8 Spillover effects worldwide

From here on, we concentrate on evidence of the spillover effects based on nine cases outside of China and the USA. Rather than presenting on a case basis, we summarize what we found based on our conceptual framework (Figure 1.3). To cover all the spillover effects, we still mention the findings from the USA and China cases but refer to Chapters 5–7 for details.

8.1 *Policy–Behavior* cross-program spillover effects

Policy–Behavior spillover effects refer to one (kind of) policy that may change actions expected from another policy (Figure 1.3). To illustrate such spillover effects, we first examined two concurrent payments in China. China launched its Grain-to-Green Program (GTGP) from 1999 to 2001, providing farmers with cash and/or grain subsidies to convert croplands on steep slopes or otherwise ecologically sensitive areas into forestland or grassland. In 2001, China started another program providing extensive payments for environmental services (PES)—the Forest Ecological Benefit Compensation (FEBC) Fund—seeking to protect and manage natural forests for public benefit. Since 2004, these two programs have been implemented simultaneously in 20 provinces, autonomous regions, and municipalities. GTGP-eligible land parcels are farmland on steep slopes, whereas FEBC parcels are natural forestlands, thus spatially disconnected from GTGP parcels. In many regions, parcels of both types of land are contracted to the same households (Yost et al., 2020), making them horizontally stacked payments.

Our case studies show that *Policy–Behavior* spillover effects have occurred in two nature reserves in China. Using data from Fanjingshan National Nature Reserve, China, we found that each 1,000 Yuan of FEBC payment (Policy 2) would increase land enrollment (Behavior 1) in response to GTGP payment (Policy 1) by 0.4487 mu (1 mu = 1/15 ha; p = 0.0064; Table 5.2). Compared to GTGP, the payment rate of FEBC is much lower while the land area of FEBC is much higher (Table 4.1); households in Tianma National Nature Reserve, China, received about twice as much compensation from FEBC as that from GTGP. Similar *Policy–Behavior* spillover effects are found in Tianma: under FEBC payment (Policy 2), every 100 mu of forest land generates 0.47 mu more land enrolled in GTGP (Policy 1, p = 0.0020; Table 6.7) based on our regression model

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when controlling for other confounding variables (for the potential mechanism, see Section 6.5.5).

Policy–Behavior spillover effects have also taken different forms or pathways. Every additional 1,000 yuan of FEBC payment (Policy 2) decreases the odds of out-migration (Behavior 1) by 34% (Table 6.3) at Tianma. However, the classic PES literature has long indicated a *Policy–Behavior* internal link: payment from GTGP (Policy 1) may trigger and/or facilitate out-migration (part of Behavior 1), which is also observed by our data for the Tianma case. Also, at Tianma, every 100 m closer to the nearest GTGP (Policy 1) land increases the odds of FEBC tree theft (Behavior 2) by 15.5% (Section 6.5.4), suggesting that FEBC trees are more likely to be illegally logged by other rural residents if a household is located in proximity to GTGP land. For the potential mechanism, see the analysis regarding hidden *Behavior–Behavior* spillover effects in Section 8.5.

8.2 *Behavior–Goal* spillover effects

Behavior–Goal spillover effects refer to the situation where behaviors expected from one policy might give rise to unexpected changes in the goal(s) targeted by another policy (Figure 1.3), illustrated in our case for Australia. The country has 85.3 million ha of intensive agricultural land, subject to reforestation under carbon farming policies. Modeling by Bryan et al. (2015) shows that under a policy scheme that focuses on carbon sequestration, people would establish "carbon plantings" (Behavior 1) of fast-growing Eucalyptus monocultures (Behavior 1), which sequesters a large amount of carbon (i.e., a strong internal *Behavior–Goal* link) but adds little to biodiversity (i.e., a weak *Behavior–Goal* spillover effect) (Bryan et al., 2015). Under a policy scheme that highlights both carbon and biodiversity services, the practice of "environmental plantings" (Behavior 2; mix of native trees and shrubs) gives rise to not only high levels of carbon sequestration (a strong internal *Behavior–Goal* link; only 1.32% of total carbon stock sacrificed) but also a significant gain in biodiversity (a strong *Behavior–Goal* spillover effect)—96 times that from the carbon plantings (Bryan et al., 2015).

