Comparing Chinese and American experiences of restoring degraded cropland: What can be learned for governing payments for ecosystem services?

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Abstract: Payments for ecosystem services (PES) have attracted broad attention as a novel approach for using economic incentives to provide ecosystem services more sustainably. However, there have been inadequate efforts addressing the basic question of how to design and execute PES at the program level. By comparing and contrasting the experiences of restoring degraded cropland to forest and grass covers in China and the U.S., this paper aims to tackle that question and provide some valuable and timely policy insights that can be used by China and other countries in improving the performance of their PES programs in terms of effectiveness, efficiency, and/or equity. Our analysis will unfold through examining such specific question as: What are the socioeconomic and environmental backgrounds for one country to launch a large PES program? How was it designed initially and has evolved over time? How has its performance been evaluated and what are the main outcomes? How likely is it for the enrolled land to be reconverted or for the contract to be expired? What are the primary challenges to its long-term success? Finally, this study calls for a more practical and focused approach to PES design, implementation, and evaluation that will lead to improved outcomes of forest and grassland ecosystem restoration and biodiversity conservation.

Keywords: Payments for ecosystem services; ecological restoration; conservation reserve; forest cover; policy design and implementation; program effectiveness and efficiency

1. Introduction

Despite the consensus that payments for ecosystem services (PES) represent a novel, incentive-based approach to providing ecosystem services more sustainably [1,2], there remains a dearth of concrete and practical analyses of how to govern them properly [3-5]. Further, while project-level issues like conditionality and additionality have received wide attention [6,7], it seems more relevant and appropriate to consider PES governance at the program level [8,9]. A program, made up of multiple, specific projects with clearly defined targets and means and mechanisms to achieve them, tends to be more complicated in content, larger in space, longer in time, greater in investment, and thus more closely linked to external socioecological settings [9,10]. This study is thus motivated to address how to design and implement large PES programs by comparing the experiences of China and the United States in restoring degraded cropland to forest and grass covers.

Of course, asserting a lack of concrete and practical analyses of PES governance at the program level does not imply that only a few attempts have been made. In fact, an explosive body of literature, including a set of journal special sections or features, has been devoted to advancing alternative perspectives of and approaches to PES, in addition to addressing issues related to the linkages between conservation and development, the valuation of ecosystem services, and the performance of PES.[1] For instance, PES were once viewed as "voluntary, conditional transactions between at least one buyer and one seller for well-defined environmental services or corresponding land use proxies" [6]. Now, it is generally known that carrying out PES, especially those large programs, entails complex, long-term interactions of various components of the underlying social-ecological systems and lead to multiple, often mixed, and uncertain outcomes [16-18]. Therefore, commodification of ecosystem services may not be realistic in many situations and market alone may not be adequate under certain circumstances. Other institutions and mechanisms, such as hierarchy and local collective action, are also necessary [19-21] and different types of institutions can be used together to complement one another [22-24].

Progress on how to govern PES by past efforts notwithstanding, there is a long way to go before we have gained a clear understanding of the multi-faceted, intricate nature of PES and thus a capability of prescribing more appropriate, better suited mechanisms and measures for their execution [3-5, 25-26]. Meanwhile, this kind of knowledge and capability is urgently needed for the wide-ranging adoption and long-term success of PES in enhancing ES and human wellbeing [4,14,27]. This situation is even more urgent in view of the fact that the limited empirical insights on this subject have come mainly from small private-sector experiments and small country cases in Latin America.[2] In the words of Gregersen and others [28], "most of the available literature does not get into the subject of governance improvement in depth, particularly not at the country level."

As the biggest PES program in the developing world, China's Sloping Land Conversion Program (SLCP) has been in place for well over a decade—sufficiently long for its impacts, as well as its challenges, to be manifest and identified [29,30]. This paper will analyze the crucial aspects of governing the complex processes of interactions and outcomes involved in the SLCP by comparing it to the U.S. Conservation Reserve Program (CRP), which is the largest PES program in the developed world [6,31]. Emerging from this nuanced analysis will be important lessons regarding how to govern the SLCP and other PES programs more effectively, efficiently, and equitably.

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[1] Some early examples of the special sections or issues are: Ecosystem services: From theory to implementation edited by Daily and Matson [11]; Payments for ecosystem services and poverty reduction: Concepts, issues, and empirical perspectives edited by Bulte et al. [12]; Payments for environmental services in developing and developed countries edited by Wunder et al. [13]; Payments for ecosystem services: Reconciling theory and practice edited by Pascual et al. [14]; and Payments for ecosystem services: From local to global edited by Farley and Costanza [15].

[2] Examples of the former contain agreements negotiated by the water-bottling firm Vittel with French farmers in the catchment feeding its spring source, while those of the latter include the purchase contacts of avoided deforestation and forest degradation for hydrological services by the governments of Costa Rica and Mexico [2,5-7].

