	26180	21.2	102.3	1994-2003	**0.60**	19	620	Shrestha et al. (2013)
14. **Clearwater** River North Fork	U.S.	Ds	3355	46.7	−115.3	1970-2005	**0.75**	175	684	Ficklin et al. (2012a)
15. North **Santiam** River	U.S.	Cs	558	44.6	−121.9	1950-2005	**0.53**	131	491	Ficklin et al. (2012a)
16. **Po Ko** Catchment	Vietnam	Am	3232	16.0	107.5	2000-2011	**0.66**	43	370	Tram et al. (2014)
17. **Pra** River Basin	Ghana	Aw	19584	6.4	−1.4	1964-1991	0.74	27	158	Kankam-Yeboah et al. (2013)
18. **Reda** Catchment	Poland	Df	483	54.6	18.1	1998-2006	**0.68**	30	465	Piniewski et al. (2014)
19. Rio **Nuevo**	Jamaica	Af	95	18.3	−77.0	2002-2007	0.63	20	148	Goyal et al. (2015)
20. **Axe** River	U.K.	Cf	400	50.8	−3.0	1988-1997	**0.56**	25	198	Glavan et al. (2012, 2011)
21. **Sacramento** River	U.S.	Ds	18836	41.3	−120.9	1950-2005	0.76 **(0.63)**	82	636	Ficklin et al. (2013a, 2012b)
22. **Santa Cruz** Watershed	U.S.	BS	9090	31.8	−110.8	1987-2006	0.77	131	2705	Niraula et al. (2015, 2012)
23. **Shibetsu** Watershed	Japan	Df	672	43.6	144.9	2004-2008	**0.54**	77	280	Jiang et al. (2014, 2011)
24. **Cour d'Alene** South Fork	U.S.	Ds	733	47.5	−116.1	1991-2010	**0.70**	23	555	Praskievicz (2016)
25. **Tangwang** Catchment	China	Dw	20807	46.7	129.9	1965-2000	0.87 **(0.67)**	121	801	Zhang et al. (2015)
26. **Tordera** Catchment	Spain	Cs	858	41.8	2.6	1996-2004	**0.50**	17	158	Pascual et al. (2014)
27. **Upper Upatoi** Watershed	U.S.	Cf	878	32.5	−84.7	2004-2011	**0.61**	39	1270	Dennedy-Frank et al. (2016)
28. **Xiangxi** River	China	Cw	3175	31.2	110.8	1981-1993	**0.64**	38	797	Bieger et al. (2015, 2014, 2013, 2012)
29. **Yass** River Catchment	Australia	Cf	1597	−36.0	147.5	1993-2011	**0.65**	20	482	Saha et al. (2013); Saha and Zeleke (2014)

[1] Af: Tropical fully humid; Am: Tropical monsoon; Aw: Tropical dry winter; BW: Arid desert; BS: Arid steppe; Cf: Temperate fully humid; Cs: Temperate dry summer; Cw: Temperate dry winter; Df: Cold fully humid; Ds: Cold dry summer; Dw: Cold dry winter.
[2] Bold indicates daily calibration; parantheses indicate daily calibration at the same site.
[3] Subbasins in model.
[4] Hydrologic response units in model.

Table 2
Forest cover transitions and percent change for forest restoration, agricultural development, and urban development scenarios.

Site	IGBP land cover classification	Forest restoration		Agricultural development		Urban development	
		Type	Percent	Type	Percent	Type	Percent
Be River	Cropland/natural vegetation mosaic	Agriculture →Evergreen forest	**10**	Evergreen forest→ Agriculture	**10**	N/A[1]	
Cachoeira River Basin	Woody savannas	Pasture →Mixed forest	**10**	Mixed forest→ Pasture	8.2	Mixed forest → Urban	4.8
Cedar Creek Watershed	Cropland/natural vegetation mosaic	Row crop agriculture → Deciduous forest	**10**	Deciduous forest → Row crop agriculture	**10**	Deciduous forest → Urban	8.6
Galo Creek	Evergreen broadleaf forest	Pasture, agriculture → Primary forest	**10**	Primary forest → Pasture, agriculture	**10**	N/A[1]	
Hamadan-Bahar Watershed	Croplands	Winter wheat → Deciduous forest	6.4	N/A[1]		N/A[1]	
Hii River Basin	Mixed forest	Rice[2]→ Mixed forest	**10**	Mixed forest → Rice[2]	**10**	Mixed forest → Urban	**10**
Hunhe Catchment	Mixed forest	Winter wheat[2]→ Mixed forest	9.3	Mixed forest → Winter wheat[2]	**10**	N/A[1]	
Itapemirim River	Savannas	Pasture → Secondary forest	**10**	Secondary forest → Pasture	**10**	Secondary forest → Urban	7.0
Karkeh River Basin	Croplands	Crop/woodland mosaic → Mixed forest	7.8	N/A[1]		N/A[1]	
Kern River	Open shrubland	Brush range → Evergreen forest	**10**	Evergreen forest → Row crop agriculture	7.8	Evergreen forest → Urban	**10**
Manafwa River Basin	Croplands	Corn → Mixed forest	8.2	N/A[1]		N/A[1]	
Mono Lake Catchment	Grasslands	N/A[1]		Evergreen forest → Brush range	**10**	Evergreen forest → Urban	**10**
Nam Ou Basin	Evergreen broadleaf forest	Crop mosaic → Mixed medium cover forest	5.3	Mixed medium cover forest → Crop mosaic	8.1	Evergreen woodland/shrubland → Urban	**10**
North Fork Clearwater River	Evergreen needleleaf forest	N/A[1]		Evergreen forest → Switchgrass	**10**	Evergreen forest → Urban	**10**
North Santiam River	Evergreen needleleaf forest	Brush range →Evergreen forest	6.2	Evergreen forest → Brush range	**10**	Evergreen forest → Urban	**10**
Po Ko Catchment	Evergreen broadleaf forest	Row and close-grown crop agriculture → Evergreen forest	6.5	Evergreen forest → Close-grown crop agriculture	8.0	N/A[1]	
Pra River Basin	Cropland/natural vegetation mosaic	Dryland cropland/pasture → Evergreen broadleaf forest	**10**	Evergreen broadleaf forest → Dryland cropland/ pasture	**10**	N/A[1]	
Reda Catchment	Evergreen needleleaf forest	Oats → Evergreen forest	**10**	Evergreen forest → Oats	**10**	Evergreen forest → Urban	**10**
Rio Nuevo	Cropland/natural vegetation mosaic	Fields → Broadleaf forest	5.3	Broadleaf forest → Fields	5.6	Broadleaf forest → Fields	8.5
River Axe	Grasslands	Hay silage → Deciduous forest	**10**	Deciduous and evergreen forest → Hay silage	4.0	N/A[1]	
Sacramento River	Evergreen needleleaf forest	Brush range → Evergreen forest	**10**	Evergreen forest → Hay	**10**	Evergreen forest → Urban	**10**
Santa Cruz Watershed	Open shrubland	Grass range → Evergreen forest	5.4	Evergreen forest → Agriculture	**10**	Evergreen forest → Urban	**10**
South Fork Cour d'Alene	Evergreen needleleaf forest	Brush range → Evergreen forest	5.1	Evergreen forest → Agriculture	**10**	Evergreen forest → Urban	**10**
Shibetsu Watershed	Mixed forest	Pasture → Mixed forest	**10**	Mixed forest → Pasture	**10**	N/A[1]	
Tangwang Catchment	Mixed forest	Winter wheat → Mixed forest	4.1	Mixed forest → Upland cotton	4.9	Mixed forest → Urban	**10**
Tordera Catchment	Mixed forest	Grass crops → Mixed forest	9.9	Mixed forest → Grass crops	**10**	Mixed forest → Urban	**10**
Upper Upatoi Watershed	Mixed forest	Grass range → Evergreen forest	**10**	Evergreen forest → Grass range	**10**	Evergreen forest → Urban	**10**
Xiangxi River	Mixed forest	Rapeseed → Mixed forest	**10**	Mixed forest → Rapeseed	**10**	N/A[1]	
Yass River Catchment	Grasslands	Grass range → Evergreen forest	**10**	Evergreen forest → Close-grown agriculture	**10**	Evergreen forest → Urban	**10**