Behavior–Goal spillover effects are also evidenced at the Mazar Wildlife Reserve in Ecuador (Bremer et al., 2016). PROFAFOR (Programa FACE de Forestación del Ecuador; Policy 1) aims to promote afforestation with *Pinus* species and some native Andean species (Behavior 1) in the hope of enhancing carbon sequestration (Goal 1). At the same time, the SocioPáramo program (a subprogram of the more extensive SocioBosque program; Policy 2) seeks to exclude burning in Páramo grasslands (Behavior 2) for multiple ecosystem services of carbon storage, biodiversity protection, and water provision (Goal 2). However, afforestation (Behavior 1) has caused decreased soil moisture and loss of native plant diversity, compromising Goal 2; the soil was significantly drier under pines, having a volumetric soil moisture content of 13–22% compared to 50–74% at grassland sites. In pine plantation plots, the estimated species richness decreased to 42% of the native Páramo communities. In the same site, the burning-exclusion action (Behavior 2) may not achieve optimal carbon sequestration results (Goal 1), representing another *Behavior–Goal* spillover effect.

8.3 *Goal–Policy* spillover effects

Goal–Policy spillover effects imply that goals or outcomes (e.g., Goal 1) generated or enhanced from one policy (Policy 1) may loop back to affect the other policy(s) (Policy 2; Figure 1.3). Recently in the USA, a concurrent PES scheme called PES stacking has emerged. However, "there are no regulations addressing stacking or any guidance documents from US federal resource agencies" (Robertson et al., 2014), nor any evidence-based guidelines about how to achieve or improve the intended ecosystem services.

Evidence of *Goal–Policy* spillover effects was found in the Neuse River Basin in North Carolina, USA (Program Evaluation Division, 2009). The North Carolina Department of Transportation paid US$3.5 million for wetland credits in 2000, aiming to restore ecosystem services on 438.5 acres of wetlands. Of these 438.5 acres, 69.5 acres were used by another government agency—the Division of Water Quality (under the North Carolina Department of Environment and Natural Resources)—to certify nutrient offset credits in 2008. Of these 69.5 certified acres, 46 acres received US$698,372 for nutrient offset credits in 2009. This payment of US$698,372, temporally stacked on the same 46 acres that had received wetland payment, was considered "double-dipping" for generating no additional value. In response to this controversy and related public pressure, the North Carolina Division of Water Quality decided to rescind this stacking in the future while still honoring the overlapping nutrient offset credits certified in 2008. Assuming a uniform payment rate for the 438.5 acres of wetland, the 46 acres of wetland should have received US$367,160 for wetland credits in 2000. Given that the same 46 acres of wetland had also received a payment of US$698,372 for nutrient credits in 2008, such stacking of payments has amounted to a "double-dipping" rate of 190% (i.e., US$698,372/US$367,160), implying a big waste in conservation payments.

8.4 *Policy–Policy* spillover effects

Policy–Policy spillover effects occur when one policy directly leads to changes in another policy (Figure 1.3), as in the Baltic Sea case (Gren & Elofsson, 2017). To counter the severe eutrophication problem in the Baltic Sea, nine countries in the catchment (i.e., Denmark, Finland, Germany, Poland, Sweden, Estonia, Latvia, Lithuania, and Russia) have agreed to reduce nitrogen (N) and phosphorous (P) loads entering the sea. Specifically, these countries implemented several abatement measures that reduce the total N and P loads below predetermined annual limits. To minimize total abatement costs, N and P emission permits can be traded in markets among various actors (e.g., abatement firms), between upstream and downstream areas, and across different abatement measures as long as the total N and P caps are observed (Gren & Elofsson, 2017). Gren and Elofsson have demonstrated mathematically that, to be cost effective, payments for N and P abatement (Policies 1 and 2) must coexist and be stacked (Gren & Elofsson, 2017). Many abatement actions, such as wetland construction, generate N and P reductions. If only one policy (through markets of N or P credits) is allowed, the outcome would be much worse, e.g., either higher costs or caps not observed (Gren & Elofsson, 2017).