Certainly, China's expanding portfolio of ecological restoration efforts can benefit from this kind of work [32]. Similarly, as more PES programs are launched worldwide [33,34], this study can shed light on how to improve PES design and implementation in many other countries. Indeed, a priority outcome of the 2012 United Nations Rio+20 Conference on Sustainable Development was the target to restore 150 million ha of degraded land globally by 2020 [35]. To accomplish this huge task and to make headways in carrying out REDD+ (reducing emissions from avoided deforestation and forest degradation and enhancing carbon stock through reforestation and forest management) [36,37], however, it is imperative for the international community to assess and synthesize the current PES experience and evidence around the world, including the American CRP and the Chinese SLCP.

The paper is organized as follows. In the next section, we outline our research methodology—a comparative study based on information and data that have been accumulated, and concepts and principles that have been articulated in the governance literature. In sections 3 and 4, we highlight the U.S. CRP and the Chinese SLCP experiences and effects, respectively. In section 5, we examine the major challenges that the SLCP has faced. Finally, some closing remarks are made in section 6.

2. Methodology

Here, we adopt qualitative and quantitative analyses in comparing and contrasting the experiences of restoring degraded cropland in the U.S. and China. That is, we will look into the various facets of program design, implementation, and evaluation by drawing information from Stubbs [38] for the American experience and from Yin et al. [29] for the Chinese experience. Our analysis will unfold around a set of similar, if not common, questions. These questions include but are not limited to: What are the socioeconomic and environmental backgrounds for one country to launch such a PES program? How was the program designed initially and how has it evolved over time? How has it performed in terms of efficiency, effectiveness, and/or equity? How likely is it for the enrolled land to be reconverted or for the contract to be expired? What are the main challenges to its long-term success?

Moreover, we will examine the SLCP performance and challenges by taking advantage of a large panel dataset available to the authors. The dataset, covering over 1,000 households for the period of 1999-2008, was built from repetitive surveys in six counties—Nanbu, Nanjiang, Mabian, and Muchuan in Sichuan, and Zhen'an and Yanchang in Shaanxi based on a stratified random sampling strategy [29]. These six counties in the two provinces, selected according to the geographic coverage of the program, their general regional conditions, and the distribution of farmers' income, among other factors, represent some primary sites of the SLCP piloting and implementation. The dataset contains information on land enrollment status and land-use dynamics, subsidy payment, family demographics, on- and off-farm production and employment activities as well as expenses and revenues for both participants and non-participants. The nominal price, cost, and revenue information has been converted to real values using the provincial Consumer Price Indices, with 1994 as the base year.

3. American experience

Land retirement has been a mainstay of U.S. agri-environmental policy. According to Claassen et al. [39], ever since the 1930s, the U.S. has relied primarily on voluntary payment programs to encourage soil conservation and other improvements in agri-environmental performance, although cross-compliance and regulation have also been used. The CRP is the largest federal, private-land retirement program in the U.S., which provides financial compensation for an extended period (typically 10-15 years) for the benefit of soil and water quality improvement and wildlife habitat. The program, first authorized in the Food Security Act of 1985, is administered by the Farm Service Agency (FSA) of the U.S. Department of Agriculture (USDA), with technical support from the Natural Resources Conservation Service and other USDA agencies.

How does the CRP work? The CRP is a competitive program, in which landowners offer eligible land for enrollment into it. There are two types of enrollment into the CRP: general and continuous sign-up. The former is a specific period of time during which the federal agency accepts offers from farmers, which are ranked according to an Environmental Benefits Index (EBI) to determine the relative environmental benefits for the land offered. For each general sign-up, the FSA collects data on each of the EBI factors and ranks all eligible offers across the country. After the sign-up ends, the USDA determines an EBI threshold. Acceptance for enrollment into the CRP is extended to offers that scored above the EBI threshold, which varies by sign-up depending on the offers received. As of July 2014, 19.7 million acres (7.97 million ha, or 77% of total the CRP areas) were enrolled under general sign-up contracts.

Continuous sign-up is designed to enroll the most environmentally desirable land into the CRP through specific conservation practices or resource needs. Unlike the general sign-up process, land offered under continuous sign-up may be enrolled at any time and is not subject to the competitive bidding. If offers meet certain eligibility requirements, then they are accepted automatically. Continuous sign-up covers initiatives that either target acres with specific resource concerns or support additional conservation practices. As of July 2014, 5.75 million acres were enrolled in the CRP continuous sign-up, with 1.6 million acres covered through the two largest initiatives—the Conservation Reserve Enhancement Program and Farmable Wetland Program.