[1] N/A indicates that a transition could not be made because a 4% land-cover change could not be achieved.
[2] Irrigated agriculture.

site was exempted from the scenario if under 4% of the watershed area could be changed. See Table 2 for the land-cover changes made at each site.

2.1.4. Simulation indices

Simulated responses to forest cover change are compared using three streamflow indices (Richter et al., 1996):

- the mean annual water yield for the calendar year (**indicator of total flow magnitude**),
- number of periods of continuous flow above 7 times the median flow, (**indicator of high flow**), hereafter referred to as flow events above 7 times the median flow, and
- average annual minimum 7-day flow (**indicator of low flow**).

These indices provide estimates of the hydrologic response at both long- and short-period timescales and quantify total, high-flow, and low-flow watershed service processes. They also have relatively simple interpretations, and allow for intuitive reasoning, unlike more complex indices (Poff, 1996). Annual water yield is the average discharge per area per year at the watershed outlet. High flow is represented by the number of events with flow greater than 7 times median flow before application of land-cover change. The low flow indicator is measured during the continuous 7-day periods of minimum discharge annually. Calendar years are used in all calculations for simplicity given the different onset of water years across the world. We report the percent changes in the streamflow indices' responses to land-cover change, which allows comparison across sites.

2.2. Index comparison and statistics

We analyze the effects of forest change in the models by comparing the percent change in the indices between the site model with forest change and without forest change. We compare these through inspection of the distribution of responses and their geography, as well as both simple and sophisticated statistical approaches.

2.2.1. Simple linear regression and analysis of variance

We use simple linear regression to test the ability of many watershed characteristics to explain the variance seen in the hydrologic response to forest change. The coefficient of determination (R^2) represents the portion of the variance in the streamflow index change explained by the watershed characteristic. The watershed characteristics examined as potential explanatory factors include:

- Climate: Average and per-storm precipitation, annual and monthly potential evapotranspiration, and seasonal dryness
- Physiography: watershed area, perimeter, geographic location, slope, elevation, shape
- Soil: available water content, saturated hydraulic conductivity, depth, percent clay, sand, and soil
- Land cover: percent forest, agriculture, pasture, urban; watershed increase in forest, agriculture, pasture, urban; watershed decrease in forest, agriculture, pasture, urban.

We use a multiple hypothesis testing approach to avoid false positives that arise when performing many statistical tests. The k-family-wise error rate (Lehmann and Romano, 2012) is the probability of finding k or more false positives in multiple hypothesis tests, and provides a stricter criteria for statistical significance than the traditional 5% threshold, discussed further in Appendix A.4.1.

We also use one-way analysis of variance (ANOVA) to test the hypothesis that streamflow index value changes have different means depending on: 1) the forest change group based on forest type and developed land-cover type (e.g., agriculture to evergreen forest vs. rangeland to broadleaf forest); or 2) the Köppen-Geiger climate zone and associated seasonality.

2.2.2. Multiple linear regression

We use a multiple linear regression approach to test the combined linear predictive power of these same watershed characteristics (section 2.2.1) on the streamflow indices. The approach uses the Least Absolute Shrinkage and Selection Operator ("LASSO")(Tibshirani, 1996) and cross-validates the results. LASSO simultaneously: 1) selects the best set of predictor variables from selected watershed characteristics (Appendix A.4.2) as well as indicator variables for climate zone, climate seasonality, land-cover class, and forest change group; and 2) and estimates the regression parameter values for each predictor. The six-fold cross-validation trains the LASSO-based model with 5/6 of the data and tests it with the other 1/6, and is repeated 500 times. A coordinate descent algorithm varies the penalty value to select the subset of predictor variables that provides the maximum predictive power (Friedman et al., 2010; Pedregosa et al., 2011).The testing stage R^2 represents the predictive power of the regression model for data on which it was not trained.

2.2.3. Identifying non-linear changes in watershed responses: permutation tests

In addition to the univariate (2.2.1) and multivariate (2.2.2) linear statistical models, we also use a two-sample permutation test (Good, 2005) to examine non-linear relationships between streamflow index value changes and watershed characteristics. In particular, we test if the difference in the mean of the streamflow index value changes between two groups, selected based on the value of a watershed characteristic, is unlikely to occur by chance (see Appendix A.4.3). Based on this permutation analysis we calculate the p-value, representing the portion of permutations for which the mean value is larger than that observed, and Cohen's d, representing the size of the difference in mean values relative to a pooled variance (Basilevsky, 1994). Low p-values suggest that such a large difference is unlikely, and large Cohen's d values indicates that the difference is consequential.

3. Results and discussion

We present the results in two steps. First, we describe the general pattern of land-cover change impacts on water yield, low flow and high flow in the watershed site models and then particular cases where land-cover changes had a greater effect on hydrologic response. Second, we review the results of the statistical tests and show that there are few linear dependencies of the hydrologic response on watershed characteristics, and some apparent non-linear relationships.