The Jordan Lake case provides additional evidence of *Policy–Policy* spillover effects, where the stacking of two payments may or may not function well depending on the relative sizes of the payments (Motallebi et al., 2018). In this case, the primary goal—ecosystem service—is the reduction of N loads into Jordan Lake (Goal 1) by all farmers in the watershed, as required by the water quality trading program in North Carolina. In a hypothetical scenario, Goal 2 is considered to reduce P by providing P credits. Both goals are a joint outcome of single conservation practice: building or extending a vegetated riparian buffer (Behavior 1), which aims to increase Goal 1. When the amount of payments under Policy 2 (for P reduction) is between 20% and 30% of Policy 1 (for N reduction), the so-called "double-dipping" occurs, in which a stacked payment (under Policy 2) increases farmers' revenue but does not change their conservation action (Behavior 1).

Complementary to the Jordan Lake case, the Rio Grande catchment case from Bolivia presents spillover effects between concurrent payments made to different parcels but contracted to the same individuals (i.e., horizontal stacking) to conserve biodiversity and improve water quality (Goal 1) (Bottazzi et al., 2018). Payments at level 1 (Policy 1), which are much higher in amount and stricter in monitoring for compliance, seem to downgrade or nullify payments made at levels 2 and 3 targeted on different lands (Policy 2). In this case, local farmers were less compliant with their contracts that required them to stop farming (Behavior 2).

8.5 *Behavior–Behavior* cross-program spillover effects

Behavior–Behavior spillover effects refer to cross-behavior influences, where payment-induced actions affect each other. Our first evidence comes from Wolong Nature Reserve, which is conserved for the giant panda (*Ailuropoda melanoleuca*) and many other plants and animals of high value in the same area. Two concurrent payments exist: The National Forest Conservation Program (NFCP) and a GTGP-like program called Grain to Bamboo Program (GTBP). Yang et al. (2016) found that payments from GTBP and NFCP, when implemented alone, each had a significant negative impact on the growth in household income from 1998 to 2007. However, the two payments positively impacted this income growth when implemented together. This surprising outcome could arise from local people's changes in their livelihood strategies; local farmers switched from agricultural intensification (Behavior 1) to out-migration (Behavior 2), substantially reducing or even abandoning farming activities (detail in Chapter 7).

The Australian case also supports this *Behavior–Behavior* type of spillover effects. The two actions, i.e., establishment of "carbon plantings" (CP; Behavior 1) and "environmental plantings" (EP; Behavior 2), are subjected to a quantitative restriction; the sum of the CP area, the EP area, and the traditional cropland area should be 85.3 million ha, where increases in CP are coupled with decreases in EP due to constraints in total budget and area of land available (Bryan et al., 2015). *Behavior–Behavior* spillover effects can also be observed at Tianma National Nature Reserve in Anhui Province, China. As shown in our discussion of *Policy–Behavior* spillover effects (Section 8.1), payments from GTGP (Policy 1) may lead to an unintended behavior of tree theft on FEBC land (Behavior 2). This *Policy–Behavior* spillover effect may arise from a *Behavior–Behavior* spillover effect: migration of the whole family or farm laborers to cities for higher-paying employment (Behavior 1) may fail to monitor FEBC forests belonging to the corresponding households, increasing the chances of timber theft in these forests (Behavior 2).

8.6 *Goal–Goal* spillover effects

Goal–Goal spillover effects arise when ecosystem services, especially those implemented nearby or in a co-located manner, interact with one another via various biophysical and ecological processes operating at various spatial and temporal scales (Figure 1.3). Here we use the Foglia River Basin and Marecchia River Basin in Italy as an example. Among the forest-based ecosystem services identified by Morri et al. (2014), water retention (Goal 1) and drinking water supply (Goal 2) are linked conceptually and quantitatively; water retention—a function of forest type (which determines the percentage of runoff retained) and its area—is the source of drinking water. In addition, the two goals of soil protection (amount of soil erosion avoided) and CO_2 sequestration are both a function of forest type and its area, with a few other variables under control. For the policy schemes related to the above goals and the potential "double-dipping" problem, we refer to Sections 2.3 and 8.3.