Producers have a number of conservation practices to consider for use on their land when enrolling in the CRP. The selection of practices is part of the voluntary enrollment process and is determined by the landowner, with assistance from the USDA. Once an offer is accepted for enrollment, the participant must develop a conservation plan of operation that serves as a guide for which practices will be used, where, and for how long. Once the plan is approved and the contract signed by the participant, the land is considered enrolled. Certain continuous sign-up initiatives require specific conservation practices for enrollment. Under current law, a farmer or rancher wishing to terminate a CRP contract early faces a penalty of full repayment, with interest, of all the funds already paid to the producer, plus a fee of 25% of the rental payments received. Even though the Secretary of the USDA has the authority to release land from the CRP without penalty, this option has not been commonly used. Regardless, however, environmentally sensitive acres were not released and certain restrictions applied to acres returning to production or harvesting and grazing.

How is the EBI formulated? Following the 1990 farm bill, the CRP was required to consider the environmental benefits of the land offered for enrollment. The EBI is a standardized way to compare different land types with varied resource needs across the country [38]. It is designed to compare the benefits that the offered land can provide. Presently, the FSA collects data for each of the following EBI factors for the land offered, which are weighted and scored based on its potential to generate the desired environmental benefits. Some factors are made up of sub-factors (listed in parentheses).

The formulation of the EBI has changed over time, including becoming more transparent to participants. The most recent general sign-up included the following factors and weights: (1) Wildlife Factor (10-100 points) evaluates the expected wildlife benefits of the offer (wildlife habitat cover benefits, wildlife enhancement, and wildlife priority zones); (2) Water Quality Benefits Factor (0-100 points) evaluates the potential impact that the offer may have on both ground and surface water quality (location, ground water quality, and surface water quality); (3) Erosion Factor (0-100 points) evaluates the potential for the land to erode from wind or water and is measured using an erodibility index; (4) Enduring Benefits Factor (0-50 points) evaluates the likelihood for certain practices to remain in place beyond the CRP contract period (weighted average for all practices); (5) Air Quality Benefits Factor (3-45 points) evaluates the air quality improvements made by reduced particulate matter and increased carbon sequestration (wind erosion impacts, wind erosion soils list, air quality zones, and carbon sequestered); and (6) Cost (0-25 points) indicates environmental benefits per dollar spent (cost of the offer and how much the offer is below the maximum payment rate). In 2013, the 45th general sign-up recorded 1.57 million acres of the 1.68 million acres deemed acceptable, with an EBI score of 209 and above.

How has the enrollment progressed? The authorizing statute establishes the maximum number of acres that can be enrolled in the program at any one time. This maximum was 27.5 million acres for fiscal year (FY) 2014 and 26 million for FY 2015; and it will be 25 million for FY 2016 and 24 million for FYs 2017 and 2018. The program is authorized to spend as much as necessary to enroll up to the maximum level of allowable acres, with mandatory funding that is provided through the borrowing authority of the USDA’s Commodity Credit Corporation and is not subject to annual appropriations. Currently, the average annual federal cost for the CRP is close to $2 billion ($1 US ≈ 6.6 Chinese yuan as of August 1, 2016). The majority of this cost is annual rental payments, which average $63.65 per acre, or $157.28 per ha, but it can vary greatly by location. For instance, the highest average rental rate for a state (for all sign-ups)—$207.20/acre— occurred in Massachusetts in 2014 (only 10 acres were enrolled). The lowest was in Wyoming at $26.57/acre. Texas had the most acres enrolled—at 3.2 million acres, with an average rental rate of $36.78/acre.

CRP contracts vary in length, although most are 10 years in duration. At the end of a contract, the participant may either seek reenrollment in the program or let the contract expire. Notably, this 10-year cycle resulted in more than 16 million acres enrolled in

1997, potentially expiring all at once in 2007. To stagger this expiration process, the USDA offered 2-5 year reenrollment and extension contracts in 2006 to contracts expiring between 2007 and 2010 (27 million acres altogether) [38]. During that time, roughly 83% of the contracts whose holders were offered these extensions accepted them (accounting for 23 million acres), and contracts involving over 8.5 million acres expired. Between FY 2007 and FY 2014, over 17.1 million acres under contract expired and were not reenrolled.

While the CRP enrollment has fluctuated since its creation, recent enrollment has declined from its peak in FY 2007 of 36.8 million acres to 25.6 million acres in FY 2013. Reduced enrollment is thought to be a product of high commodity prices, low rental rates, and declining interest in retiring land from production [38]. Further reduction in the farm bill was viewed as inevitable given the fiscal challenges. Several programmatic changes have been centered around permitted activities. Emergency harvesting, grazing, other uses of forage, as well as livestock grazing for a beginning farmer or rancher, are sometimes permitted, without a reduction in rental rate. Other approved activities, such as annual or routine grazing, may continue to require a reduction in rental rate.