3.1. Patterns of hydrologic response to land-cover change

3.1.1. Insensitivity to land-cover change

Here we present estimates of hydrologic response to land-cover change. Fig. 2 summarizes the change in water yield, low flow, and high flow at all 29 sites for three scenarios: forest restoration, agricultural development, and urban development. For a 10% forest-cover change, median water yield is reduced by 1.4% for the restoration case, and for agricultural and urban development water yield increased by 1.8% and 1.1%, respectively. The median low flow reduction for restoration was 3%, and increases in low flow for the agriculture and urban cases were 1.1% and 0.2%. Median high flow decreased by 3.7% for restoration, and increased for agriculture and urban cases 7.1% and 5.5%. Even considering the interquartile range, we see modest if any beneficial impacts on all three streamflow indices. This is consistent with evidence from paired-catchment assessments that suggest they cannot statistically discriminate streamflow changes after forest-cover changes of less than 20%. Cases showing large changes in water yield are discussed in section 3.1.2.

The water yield changes shown are consistent with a simple water-

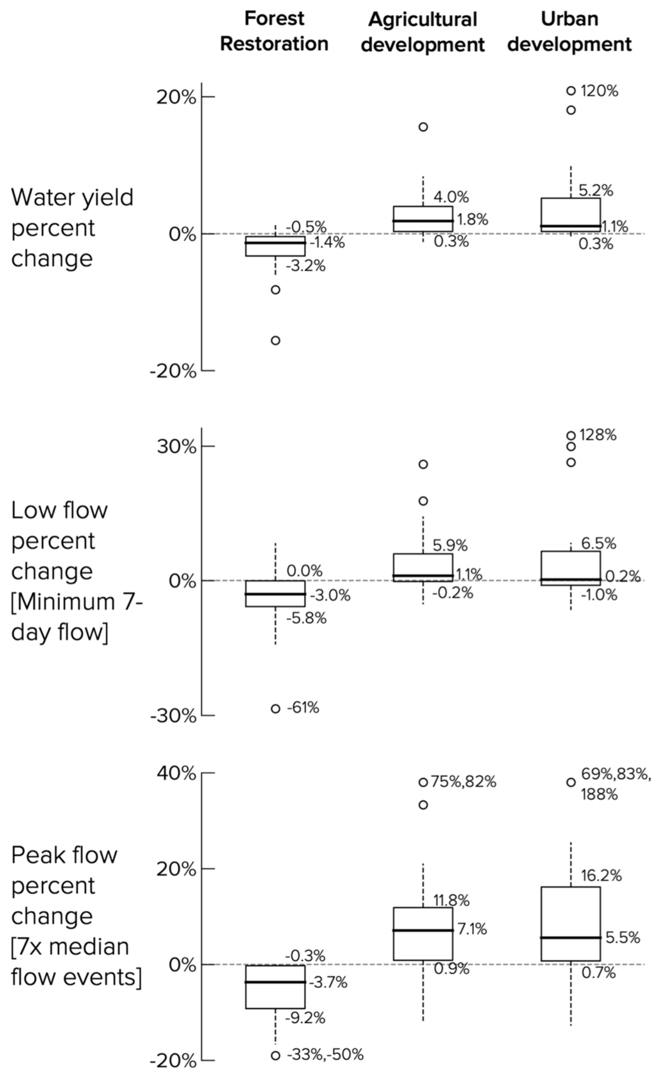

Fig. 2. The response of water yield, low flow, and high flow are relatively small for a 10% forest change, though there are a few larger changes. The boxplots summarize the percent change in the streamflow indices under three land-cover change scenarios: forest restoration from agricultural or degraded lands, agricultural development from forest, and urban development from forest, and three measures: water yield; the minimum 7-day flow, which represents low flow; and the number of events at 7x median flow, which represents high flow. Box plots show the median and interquartile range (± 25%), which are labeled, and whiskers at 1.5 x the interquartile range. The rest of the sites are presented as circles, and are labeled when they extend beyond the plot bounds. Note that beneficial effects would be increased water yield, increased low flow, and decreased high flow, which are limited in these results.

balance model shown in Appendix A.5. This model shows that water yield changes are unlikely to be appreciable for most water-fund style land-cover change investments unless the difference in water yield between the restored and unrestored land covers is larger than 50%, which may not be common under the same climate (e.g., the relatively low difference of evapotranspiration for locations with the same precipitation or aridity in Peel et al., 2010; Zhang et al., 2001).

Fig. 3 shows the changes in water yield, low flow, and peak flow for each watershed for which land-cover change was simulated. We note two key observations.

First, there is substantial consistency in whether water yield and peak flow increase for each forest cover change. For restoration, 96% of watersheds showed a decrease in water yield and all sites showed decreased peak flow. For agriculture and urban, 93% and 94% of watersheds showed an increase in water yield while 84% and 88% of sites showed increased peak flow. Low flow had a more variable response, with 78% of sites decreasing under restoration, and 69% and 56% increasing under agricultural and urban development, respectively.

Second, there are few spatial patterns of consistent hydrologic response. Two examples show this lack of pattern. In Brazil there are three basins in close proximity. The Galo and Itapemirim are both in the tropical climate zone, but have very different magnitudes of water yield change: 4.6% vs. -0.5% and 4.3% vs. 0.6% under forest restoration and agricultural development, respectively. In contrast, the Galo and Cachoeira each start with about 50% forest cover, as opposed to ~20% for Itapemirim. They have similar changes: -4.6% vs. -4.4% and 4.3% vs. 4.9% under restoration and agricultural development. In general, the original land cover tends to be related to water yield change more closely than the climate zone, as shown in Appendix A.6. In four close sites in southern China, Laos, and Vietnam, there is a reduction in water yield when forest cover is restored and an increase when agriculture is developed. However, under forest restoration the Po Ko in Vietnam experiences a much larger decrease in water yield than the other three. One exception to the lack of consistent spatial pattern is sites on the west coast of the US that show consistently limited water yield decreases for the forest restoration case while agricultural consistently increases and urban development consistently decrease water yields, but by a limited amount.

The direction of hydrologic response (increase or decrease) generally can be explained based on simple hydrologic behavior, even if the response is small. First consider water yield. After forest restoration water yield generally decreases. The forest vegetation evapotranspires more water than either non-irrigated agricultural or urban land cover. Irrigated agriculture can evapotranspire a great deal of water, but in these sites irrigation water may be sourced from deep aquifers that increase the water available for both evapotranspiration and stream discharge. We note that this is not a sustainable process, as eventually the limited store of water will be depleted; in addition, it may reduce downstream flow (Faunt, 2009; McGuire, 2017) or cause land subsidence (Erban et al., 2014). For agricultural development, less water is lost because crops are fallow during parts of the year and have shallower roots, which results in an increased water yield compared to forest. For urban development, the increase in impervious surface typically results in more surface flow and reduced infiltration and recharge (McGrane, 2016). Water yield increases as less water is lost to evapotranspiration.