Data from the New World (the Americas and Oceania) and Great Britain may contribute to understanding these *Goal–Goal* spillover effects. Compared to a carbon-only (Goal 1) strategy, a combined carbon-biodiversity strategy—weighing the two goals and adjusting subsequent spatial allocation—could simultaneously protect 90% of carbon stocks and more than 90% of the biodiversity protected under a biodiversity-only (Goal 2) strategy. This win–win gain arises from heterogeneous spatial distributions of—and site-specific interactions between—biodiversity and carbon goals (Thomas et al., 2013). Similar win–win outcomes are also observed in the Australia case due to reallocating payments to sites with abundant biodiversity and carbon goals (Bryan et al., 2015).

Goal–Goal spillover effects are also found in the Neuse case, where Goal 1 (derived from wetland payment) automatically entails Goal 2 (derived from nutrient payment). This *Goal–Goal* spillover effect becomes the rationale for the aforementioned *Goal–Policy* spillover effect at Neuse: Goal 1 (wetland credits) already existed and should continue generating co-benefits of nutrient offset (Goal 1) for which Policy 2 (nutrient offset credits) is intended, leading to the North Carolina

Division of Water Quality's decision to rescind Policy 2. Similarly, in the Jordan Lake case, nitrogen reduction (Goal 1) that comes with constructing riparian buffers would bring in phosphorous reduction (Goal 2) because of N and P cycling processes (Motallebi et al., 2018). In the context of many ecological services of GTGP and FEBC, it has been reported that such *Goal–Goal* spillover effects exist. Once forests are established under GTGP that are often closer to households, FEBC forests are better protected, as GTGP forests may act as buffers for human activities such as fuelwood collection and grazing that would otherwise occur in FEBC forests (Song et al., 2018).

8.7 Yucatán and Chiapas

Mexico is a pioneer Latin American country in implementing a nationwide Ecosystem Services-Hydrological program (PSA-H) policy to protect critical forests for water provision and regulation services. Beginning in 2003, the PSA-H payments were granted to forest communities for five consecutive years after signing a contract by community elected leaders and Mexico's National Forestry Commission (CONAFOR). Simultaneously, another concurrent PES program was named the Forest Ecosystems Conservation and Restoration Program (PROCOREF in Spanish). According to Ezzine-de-Blas et al.'s survey of 77 communities (*ejidos*) in 2013 in Southern Yucatán (Ezzine-de-Blas et al., 2016), we performed a correlation analysis among the area (unit: ha) of enrollment in PSA-H (ha_psa), the payment that each community received from PSA-H in 2012 (value_psa), and the area (unit: ha) of enrollment in PROCOREF (procoref1_ha). The goal is to examine whether spillover effects exist between the two PES programs.

PSA-H had a positive correlation with the area of PROCOREF enrollment in terms of both area ($r=0.3453$, $p=0.1360$) and amount of payment ($r=0.3500$, $p=0.1303$), suggesting potential *Behavior–Behavior* spillover effects (Table 8.1). However, these two relationships were insignificant (but close to significant) at the alpha = 0.10 level (Table 8.1). The effective sample size was only 20 compared to a total sample size of 77. As the original paper by Ezzine-de-Blas (2016) did not focus on cross-program spillover effects, most records lacked data in one, two, or all of the three payment-related variables. This case shows that a lack of explicit focus on cross-program spillover effects would limit the usefulness of such data when examining cross-program spillover effects.

8.8 *Time–Time* spillover effects

Time–Time spillover effects occur among various payments that evolve (Figure 1.3), depending on the socioecological context in which they are embedded. As shown previously, we have observed positive spillover effects from FEBC payment to GTGP enrollment in China's two nature reserves. To explore whether existing spillover effects may change over time, we designed questions in a household survey, asking local villagers about their willingness to enroll

Table 8.1 Results of Pearson correlation analysis

Variable	*Area of enrollment in PSA-H (unit: ha)*	*Payment that each community received from PSA-H in 2012*	*Area of enrollment in PROCOREF (unit: ha)*
Area of enrollment in PSA-H (unit: ha)	1.0000 51	0.9756 (<0.0001) 51	0.3453 (0.1360) 20
Payment that each community received from PSA-H in 2012	0.9756 (<0.0001) 51	1.000 51	0.3500 (0.1303) 20
Area of enrollment in PROCOREF (unit: ha)	0.3453 (0.1360) 20	0.3500 (0.1303) 20	1.0000 28