How about the environmental benefits? Since its inception, the CRP has contributed to soil erosion reduction, water quality improvement, and wildlife habitat development.[3] According to the FSA, since 2002, the CRP has reduced soil erosion by 325 million tons from pre-CRP levels each year. Other conservation benefits include a net CO_2 sequestration of about 52 million tons; a reduction in fuel use and avoidance of nitrous oxide emissions due to non-use of fertilizer; establishment of more than 2 million acres of wildlife habitat; and a reduction of 607 million pounds of nitrogen and 122 million pounds of phosphorus. From a wildlife perspective, it is estimated that the habit availability provided by CRP land sustains over 13.5 million pheasants and 2.2 million ducks each year [38].

4. Chinese experience

Following a piloting phase of only two years, the Chinese government launched the SLCP in 2001, under which farmers are subsidized at high, uniform rates for restoring degraded cropland [29]. The SLCP is one of several large ecological restoration programs (ERPs) that the Chinese government initiated in the late 1990s in response to a series of environmental disasters, including flooding in the Yantze River basin in the southwest and the Songhua River basin in the northeast, and soil erosion and land sliding across the west [9]. Also, most of those regions of heavily degraded ecosystems in China happened to have a higher concentration of poverty incidence and slower economic growth [40]. The ERPs have thus been aimed at both environmental improvement and poverty alleviation [9].

The original SLCP policy stipulated that farmers would receive a grain subsidy of 2.55 tons/year/ha in the Yangtze River basin and 1.50 tons/year/ha in the Yellow River basin, in addition to an annual cash outlay of 300 yuan/ha for purchasing seeds/seedlings

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[3] Notably, reducing poverty or income inequality has not been a concern to the U.S. CRP.

and tending activities [31]. The durations of subsidy would vary depending on the type of restoration: eight years if ecologically benign trees (species mostly providing ecological functions and services) are planted; five years if commercial trees (species mainly providing timber, fruits, and other products) are established; and two years if grassland is rehabilitated [29].

How has the program evolved? The program has undergone significant modifications. First, in most places farmers opted for the eight-year scheme in order to take advantage of the generous subsidy, regardless of the suitability of the chosen practice(s). Second, due to dwindling public grain reserves, the food subsidy was quickly replaced with a monetary compensation in 2004, with a price of 1.40 yuan/kg [41]. Further, it was scaled back considerably because of concerns about food security induced by the significantly reduced amount of cropland and the mushrooming financial burdens placed upon governments [31]. By the end of 2013, about 8.0 million ha of degraded cropland had been retired, whereas the original target was set at 14.5 million ha by 2010 [42]. The cumulative investment reached 243.1 billion yuan by the end of 2013 [43]. Finally, the program has been extended for another eight years to further reduce poverty and generate alternative jobs and thus earnings, while compensation for farmers' lost grain yields has been halved [44].

Table 1. Land-use dynamics largely induced by retiring degraded cropland
(unit: mu/household)

	Shaanxi			Sichuan		
	Enrollment	Farmland	Forestland	Enrollment	Farmland	Forestland
1999	4.57	10.19	9.75	3.91	3.37	9.45
2003	11.68	7.17	18.04	4.62	2.83	12.02
2008	14.87	4.63	27.22	5.01	2.91	14.38

Note: Data came from the authors' surveys. One mu (a Chinese measure of land area) is 1/15 of a hectare. Paddy fields are excluded from farmland because they are not considered for setting aside; forestland is land designated to growing trees, some of which may not have reached adequate canopy cover yet.

At the local level, our data suggest tremendous land-use changes, mainly driven by implementing the program. Table 1 shows that in Shaanxi, program enrollment started at a rate of 4.6 mu per family, and continued its climb to almost 15 mu in 2008. In Sichuan, enrollment began at a rate of 3.9 mu per family and gradually reach 5.0 mu in 2008. Meanwhile, cropland per household in Shaanxi experienced a dramatic decline—from 10.2 mu in 1999 to only 4.6 mu in 2008; forestland increased even more—from 9.8 mu to 27.2 mu. In Sichuan, the cropland reduction from 3.4 mu to 2.9 mu was small, but the forestland increase from 9.5 mu to 14.4 mu was substantial during the period. The varied rates of participation have to do with the availability of total and degraded cropland in a province; and the smaller contraction of cropland than the amount of land enrolled in the program is due to the inclusion of non-permanent farming plots or the gain in forestland due to tree planting elsewhere [29].

How are the subsidies compared to opportunity costs? Table 2 indicates that the average net revenue from farming at the beginning of the program were way below the government subsidies. Here, we approximate participants' net revenue from grain production with the average value of all the surveyed households in each sample county.

It can be seen that even in the sample county with the highest net farming revenue in a province, the government nominal subsidies during the initial round—3450 yuan/ha/year in Sichuan and 2400 yuan/ha/year in Shaanxi—were very generous and, indeed, overcompensation occurred in most of the years and counties. Note that our approximation could be biased upwards because the retired plots of cropland tend to be the marginal ones with low yields.