Next consider low flow, which typically shows a limited response to forest-cover change. After forest restoration, the improved soils of the forest increase infiltration. However, in most cases low flow still decreases because nearly all of the resulting soil water is evapotranspired, leaving little to recharge aquifers for dry season discharge. In a few sites the water infiltrates through the soil to recharge the aquifer and contributes to low flow. The Karkeh basin, where irrigation is withdrawn from the shallow groundwater, is an exception because the aquifers can contribute water to low flow that would have otherwise been used for irrigation. Agricultural development results in increased low flow because some soil water is inaccessible to shallow-rooted crops, and in some cases dry-season irrigation supplements low flow in the site models. Urban development decreases transpiration more than recharge, increasing low flow in most sites.

High flow shows the strongest response to land-cover change. The decrease in high flow under forest restoration is attributable to increased infiltration because of available soil water storage and greater permeability, represented by a lower curve number (Rallison and Miller, 1982) for forest land cover. Under agricultural development high flow increases because reduced available soil moisture storage and shallow rooting depths lead to increased runoff and interflow (rapid flow). Urban development increases overland flow because of reduced infiltration and a consequent reduction in soil moisture storage.

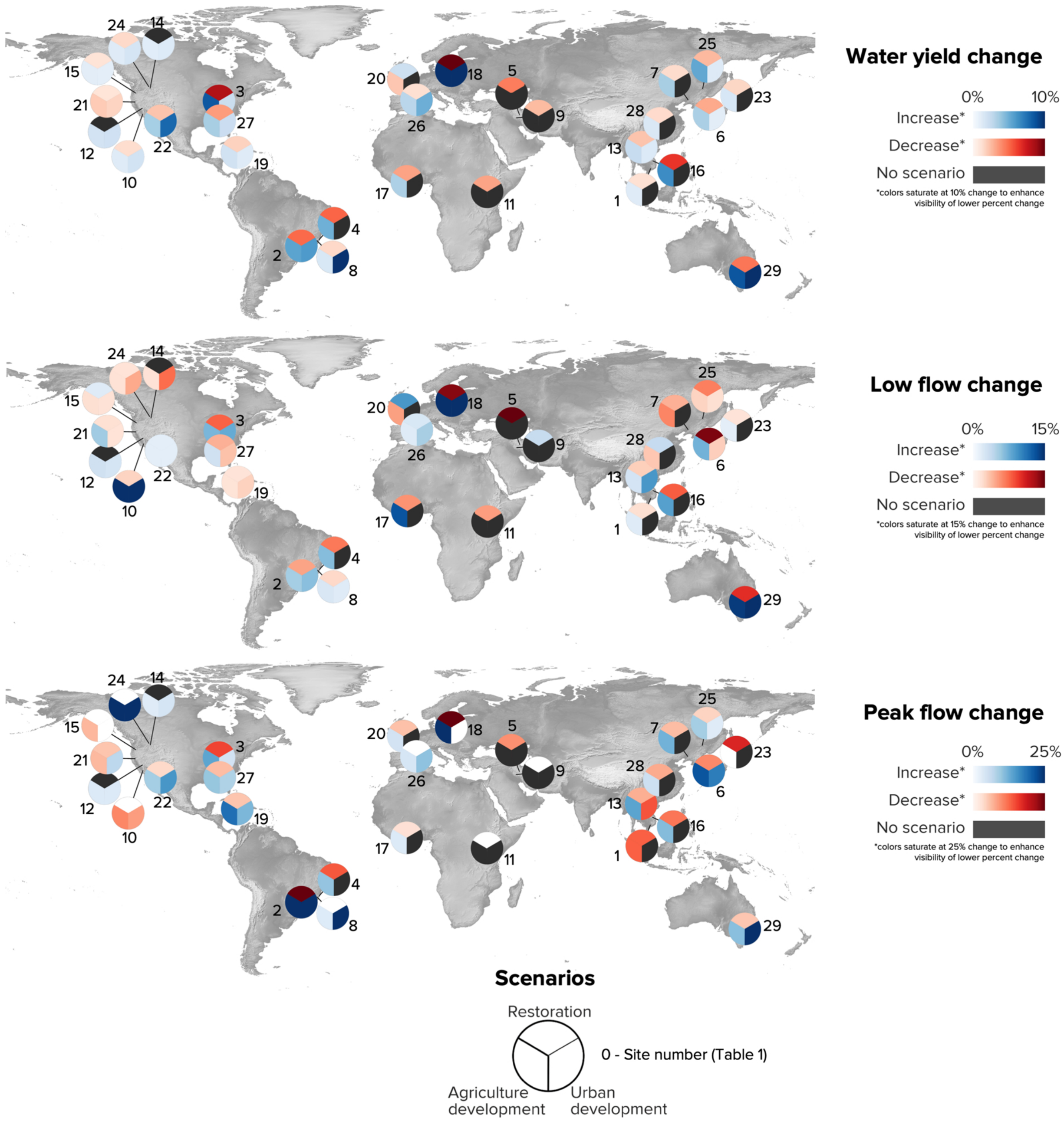

Fig. 3. Water yield, low flow, and peak flow changes in three scenarios of 10% land-cover change at 29 sites around the globe. Forest restoration, agriculture development from forest, and urban development from forest scenarios are run with 10% change. When 10% land-cover change cannot be achieved the results from smaller changes are linearly extrapolated to 10% (done in 35% of scenarios, with an average of 6.5% change); for changes less than 4% no scenario is run as noted. These low land-cover change scenarios occur when there is not any growing or enough shrinking land cover in the SWAT model; such scenarios are especially common for the urban development scenario type because a number of the sites lack any urban land cover to start. Color scales are cut off as noted in the figures to enhance color visibility for smaller flow changes; those with larger changes are specifically discussed in the text. Numbers correspond to numbering in Table 1 and Fig. 1. Background is shaded-relief map (Patterson and Kelso, 2015).

3.1.2. *Cases where land-cover change strongly affects hydrologic response*

Overall the streamflow indices show single-digit percent changes in response to land-cover change, but a few anomalously large changes provide important information about sites with larger effects. Two watersheds see water yield reductions larger than 8% in the forest restoration scenario: the Reda catchment in Poland and the Cedar Creek Watershed in the US. Both sites transition from agricultural crops to forest in cold, fully-humid climates. The Reda has a much stronger reduction in water yield, of about 16% versus 8% for Cedar Creek. Reda has a smaller water yield, so similar absolute yield reduction occurs in both sites. The Yass River catchment sees a water yield increase of a similarly large 8% after forest is developed to agriculture. However, this eastern Australian basin in a temperate, fully humid climate only experiences a 4% decrease in water yield after forest restoration, perhaps because pasture rather than agriculture is converted to forest.