Note: Correlation coefficient, *p*-value, and number of records used in the analysis (sample size) are included in the table.

more cropland in GTGP under a set of hypothetical conditions. Our data analysis revealed a negative relationship: each mu of FEBC land (a proxy of FEBC payment) decreased the odds of GTGP participation by 0.30% (equivalent to a probability decline of 3.0% for a typical household at Fanjingshan). This finding is also corroborated by a similar study conducted one year earlier at the same site (Yost et al., 2020), which found that each mu of the FEBC land would decrease the odds of GTGP participation by 0.58%. Although these negative spillover effects are minor in magnitude, they have evolved from significant positive spillover effects (Table 5.4) over a relatively short period. Farmers with more FEBC land may have enrolled most of their eligible cropland parcels in GTGP, leaving little additional land for the hypothetical GTGP. Aside from this land-scarcity issue, concerns for food security may preclude farmers from enrolling additional cropland (Yost et al., 2020).

The PVPF-KPWS case in Cambodia also shows evidence of *Time–Time* spillover effects (Clements et al., 2010). Payments from the Bird Nest Protection (BNP) program (Policy 1) are made to eligible individuals who then locate, monitor, and protect the remaining nesting sites (Behavior 1a). However, villages receiving such BNP payments allowed in-migrants to settle locally (Behavior 1b); these in-migrants tended to clear forests and cause a more significant loss of bird habitat, offsetting the conservation effects of BNP in the long run. On the other hand, payments from the Ecotourism and Agri-Environment (E&AE) programs (Policy 2) may take several years to build up the capacity of all participating villages and individuals. However, once such capacity is established, such payments may lead to restraining in-migration (Behavior 1b) and the associated deforestation, contributing to bird conservation. The best conservation outcome may come from sequential implementations of these two payments: the BNP first (for immediate effect) and E&AE later (for long-term protection), manifesting a *Time–Time* spillover effect.

8.9 Intertwined spillover effects

As earlier sections show, spillover effects could co-occur, manifesting in an intertwined, multi-dimensional style. We examine two concurrent green initiatives that are widely implemented globally as an example. The first one is Community-based Forest Management (CFM), which currently manages about 18% of the global forest in 62 countries, supporting hundreds of millions of people (Gilmour, 2016). The CFM is decentralized management of natural resources, aiming at sustainable forest development and livelihood improvement for the local communities participating in forest management. CFM devolves the right of forest management decision-making to the local communities and the responsibility of forest conservation. Devolvement of responsibility from the central government to the local communities is perhaps the most significant paradigm shift in forest management policy since the mid-1980s when the central government dominated forest management decision-making without considering the need of the local communities living around the forests. The centralized forest management has led to widespread deforestation and forest degradation, such as the well-known Himalaya Ecological Crisis (Eckholm, 1975).

The second green initiative is the Reducing Emissions from Deforestation and forest Degradation, plus sustainable forest management, conservation, and enhancement of forest carbon stocks (REDD+), which was developed by the Parties to the UN Framework Convention on Climate Change. REDD+ provides results-based payments for the carbon stored in forests in developing countries. REDD+ has been recognized as an effective mechanism for global warming mitigation. As of 2019, UN-REDD had made significant progress toward REDD+ participating countries, primarily in the developing world, toward REDD+ goals (http://www.un-redd.org). For any developing country with rich forest resources to receive payments from REDD+ for carbon storage, it first has to develop a national policy, then implement the policy, and finally fully measure, report, and verify the implementation results. Because CFM has improved forest conditions worldwide, many REDD+ pilot projects have been implemented in community forests. Dynamic spillover effects occur between the two policies and the subsequent actions and gains.

REDD+ and CFM can mutually benefit from each other because of the shared goal of sustainable forest management. The REDD+ payment for carbon storage can benefit the members of Community Forest User Groups (CFUGs). At the same time, REDD+ can take advantage of the natural, social, and institutional capitals that have been accumulated through CFM to achieve its goals (Newton et al., 2015). The improved forest conditions under CFM provide the biophysical basis for REDD+ projects. The bonding social capital developed over time within the CFUGs and the institutions for forest governance would significantly benefit REDD+ project management. However, there are also divergent goals between the two initiatives. Conservation and enhancement of forest carbon stocks for global warming mitigation pursued by REDD+ may compromise the use of forest products in community forests for livelihood support for forest-dependent people under CFM. Although REDD+ payments are made to the corresponding forest management communities for carbon storage, the payments may or may not make up for the loss of support they used to derive from the community forests (Maraseni et al., 2014; Marquardt et al., 2016). The use of forest products often constitutes carbon leakage from the community forests, compromising the REDD+ goals.