Of course, the overcompensation in China could be viewed as a measure of poverty reduction, as articulated in the initial program document [45]. If so, the poverty reduction benefit for participants ranges from 1230 to 1500 yuan/ha in Shaanxi and 1770 to 2040 yuan/ha in Sichuan during the first three years, which is much higher than the lost revenues from grain production. Given the apparent overcompensation and the financial need for retiring more marginal lands and retaining those retired, the program efficiency could be much improved. Unfortunately, the government failed to evaluate the opportunity costs of retiring degraded cropland in the designing and piloting phases of the program, let alone to introduce a market-based mechanism for farmer to offer their bids for cropland restoration contracts.

Table 2. A comparison of net farming revenue to government set-aside subsidy

(unit: yuan/ha)

Year	Shaanxi			Sichuan		
	Yanchuan	Zhen'an	Subsidy	Nanbu	Muchuan	Subsidy
1999	69.1	30.0	160.0	103.3	14.9	230.0
2000	77.8	33.1	153.8	111.7	19.1	221.2
2001	73.2	33.9	147.9	130	21.6	212.6
2002	77.4	38.8	142.2	124.9	27.5	204.5
2003	63.0	45.4	136.8	143.9	41.3	196.6
2004	72.8	58.0	131.5	184.9	54.0	189.0
2005	128.1	103.2	126.5	128.7	51.5	181.8
2006	146.7	122.8	121.6	151.1	47.2	174.8
2007	159.7	68.5	116.9	133.3	57.5	168.1
2008	84.0	39.9	112.4	79.5	33.8	161.6

Note: Data came from the authors' surveys. The opportunity cost of cropland is its net revenue from grain production before being enrolled in the program, approximated by the local average of farming net revenue deflated with the CPI (1994 as the base). Because the government's subsidies were fixed without adjusting for inflation, we use an annual discount rate of 4% in comparing them with farming revenues in the later years. The figures in red indicate amounts larger than the discounted government subsidies.

The uniform subsidies in large river basins are also questionable on the grounds of fairness because they have led to places with lower grain yields (and thus lower opportunity costs of retired cropland) benefiting more while others with higher yields benefitted less or even suffered a net loss. This situation is further reinforced by the different amounts of retired cropland. Although households in Sichuan received 1050 yuan/year more subsidy per ha in the first round, they did not benefit as much as their counterparts in Shaanxi because farmers there enrolled a greater amount of degraded cropland.

How about the induced income inequality? As shown in Table 3, the average household total income in Shaanxi increased from 3,791 yuan in 1999 to 10,040 yuan in 2008. In Sichuan, it rose from 5,031 yuan to 12,902 yuan. These remarkable income growths reduced poverty and improved livelihoods substantially. Notably, given China's official poverty line of an annual per-capita income of 637 yuan in 1994, which lasted until 2007 when it was raised to 785 yuan [29], the rate of households in poverty reduced from close to 30% to less than 5% within the decade.

Structurally, compared to a modest gain of less than 460 yuan in agricultural income in Shaanxi, the off-farm income rose from 1,049 yuan in 1999 to 4,775 yuan in 2008. Likewise, agricultural income in Sichuan increased from 3,241 yuan in 1999 to 5,745 yuan in 2008. During the same time period, household off-farm income jumped from 1,701 yuan to 6,174 yuan and the share of agricultural income declined from 64.4% to 44.5% in Sichuan and from 64.0% to only 28.7% in Shaanxi. These changes indicate that income inequality has replaced absolute poverty as a primary cause of concern.

Table 3. The composition and structural change of house income over time
(unit: yuan in 1994 price)