The Hamadan-Bahar in Iran has an anomalously large decrease in low flow of 61% under restoration. In this watershed, low flow is significantly reduced when agricultural return flow is reduced after the forest restoration from winter wheat. The irrigation for winter wheat is applied during the dry season and sourced from deep groundwater that is treated as an infinite source in SWAT. This loss of irrigation return flow results in moderate reduction in water yield but strongly reduces

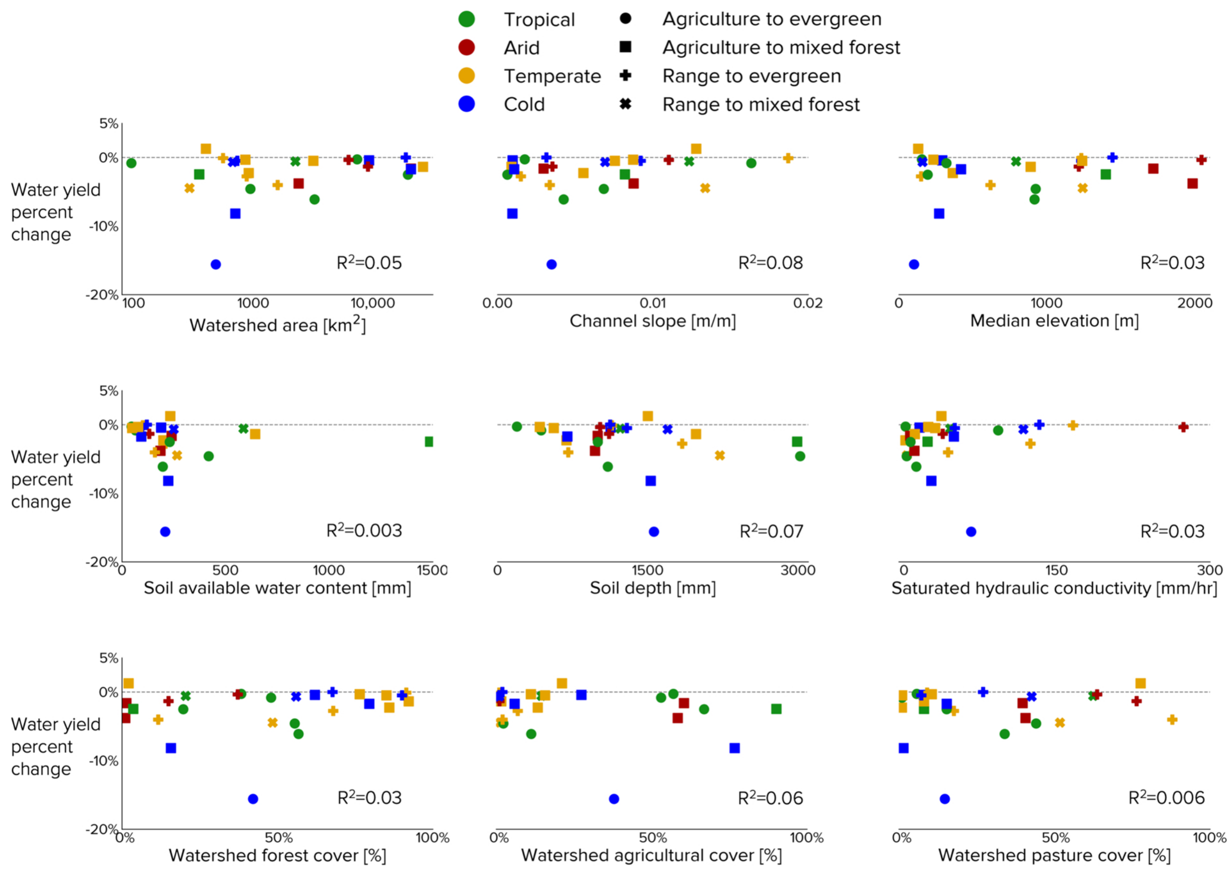

Fig. 4. No linear relationships are seen between the many explanatory variables considered and water yield, which is reduced in almost all cases after forest restoration. Water yield change from the restoration scenario is shown against variables based on watershed physiography (top row), soil (middle row), and pre-land-cover change land cover (bottom row).

low flow since irrigation is scheduled specifically during the low-precipitation, and hence low-flow, period.

The high flow measure typically is reduced when forest is restored. The Cachoeira River Basin in Brazil sees a 30% reduction in high flow at a relatively large channel slope of 1.3%, and with a particularly strong contrast in the infiltration capacity between the mixed forest and the transitioning pasture. These two factors together seem to drive this behavior. The Reda Catchment sees a 50% reduction in high flow after restoration because its small water yield is associated with a small peak flow, so a small reduction creates a large percent change.

3.2. *Statistical analysis*

3.2.1. *Simple linear regression and analysis of variance*

We find few statistically-significant linear relationships between 82 watershed characteristics tested and the changes in water yield, low flow, and peak flow across the sites. Figs. 4 and 5 show the response of water yield, low flow, and peak flow to forest restoration plotted against watershed characteristics. In none of these cases are there significant correlations between the flow response and watershed characteristic.

Fig. 4 demonstrates that there are no systematic effects on water yield change in the forest restoration scenario from any one of numerous watershed characteristics including physiography, soil, and original land cover. The R^2 is less than 0.1 for each watershed characteristic, and the relationship is not statistically significant at the 5% level for any characteristics. The examples in Fig. 4 are typical for other climate characteristics such as average precipitation, potential evapotranspiration, seasonality, maximum annual precipitation, or snowfall; watershed geometry parameters such as the watershed perimeter, elongation, or circularity; or soil characteristics such as field capacity, sand content, or clay content.

The effects of forest restoration on high flow and low flow are larger, but also lack clear relationships to most watershed characteristics. Fig. 5 shows the response of the low flow and peak flow indices to the same physiographic and soil variables shown in Fig. 5. The relationship is not statistically significant at the 5% level for any characteristic.

In only three cases are the relationships statistically significant accounting for the multiple hypothesis test, all with moderate values of $R^2 < 0.35$. The change in water yield increases with the fraction of precipitation evapotranspired ($R^2 = 0.33$) in the agricultural development scenario because sites with smaller water yields have larger percent changes with similar volume changes. The change in low flow decreases with the fraction of precipitation evapotranspired ($R^2 = 0.34$) for the forest restoration scenario as the forests transpire more water and reduce the amount available for dry-season flow. However, there is substantial scatter and sites see increased low flow along the whole range of fraction of precipitation evapotranspired. Finally, low flow change decreases with increasing soil porosity ($R^2 = 0.31$) for the agricultural development scenario. Less porous soils may be less able to store water for later evapotranspiration, leading to more sensitivity to land-cover change. Climate and soil both have important effects on the response to forest-cover change, but there are relatively few linear effects.