Key actors in formulating REDD+ policy include the national government, international donors, NGOs, and civil society organizations, while the local communities where the REDD+ projects are implemented were generally not well engaged or informed (Bastakoti & Davidsen, 2014). Such a top-down approach leads to the recentralization of forest management, i.e., the local community loses the autonomy for forest management decision-making to utilize the forest resources for livelihood support, such as generating necessary revenues for local community needs and poverty alleviation through timber trade (Phelps et al., 2010). However, CFM improved forest conditions under its management regime. Not all CFUG activities contribute to conserving and enhancing forest carbon storage. Therefore, implementing the REDD+ project in a community forest leads to changes in its governance rules in CFM, which could create opportunities for elite capture of benefits while restricting the poor and marginalized people's use of forest resources (Poudel et al., 2014). Poor people who depend more on forest resources for livelihoods would be disproportionally impacted as a result (Devkota & Mustalahti, 2018). In some cases, the implementation of REDD+ seemingly enhanced the participation in decision bodies by the poor, women, and marginalized groups, but the benefits of REDD+ did not trickle down to these people (Devkota, 2020).

The primary goal of REDD+ projects is global warming mitigation via carbon removal from the atmosphere through forest growth. Although CFUGs have use rights to timber, firewood, and fodder in the community forests, there is no clear legal ownership right for the carbon accumulated in community forests. Clarification of benefit distribution of REDD+ payments becomes critical for its success. Verifiable forest carbon storage is the only criterion for receiving monetary compensation from REDD+. Although REDD+ supports sustainable forest management, no other sustainable forest management metrics beyond carbon storage receive REDD+ compensation.

In contrast, the benefits generated in community forests are multi-dimensional, including forest conservation, sustainable forest management, support for livelihoods for the local people, preservation of biodiversity, and carbon storage for global warming mitigation that benefits the entire world. The commodification of carbon via REDD+ projects could overrun community forest priorities (Bastakoti & Davidsen, 2014). The REDD+ payments to poor households are sometimes insufficient for livelihood enhancement activities (Shrestha et al., 2017).

REDD+ projects can be successfully implemented in forests under CFM (Sharma et al., 2020). However, the multiple levels of spillover effects between REDD+ and CFM have to be addressed appropriately in REDD+ policies and the process of REDD+ implementation, including a clear definition of carbon ownership and benefit distribution mechanism, carbon price, preservation of CFUGs' autonomy in forest management decision-making, and the need to accommodate poor and marginalized people's need for subsistence products from forests, such as firewood and fodder. It turns out that the spillover effects between the two initiatives with synergistic outcomes need to be strengthened. Here we take them as an example of a *Goal–Goal* spillover effect: the REDD+ program aims at enhancing forest carbon stocks while the concurrent CFM focuses on multiple ecological and livelihood gains. However, those two programs have trade-offs. For example, CFM may stimulate local people to harvest trees to satisfy subsistence needs (Behavior 1), leading to decreases in forest carbon stocks (Goal 2), suggesting a *Behavior–Goal* negative spillover effect. Therefore, such trade-offs and the relevant mechanism must be carefully addressed to generate a win–win scenario.

8.10 Evidence from policy-mix

Empirical studies of green policy-mixes classify various policy interactions into several basic categories that include: complementary or synergistic, complementary when sequential, and redundant or conflicting (Gunningham & Sinclair, 1999; Robalino et al., 2015; Santos et al., 2015). Gunningham and Sinclair further describe a myriad of theoretical mixes that can occur between the following instruments: Command and control regulation, economic instruments, self-regulations, voluntarism, and information strategies (Gunningham & Sinclair, 1999). Below are some of the specific policy-mix examples that were described in the studies by Gunningham and Sinclair (1999), Barton et al. (2012), Robalino et al. (2015), and Santos et al. (2015). We refer to the Appendix for more information.