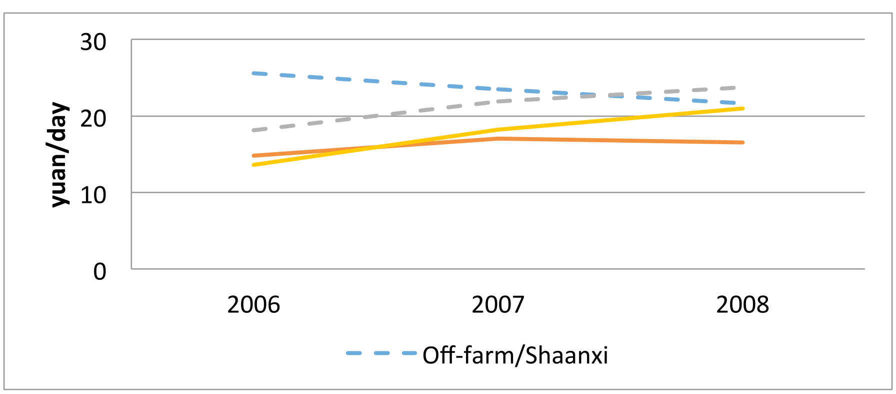
Year		Quantity					Percentage	
		Total	On-farm	Off-farm	Subsidy	Other	On-farm	Off-farm
Sichuan	1999	5031	3241	1701	89	0	64.4	33.8
	2000	5533	3324	1951	258	0	60.1	35.3
	2001	5917	3413	2223	281	0	57.7	37.6
	2002	6538	3598	2540	400	0	55.0	38.9
	2003	7138	3780	2825	534	0	53.0	39.6
	2004	7710	4059	3069	583	0	52.6	39.8
	2005	7689	3520	3096	779	294	45.8	40.3
	2006	8606	3955	3518	825	308	46.0	40.9
	2007	11495	5201	5391	577	326	45.2	46.9
	2008	12902	5745	6174	563	419	44.5	47.9
Shaanxi	1999	3791	2425	1049	318	0	64.0	27.7
	2000	4301	2518	1254	529	0	58.5	29.2
	2001	4518	2625	1379	514	0	58.1	30.5
	2002	5075	2735	1544	797	0	53.9	30.4
	2003	5331	2546	1669	1117	0	47.7	31.3
	2004	5965	2724	1809	1432	0	45.7	30.3
	2005	7404	2537	2770	1793	304	34.3	37.4
	2006	8277	2965	3139	1878	295	35.8	37.9
	2007	8781	3125	3802	1466	388	35.6	43.3
	2008	10040	2882	4775	1774	608	28.7	47.6

Note: Data came from the authors' surveys. "Subsidy" primarily comes from cropland retirement and farming; "Other" means local welfare compensation and assistance to the poor and disabled.

How about the likelihood of reconversion? Some researchers and policymakers have been concerned about the prospect of reconverting retired cropland back to farming once the restoration contract expires. For instance, Cao [47] reported that of all the participants, 37.2% in Shaanxi, 34% in Guizhou, and 29% in Ningxia indicated their intention to return to their retired plots to farming as soon as the subsidies are terminated. One way to discern this likelihood is to compare the wage rates for agricultural production and off-farm work. As shown in Figure 1, the wage rates of participants from off-farm work are universally higher than those from farming for the last three years of our data coverage (2006-08). Related to this are the trends of reduced work time in farming but increased labor days in off-farm activities. Family land-based labor time in Shaanxi reduced from 227 days in 1999 to 175 days in 2008, whereas off-farm labor time grew from 66 to 238 days. In Sichuan, family labor time in agriculture decreased from 321 to 232 days, while off-farm labor time grew from 133 to 246 days during the period.

If rural laborers can make higher earnings from off-farm activities than from on-farm ones and if the trend of rural labor transfer continues, it seems less likely that rational farmers will return to farming. So, the likelihood of reconverting the set-aside cropland back to farming is small. In fact, the continued urbanization and industrialization indicate that even without subsidies, some of the degraded cropland would have been abandoned, allowing vegetation to recover naturally. This underscores our point that the scale, level, and duration of subsidized cropland retirement should be determined in the context of overall economic growth [48]. However, certain households have not been much engaged in off-farm activities and thus continue to depend on farming for their livelihoods. Even if their income is above the official poverty line, many of them remain in a precarious status and do not enjoy a decent life. So, it is more likely that these less fortunate and relatively poor families will reconvert some of the set-aside cropland back to farming [31,41]. Efforts must be made to target them to avoid possible reconversion.

Figure 1. Estimated wage rates of different jobs for participants
(unit: yuan/day in 1994)



Note: Data came from the authors' surveys. The wage rates of labor in on- and off-farm employment are computed as the quotients of net revenues divided by the corresponding work times.

5. Challenges facing the SLCP

Quite a few interesting observations can be derived from the last two sections. Overall, it is clear that the U.S. CRP features voluntary and competitive participation through bidding for conservation contracts, which is predicated on the ranking of the land proposed for set-aside in terms of its EBI score. As such, the levels of compensation vary widely and the program's flexibility is reflected in such dimensions as contract durations, conservation practices, and sign-up types. On the other hand, enrollment into the Chinese SLCP is at best quasi-voluntary and no bidding is envisioned under the uniform standards of subsidies and durations of participation across large river basins, which are determined by the top-level of the government. As a result, the U.S. CRP is much more efficient than the Chinese SLCP, as reflected in the average annual rental payment of $63.65/acre (or $157.28/ha), which is equivalent to 975.15 yuan/ha and substantially lower than China's payment levels (given an exchange rate of $1 US = 6.6 yuan).

Therefore, even though it is admirable to launch such an ambitious PES program as the SLCP, it was not well designed and the Chinese government and society were not adequately prepared to carry it out effectively [31,41]. Specifically, it remains poorly understood how to properly incentivize the local people to pursue the basic tasks of land retirement, for how long and in what way(s) the subsidies should be provided, whether and how restoration practices should be differentiated, and how to combine incentive-based instruments with the regulatory and administrative means to carry out the targeted activities [9,40]. Below, we further elaborate on some of the issues.