The one-way ANOVA does not show significant difference between the means of the water yield, low flow, or peak flow grouped by the broad Köppen-Geiger climate classification, Köppen-Geiger climate classification seasonality, or forest change group.

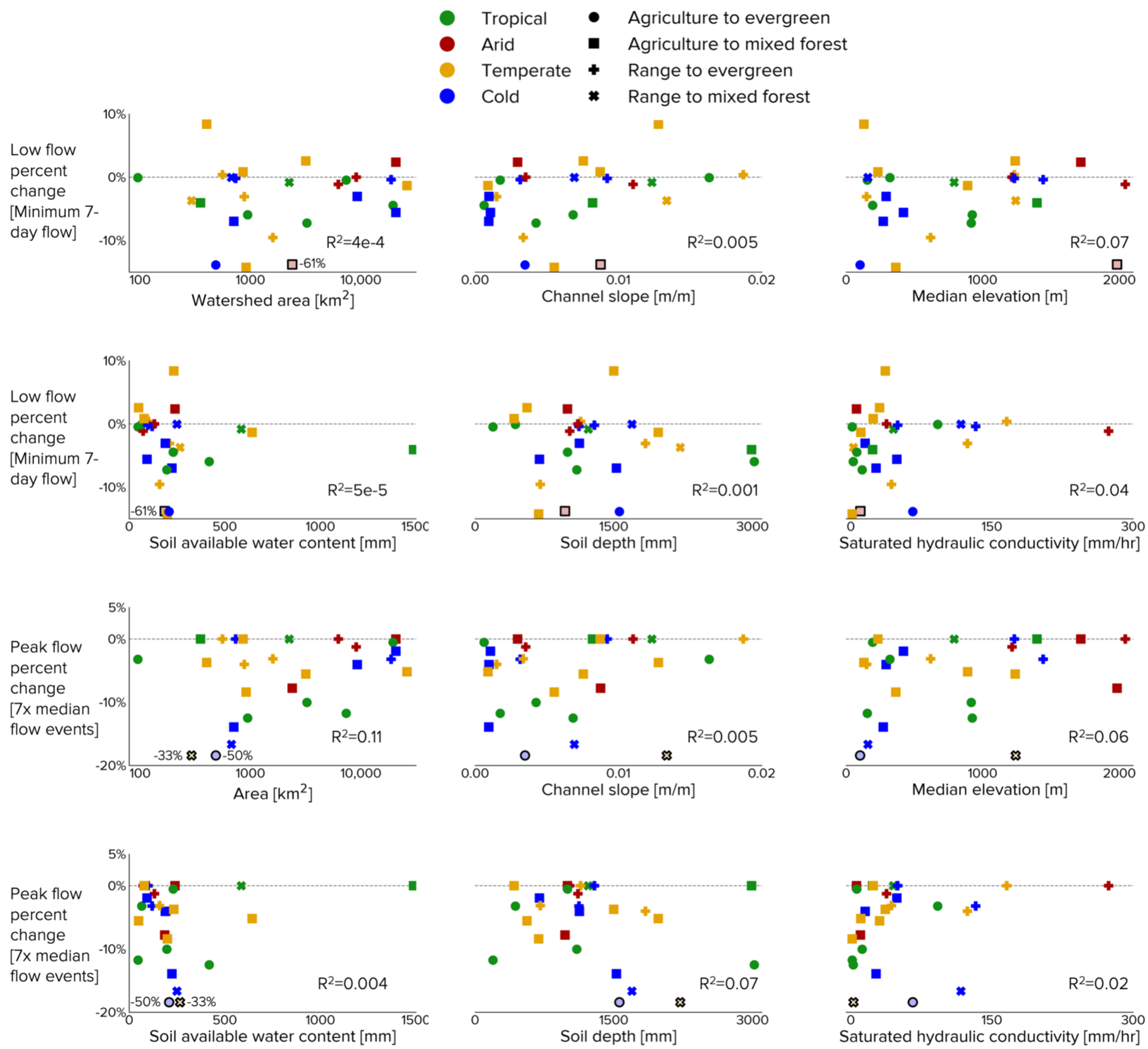

Fig. 5. No strong linear relationships are seen between explanatory variables and either a low flow measure (average minimum annual 7-day flow) or a high flow measure (number of annual flow events at 7 x median flow). Low flow and high flow measures changes from the restoration scenario are shown against several potential explanatory variables, including those based around watershed physiography (first and third row) and soil (second and fourth row). The R^2 for each relationship is shown on the plot. Changes beyond the plot bounds are noted as light-colored with dark outlines at the max/min of the figure, and the actual values are noted in the first column of each row next to the points. Though we did not find simple explanatory variables for the changes in these measures, the size of their effect is larger than the change in water yield under land-cover change.

3.2.2. Multiple linear regression

The LASSO regression with cross validation finds minimal predictive power in linear models for the change in water yield, low flow, or peak flow with watershed characteristic predictors in most cases. The outstanding exception is the model for water yield change after agricultural development, which has a resulting $R^2 = 0.69$ (Fig. 6).

The increase in water yield after agricultural development provides the one example with significant predictive power. The predictors selected for this model are: the soil porosity, percent silt, and percent organic carbon; the percent of precipitation evapotranspired before land-cover change; the probability of a dry day following a dry day; the percent of urban land use; the elevation range; and the compactness coefficient, defined as the ratio of the watershed perimeter to a circle with area equal to that of the watershed. Note that the Köppen-Geiger climate classification, seasonality, and forest change group indicator variables were not selected as predictors. Instead, several factors related to the soil provide significant predictive power, as does the percent of precipitation evapotranspired before land-cover change, which represents the integrated behavior of the watershed including the soil. The probability of a dry day following a dry day provides an important climatic indicator of short-timescale and seasonal rainfall variability, which can affect how much the water the soil can store, because either water is unavailable, or all soil storage is filled. This suggests that the percolation of water through the soil, and water storage in the soil, significantly control the response of water yield to agricultural development in these models. The percentage of urban land cover strongly affects annual water yield and is therefore selected as a predictor in the multiple regression. It remains unclear at this time why the compactness coefficient is predictive.

We suggest that the agricultural development scenario has strong predictive power for three reasons. First, the land-cover changes are relatively consistent compared to forest restoration cases that may convert from degraded lands or agriculture, depending on the site. Second, the agricultural land cover that expands is particularly consistent across sites. Third, the agricultural land may be particularly dominated by soil factors.