8.10.1 Complementary or synergistic

The 1990s witnessed the development and implementation of the US Environmental Protection Agency's 33/50 program, which encouraged relevant companies or organizations to reduce toxic chemical releases voluntarily. At the same time, existing command and control regulations for toxic chemical release remained in force. Therefore, the 33/50 program complemented the regulation policy by promoting more considerable reductions of toxic chemical release than the baseline, while companies were still required to comply with baseline levels. Similarly, the US Environmental Leadership Program provides regulatory relief for participating firms that go beyond compliance levels. Also, in the European Union (EU) there is consideration of compliance and inspection exemptions for firms that participate in eco-management and eco-audit schemes. In the US and EU examples, there is a backdrop of regulation that non-participating firms must follow. There is evidence regarding the effectiveness of the command and control regulation being complemented by voluntarism instruments (Gunningham & Sinclair, 1999).

In Australia, all vehicles built after 1985 were mandated to be fitted with catalytic converters, requiring the use of engines that ran on unleaded fuel.

Concurrently, the federal government introduced a phased price differential on the fuel price. In this context, leaded fuel became more expensive than unleaded fuel, which was an economic policy in the form of a pollution tax. While they are different approaches, these two policies complement one another because they provide the market with mutually supportive signals. The technology-based approach of requiring catalytic converters is directed at the manufacturer, and the pollution tax is aimed at the consumer (Gunningham & Sinclair, 1999).

8.10.2 Sequential relationship

Policy "sequencing" refers to certain instruments or policies being held in reserve and applied when another instrument fails or has serious problems (Gunningham & Sinclair, 1999). Here we present an example in Norway: the prime "command and control" instrument was the establishment of protected areas, primarily based on the appropriation of private land in biologically rich areas. The Trillemarka Nature Reserve, totaling 147 km^2 in size, was established as a conservation area of this kind with a relatively large size. Yet this instrument encountered opposition and conflict; it was superseded by a voluntary scheme with compensation payments. The command and control way of establishing protected areas is now almost dormant, and its future is quite uncertain, likely depending on the progress and results of the voluntary scheme (Barton et al., 2012).

In 2009, Norway passed the Nature Diversity Act, which includes—and integrates—all previous laws related to land use and biodiversity in one act. As the most important legal framework, this act stands as the fundamental guidelines for future regulatory and economic instruments in the domain of forest and biodiversity conservation—both inside and outside protected areas. The act provides guidelines for the management of priority species and selected habitat types and paves the way for Norway to fulfill its international commitments under the Convention on Biological Diversity, to which Norway is a signatory (Barton et al., 2012). An umbrella law or policy like this act will be instrumental in coordinating a variety of policies, foreseeing conflicts and optimizing synergism.

In many instances, an entire self-regulatory regime may not work well. Sequential interactions may come to help when economic incentives are imposed. In New Zealand, an industry-regulated program to reduce greenhouse gas emissions was introduced along with the announcement that, if the self-regulation failed, a carbon tax would be implemented. In Australia, a voluntary phase out of hydrochlorofluorocarbons (HCFCs) was legislated along with a call for a tradable quota policy: this policy would be implemented if the self-regulation failed (Gunningham & Sinclair, 1999).

8.10.3 Redundancies or conflicting policy interactions

Policy interactions may lead to a suboptimal—even negative—outcome when a command and control instrument is superimposed on an economic instrument, and both instruments target the same behavior (Gunningham & Sinclair, 1999).

In Costa Rica, a study was conducted to evaluate the effectiveness of two prevalent forest conservation policies that interact with each other: one is the policy of national parks and the other payments for ecosystem services (PES) programs; both focus on forest protection. The study area consists of park areas, park buffers with and without PES programs, and areas with PES programs far from parks. The study found a redundancy effect: the associated benefits of implementing parks and PES payments separately are greater than implementing them together. More specifically, it became more effective if one location was protected by a park and another by a payment than if one location was protected by both (Robalino et al., 2015).