How well has the program performed? Under the SLCP, the tasks and corresponding funds of land retirement and restoration are allocated by the central government; the selection of eligible cropland for retirement is simply approximated by the degrees of cropland slope and degradation [42,44], rather than based on a more comprehensive index like the U.S. EBI [49]. As the agency responsible for execution, the SFA has rarely coordinated its operations effectively with other agencies or regional and local authorities [9]. At the same time, these operations feature limited flexibility for individuals and communities to formulate and adopt innovative restoration practices and alternative ways of participation, even though they possess intimate knowledge about the local conditions as well as their needs [40]. Coupled with the fact that individuals do not take ownership of the program in most cases or may not even be allowed to plant trees of their chosen species, this has led them to commit little of their own resources to the program and taken very limited local collective actions [48]. Moreover, the government has not made adequate adjustments and adapted to the changing situations on the ground yet [49].

Given the critical importance of an EBI in determining land eligibility and selection for enrollment, it seems that developing such an index in China is a matter of when and how, instead of whether. Clearly, an EBI should reflect local conservation concerns and needs in China, and piloting the formulation of such an index should be first carefully pursued in different areas before its adoption across the country [5,40]. The U.S. index contains six factors—wildlife, water quality, erosion, endurance, air quality, and cost—with the first three being weighted more heavily. In comparison, a Chinese index may include no more than four factors for ease of broad adoption: erosion and flooding control, biodiversity conservation, and opportunity cost. To develop such an index as well as to better assess the program performance, the authorities ought to provide funding for relevant research projects and elicit experts' input into the policy and project design, implementation, and evaluation [5,29].

How to structure the restoration contracts? Concrete and binding project contracts should be pursued for the sake of improved program performance in China as well, and these contracts can be similarly allocated via some kind of open, competitive bidding. There are a few pitfalls to be avoided, though. First, it is impossible to execute a bidding process at the household level given the tiny amounts of their retired cropland—one family had only 1/3 ha in Sichuan and one ha in Shaanxi in 2008, as shown in Table 1. Our further estimation, based on a 2011 survey of 182 households in Wuqi—a county that has retired the largest amount of degraded cropland in Shaanxi, reveals that even there the average enrollment was only 2.2 ha per household in 3.8 plots [50]. Indeed, the maximum number of individual plots a household had in the program was as large as 13. In contrast, the contract size in the U.S. averaged 30.4 ha and the mean sign-up amount was 44.8 ha per farm as of July 2014 [38]. As such, it is more appropriate to construe ES provision and bidding contracts at the farm level in the U.S. [39]. In China, on the other hand, some local- or landscape-level projects must be structured and bidding by community groups or intermediary aggregators should be sought [29,48].

Also, the duration (how long the contracts may be) and the schedule and kinds of payment (flat rate or gradual phasing in (or out), cash or in-kind, or both) should be properly devised and practiced across the country. In the U.S., while most of the reserve contracts are initially set to last for 10 years, a majority of them can be and have been extended; at the same time, contract expiration is accepted as a normal option available to farmers and alternative channels exist to ensure the program coverage of the most environmentally sensitive lands. Further, compared to the U.S. experience, essential safeguards and technical support in China must be instituted whenever possible in carrying out and monitoring and enforcing the contractual activities [49]. Compared to the strong, costly external monitoring, internal monitoring in China is infrequent and inconsequential because of the lack of competent and/or responsible grassroots organizations [51]. Also, effective measures have rarely been taken to hold those participants who default their contractual obligations accountable; sanctioning for no compliance is simply to stop the compensation [49]. In contrast, the U.S. CRP has clear rules of sanction for early contractual termination or other defaults and responsible federal agencies have been involved extensively in gathering data for building and adjusting the EBI and sharing that information with local farmers, ranchers, and other stakeholders.

How about decoupling conservation goals with poverty reduction? Advocated by Kinzig et al. [26], this idea intends not to compromise the pursuant of efficiency with the concern with equity. It turns out that decoupling is hardly attainable in the Chinese context. First, a lot of the ecologically fragile and sensitive areas also have high levels of poverty [29,52]. While their livelihoods rely on ES, these people do not possess the basic internal assets and capabilities to alleviate their pressures on the ecosystems, leading to a vicious cycle [44,52]. As a result, a restoration initiative is doomed to fail if they do not relieve the ecosystems of the human stresses.

Thanks to increased employment and income from non-farming sources, many farmers in western China are now no longer dependent on the degraded cropland for food [29,44]. This shift has mitigated the pressures put on land as a source of livelihood. Meanwhile, farming on reduced cropland through more intensive use of modern inputs can boost crop yields and ease the loss of grain production [53,54]. However, the development of local entrepreneurship, leadership, and social capital has not received sufficient attention [18,49]. Despite the program subsidies, especially during the first round (2001-2008), the internal incentives are incoherent, which have been further diminished by the expanded compensations to growing annual crops and livestock in recent years [29].