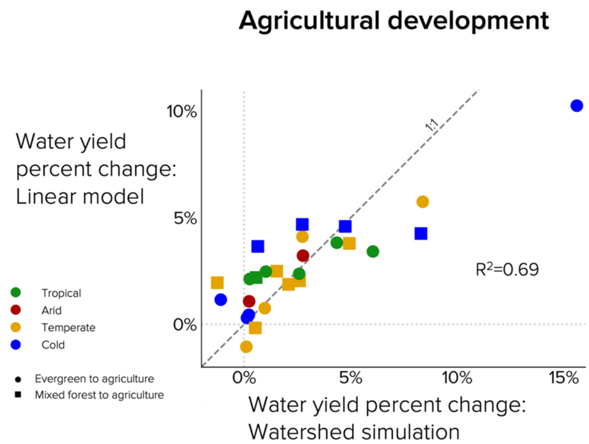

Fig. 6. The change in water yield after agricultural development is well-predicted by multiple linear regression; no other changes or scenarios have predictive power greater than the mean of the data. The estimates of water yield change obtained through LASSO regression with cross-validation are shown against the water yield changes obtained through watershed simulation. The colors and shapes correspond to the Köppen-Geiger climate classification and the broad land-cover change type. The 1:1 line is plotted to show the quality of the estimation.

In all other cases, the LASSO regression finds that though the training data has a positive R^2, the testing data instead results in a negative R^2. This negative R^2 means that the variance in the LASSO prediction relative to the watershed simulation is larger than the variance in the watershed simulation itself. There is no predictive power in the linear combination of the factors.

3.2.3. Permutation tests identify other important soil characteristics

Soil characteristics and hydraulic properties seem to have a non-linear relationship to the changes in water yield and low flow after forest restoration as seen in Figs. 4 and 5. The soil available water content, estimated here as the time-averaged soil water content in the SWAT site models, seems to be related to both water yield and low flow, with both large and small changes in water yield and low flow occurring with soil available water content from 150 mm to 500 mm, but a smaller response range when soil available water content is outside this range. For example, the Cedar Creek watershed in Indiana and the Po Ko catchment in Vietnam have soil available water content of ~224 mm and ~197 mm, respectively, and see relatively large water yield reductions of ~8.2% and ~6.1% when restoring forest.

We explore the validity of the relationship between soil available water content and hydrologic response after forest restoration with a two-sample permutation test (Good, 2005). The alternative hypothesis is that the 13 water yield values from sites that have soil available water content < 150 mm or > 500 mm have a smaller mean magnitude than sites that have 150 mm < soil available water content < 500 mm. The null hypothesis is that the means of the two groups are the same. The mean magnitude for sites with 150 mm < soil available water content < 500 mm is 4.2% while the mean magnitude for sites with soil available water content < 150 mm or > 500 mm is 0.8%, with an effect size of 1.21 using the Cohen's d measure (Basilevsky, 1994). This relationship has a p-value 0.00018, which suggests that the relationship is unlikely to occur by chance and thus is likely to be physically meaningful. The same test exploring the hypothesis for low flow (rather than water yield) has p = 0.00014 and an effect size of 0.84 (mean magnitudes of 10.2% and 1.4%, respectively), again beyond chance occurrence.

We also test the effect of saturated hydraulic conductivity, with the alternative hypothesis that low flow at 5 sites with > 100 mm/hr has a smaller mean magnitude than the other 22 sites. The group with saturated hydraulic conductivity < 100 mm/hr has a mean low flow magnitude change of 7.1% and that with soil hydraulic conductivity > 100 mm/hr a mean low flow magnitude change of 1.0%, and the difference between the mean magnitudes has a Cohen's d of 0.54 and p = 0.033. Only sites with lower hydraulic conductivity see a change in low flow after land-cover change, though some changes may be small. The effect of low hydraulic conductivity on the change in water yield is slightly stronger but less significant, with a Cohen's d of 0.68 (mean magnitudes of 2.9% and 0.8%) but a p-value of 0.07, outside the normal bounds for significance. The peak flow values did not show significant differences across these same watershed characteristic groups.

It makes sense that soil properties are important for the flow changes after forest cover change, as seen in both the multiple linear regression and non-linear permutation tests. Higher soil porosity and hydraulic conductivity leads to faster vertical transit through the soil. This decreases the time for interaction between water and vegetation and thus reduces sensitivity to forest cover change. Very low soil available water content, due to differences in soil depth and porosity, means that little is stored and may mean more surface runoff. This again reduces water interaction with the vegetation and thus the sensitivity to forest cover change. High soil available water content means that more water is stored and likely available for plant use, but with enough storage the vegetation effects may be small. Both extreme soil available water content values reduce sensitivity to land-cover change. The soil available water content is likely to be controlled by both soil porosity and soil silt percentage, which affect the physical soil structure.

4. Other considerations

This work investigates the effects of forest restoration and conservation around the world. Though we specifically collected site models stratified by climate zone, and they include 6 continents, this collection of sites still under-samples South America, Australia, and Africa. We also note that in some places removing alien tree species to restore riparian grasslands likely would have effects opposite those discussed here (Le Maitre et al., 2002; Preston et al., 2018; van Wilgen and Wannenburgh, 2016)

Model caveats: There are several structural limitations of the hydrologic model, SWAT, used in this work. First, SWAT, although widely used (Gassman et al., 2007), does not include full spatial connectivity, as noted in the Methods. Spatial connectivity might have important effects for forests in the riparian zone (Everson et al., 2007; Scott, 1999; Scott et al., 2004) or on peak flow (Jackson et al., 2008). Second, the vegetation model employed (Williams, 1995) is relatively simple, as we could not simulate ecosystem behavior on 29 sites as carefully as we might on a single site. We selected previously peer-reviewed work to provide a solid baseline vegetation behavior and tested the behavior against standard data sources to ensure that it is reasonable. Third, SWAT uses the curve number approach (Rallison and Miller, 1982), which was developed for use in US agricultural systems. The curve number approach has been criticized for its assumptions regarding infiltration-excess overland flow, but it has been applied globally and proven useful for watershed simulation in many cases. Fourth, SWAT does not include fog capture, with which forests may increase local water input in certain circumstances (Ellison et al., 2012) and our results may not apply in sites with significant fog capture. However, a only a small area of Earth's surface consists of fog-capturing forest, and the effects on watershed-scale hydrology are usually small (Bruijnzeel et al., 2011). None of our watersheds include large amounts of fog capture. Fifth, one site model in this study irrigates crops from an infinite source, which is unrealistic. This unrealistic assumption is unlikely to create short-term model errors because the deep groundwater source is assumed to be largely separated from the hydrology above. However, such groundwater pumping is not sustainable, and over time will deplete aquifers (Faunt, 2009; McGuire, 2017), allow saltwater intrusion (Werner et al., 2013), and cause land subsidence (Erban et al., 2014).