Also, the proponents of achieving conservation goals and poverty reduction separately ignore the fact that it is premised on: (1) setting clear conservation goals; (2) understanding the associated opportunity costs; and (3) knowing the local poverty dynamics. But it is difficult, if not impossible, for any country to pursue these tasks simultaneously. Alternatively, conservation and poverty reduction goals may be decoupled at a higher aggregate level, especially with the adoption of conservation contracts auctioning. Again, this is because any meaningful conservation objective has to be accomplished at the level of landscape that contains small plots of many households [44,46].

6. Closing remarks

Elucidating the complex issues involved in PES design and implementation and searching for practical solutions to PES governance has become as an important international research topic [33,34]. By comparing China's SLCP to the U.S. CRP—probably the two largest PES programs in the world, our analysis has yielded a number of valuable insights for China and many other countries who have launched PES programs or are contemplating to do so. While we do not believe that the U.S. CRP has been perfectly designed and implemented, it is beyond the scope of this study to critique it.[4] Instead, we have focused our attention on the question of what China, as a developing and transitioning economy, can learn from comparing its experience of ecological restoration, epitomized by the SLCP, to that of the CRP in the U.S., as a developed and mature market economy.

It has been made clear that PES programs, especially the large ones, are intricate and challenging undertakings [16,18,25] and carrying them out entails long-term interactions of various components of the underlying social-ecological systems and lead to multiple, often mixed, and uncertain outcomes [9,17]. Rather than narrowly characterizing them as voluntary, conditional transactions, it is more constructive to view them in terms of their provision of multiple environmental public goods on expansive spatial and temporal

[4] Readers interested in this subject may refer to Stubbs [38] and Claassen and others [39] for some detailed discussion.

scales and involving a diversity of consumers and producers [55,56]. This broader perspective has in turn enabled us to examine the governance specifics and thus propose more practical means and mechanisms for effective design and execution of PES in general and the SLCP in particular.

The SLCP differs from some local PES schemes in China and elsewhere in several important ways [9,49]. First, it was intended to deliver multiple ES, whereas local, small schemes often target one or two specific ES. Second, the SLCP also aims to reduce poverty and improve livelihoods, but many local schemes may not have to consider similar social objectives. Further, local schemes may be able to tie payments directly to the particular service(s) delivered or promised, while farmers participating in the SLCP are paid for changes to their land-use practices. Delineating and coordinating these practices for millions of smallholders is absolutely critical, which implies that individual participation may not be completely voluntary and that it seems flawed to characterize the implementation of such a large program simply as market-based transactions [22,23]. In part, this has to do with the short time for farmers to learn the restoration practices and respond to the announced policies, and the actions to be undertaken by government agencies and local communities [48,57]. Thus, partnerships between individual farmers, community organizations, and local government agencies must be formed to enhance the likelihood of successful implementation of the program.

Also, there is a significant lag between the time when payments are made and the time when ES are provided [40,41]. This indicates not only the difficulty to tie the payments and the expected ES directly but also the need to associate payments with certain preferred practices of land use, leading to improved generation of the desired ES. On the one hand, the critical relevance of monitoring the implementation effectiveness and the maintenance of policy consistency in the process calls for better conceived and more stable measures over time [29,55]. In addition, because the beneficiaries of the ES are widely dispersed, it is hard to identify them and then impose any charge on them for compensating the ES providers. So, the conditionality attached to payments is not only indirect but also low, and, as mentioned, the sanction for those who default their responsibilities has generally been weak [18], further calling the implementation effectiveness into question.

On the supply side, farmers may not have a high expectation on their individual rights to the ES, especially given the nature of the services to be generated (as public goods), coupled with the scale dis-alignment between their small plots and the landscape-level functionality of the restored ecosystems [55,58]. In this context, the right to exclude third parties may not be feasible or appealing [26,29], suggesting the difficulty of realizing land ownership and use rights under certain PES schemes. Thus, it may be an unrealistic and futile attempt to call for quick tenure clarification or ecosystem privatization for the sake of commercializing ES. At the same time, there is a great need for nurturing intermediaries and aggregators who can organize, coordinate, and verify the ES producers, consumers, or both [29,40]. Until now, however, these roles have been played by local and regional governments, which tend to be inefficient and insufficient.

In summary, the policy messages that this comparative analysis of the large PES programs of restoring degraded cropland to forest and grass covers in the U.S. and China has uncovered are important to not only China but also many other developing and/or transitioning economies. Our study has also called for a more practical and better integrated approach to PES design, implementation, and evaluation, which is expected to lead to more effective ecosystem restoration and biodiversity conservation.

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