Representing land-cover change heterogeneity: We chose simple land-cover changes between forest and one other land cover, but these are simple and imperfect. Land-cover changes proportional to actual, pre-change watershed land covers resulted similar water yield changes to the single forest-cover change cases (Appendix A.7) and compared well to the single forest-cover change simulations.

Targeted land-cover change: Ecosystem service projects target land-cover changes to watershed subregions that provide the greatest positive impact. Our results are based on uniform forest-cover changes. One question is whether more complex spatial land-cover changes have the same impact on flow. We conducted sensitivity analyses that reflected restoration localized: 1) in the headwater subwatersheds, 2) in subbasins along the main stem, and 3) in a spatially contiguous subset of subwatersheds, and found miniscule differences with the uniform forest-cover change, so the latter was selected for simplicity (see Appendix A.8). Empirical evidence suggests that forest removal in the riparian zone increases water yield and low flow more in many cases (Scott, 1999; Scott et al., 2004). SWAT's lack of full spatial connectivity may make it less sensitive to this spatial structure. We view our results as a conservative estimate of the amount of water yield reduction after forest restoration. We note that watershed service projects often cannot focus activities solely in the relatively narrow riparian corridor.

Forest restoration and conservation has other benefits: We show here that watershed-scale streamflow benefits to forest restoration and conservation are small. However, forest restoration and conservation have other benefits. McDonald and Shemie (2014) find that up to 700 million people can have their water supply secured against contamination from nutrients or sediment through land conservation. Forests also provide significant benefits in cooling (Ellison et al., 2017), habitat (Mendenhall et al., 2016), pollination (Ricketts and Lonsdorf, 2013), and precipitation recycling (Keys et al., 2017, 2016). Although realistic forest change provides minimal streamflow benefits, the small streamflow change would potentially allow other values to drive investment in forest restoration and conservation.

5. Conclusions

We find that 10% forest-cover changes cause relatively little change in key water quantity watershed services based on simulated hydrologic responses of 29 watersheds distributed globally across climatic zones. Water yield changes due to forest changes, including both afforestation and deforestation representing the inverse of forest conservation, are typically limited, with a median magnitude of < 1.5% given a 10% land-cover change representing forest restoration, or agricultural or urban development. Median low flow and high flow decreased by 3 and 4% for forest restoration. For agricultural and urban development, median high flow increased 5–7% and low flow increased by ~1%. Water yield declined by < 4% after forest restoration in 75% of the watersheds, and increased by < 3.5% after agricultural development in 78% of the watersheds. Under urban development, 78% of the watersheds saw water yield increase by < 6%. These estimates are consistent with both a simple conceptual model developed to quickly test the water yield impacts of land-cover change (Appendix A.5) and paired-catchment experiments.

There are few linear relationships between the changes in the streamflow index values considered here and simple watershed characteristics, with the noted exception of the change in water yield under the agricultural development scenario. There were few strong correlations ($R^2 > 0.25$) for simple linear regressions of water yield, low flow, and peak flow versus catchment characteristics based on climate, physiography, soil properties, and land cover. Only three cases were significant under multiple hypothesis testing. These were strongly related to the percolation of water through the soil and plant access to such water for transpiration.

A multiple linear regression did not find linear models that predicted the changes in streamflow index values for any case except the change in water yield after agricultural development. In all other cases, the variance between the prediction and the watershed simulations was larger than the variance in the watershed simulations themselves. The change in water yield after agricultural development had a strong relationship ($R^2 = 0.69$) with several soil, climate, and physiography characteristics. Both water yield and low flow have larger magnitude change after forest restoration with soil available water content in the 150–500 cm range ($p \sim = 0.00015$ for both) and with soil hydraulic conductivity below 100 mm/hr ($p = 0.033$ and $p = 0.07$ for low flow and water yield, respectively). The importance of soil properties in both the multivariate and non-linear tests suggests that soil water percolation and uptake by vegetation is particularly important in the hydrologic response to forest restoration.

The small magnitudes of hydrologic change seen here are important given the desire to have increased water yield and base flow encourage investment in forest conservation, restoration, and management (Bennett and Carroll, 2014; Brauman, 2015; Bremer et al., 2016). This study suggests realistic forest-cover changes for many moderate-sized watershed service programs are unlikely to appreciably enhance streamflow. In many cases restoration or conservation of forest cover by < 10% will slightly reduce both water yield and dry-season flow. Watershed service programs may incur substantial reputational risk if they fall short of expectations of large enhancements in water quantity watershed services, likely except in certain cases such as removing high-water-use alien trees from riparian zones. Moderate-sized watershed service programs should perform realistic and careful assessment before promising enhanced water yield and dry season flow, or focus on other services.

Acknowledgements

This work was supported in part by the National Science Foundation under grant ICER/EAR-1829999 to Stanford University as part of the Belmont Forum – Sustainable Urbanisation Global Initiative (SUGI)/ Food-Water-Energy Nexus theme. Any opinions, findings, conclusions or recommendations expressed in this material are those of the authors and do not necessarily reflect the views of the National Science Foundation. The authors also thank Stanford University's Department of Earth System Science for financial support. The authors thank Paul Switzer, Professor Emeritus at Stanford University, for his significant assistance with designing the statistical approach. The authors thank Gretchen Daily for her helpful comment and insights into ecosystem services. The authors acknowledge all those who shared their existing site models, which made work of this breadth possible. We especially thank Darren Ficklin, who shared several site models in the western US, Karim Abbaspour who assisted us in collecting several site models from Iran, and Aijin Zhang who shared two watersheds in China. We acknowledge everyone else who was willing to share their site models: Hau Zhang, Khoi Dao, Dulce Rodrigues, Donizete Pereira, Samira Akhavan, Hiro Somura, Danilo Fukunaga, Saeed Vaghefi, A. Pouyan Nejadhashemi, Bikesh Shrestha, Tram Lo, Emmanuel Obuobie, Pawel Marcinkowski, Mikolaj Piniewski, Johanna Richards, Matjaz Glavan, Milad Jajarmizadeh, Rewati Niraula, Sarah Praskievicz, Rui Jiang, Diana Pascual, Katrin Bieger, Partha Saha, and Deepthi Rajsekhar, along with all their collaborators.

Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at doi:https://doi.org/10.1016/j.gloenvcha.2019.101938.

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