Self-regulation and broad-based economic instruments may become incompatible, and here we show an example regarding the policy-mix used to phase out chlorofluorocarbon (CFCs) in Australia. As part of the National Ozone Strategy, the federal government imposed a cap on the production and importation of CFCs. Firms were allowed to trade CFC quotas under the condition that total CFCs were below the cap. After the inception of this program, federal and state governments brokered self-regulatory agreements with sector-specific industries to phase out the use of CFCs. This self-regulatory policy contradicted the cap, ultimately leading to the economic policy's failure (Gunningham & Sinclair, 1999).

In the USA, the Environmental Protection Agency's XL (eXcellence in Leadership) initiative was designed to give firms the flexibility of adopting less prescriptive, performance-oriented regulations. This policy was not successful partially because firms were concerned that the best available technology (BAT) regulations might still apply. Therefore, even if firms participated in an XL project, they might still be subject to penalties for failing to comply with the Clean Air and Clean Water Acts (Gunningham & Sinclair, 1999).

8.10.4 Economic instruments

Habitat Banking and Tradable Development Rights (TDR) stand as two beneficial economic instruments that play a role in the policy-mix sequential relationships. Such relationships include complementarities, redundancy, and conflicts with other instruments (Santos et al., 2015). Habitat banking aims to restore, create, enhance, or preserve off-site areas to provide compensatory mitigation for authorized impacts on habitats or biodiversity. A public agency, private organization, or landowner, rather than the developer, can establish conservation areas as mitigation for permitted impacts on biodiversity and ecosystems. The Wetland Mitigation Banking (the USA), Conservation Banking (the USA), New South Wales BioBanking (Australia), and BushBroker (Australia) are good examples of habitat banking schemes (Santos et al., 2015).

TDRs belong to a market-based approach, which aims to enhance land-use zoning by limiting land development and promoting biodiversity conservation. In designated areas, landowners are assigned TDRs as compensation for restricted development options, whereas in predicted growth areas, developers can choose to build at a baseline density or buy TDRs in order to realize a denser level of development. However, these methods may still generate ecological losses in return for the recreation or restoration of equivalent habitats. Both habitat banking and TDRs work in conjunction with a strong regulatory framework because the framework is necessary to ensure the adoption of the mitigation hierarchy and to determine the impacts to be offset. In most instances, habitat banking and TDR build on—and temporally follow—existing regulatory approaches to biodiversity offsetting and land-use zoning. For this reason, both instruments are characterized by sequential interactions or path dependence (Santos et al., 2015).

Under the Australian Threatened Species Legislation Amendment Act 2004, the New South Wales (NSW) BioBanking scheme was designed to support the biodiversity certification process. This scheme was consistent with the property vegetation planning process. It leverages provisions from other acts to ensure that the scheme and management actions are enforceable. Developers can choose between adopting the habitat banking scheme or negotiating an offset with the NSW government. The latter was their only option before introducing the BioBanking scheme. Developers are thus free to choose between offsetting the impact by themselves and purchasing the required credits. This kind of overlap promotes the flexibility and cost effectiveness of the overall policy-mix, likely leading to better achievement of conservation goals.

Habitat banking complements several other European Union policies, such as the Common Agricultural Policy and the Habitats Directive. Habitat banking may be instrumental in tackling the cumulative fragmentation of Europe's habitats by helping to restore, enlarge, and reconnect high nature value habitats. Yet the implementation of habitat banking and TDRs may give rise to the problem related to lack of additionality. If biodiversity outcomes would have occurred automatically as a result of existing instruments, such as management obligations set up for Natura 2000 sites, then people may ask why additional efforts were made for habitat banking and TDRs (Santos et al., 2015).

8.11 Summary of concurrent green initiatives

We present the descriptive data of all 15 cases, showing the country or continent, population size, area, urban or rural area, developed or developing countries or regions, funder type, and name of concurrent programs (Table 8.2). This summary table shows that concurrent green initiatives can be observed in most parts of the world regardless of the above variables. We also include empirical studies of green policy-mixes, which also point to the widespread existence of concurrent green initiatives and spillover effects among them.

Furthermore, we present the data sources regarding the population size and area of all the 15 cases in Tables 8.3 and 8.4, respectively. In many types of spillover effects, we have found both positive and negative effects—e.g., for *Behavior–Goal* spillover effect, we found a positive one in Australia and a negative one in Páramo (Section 8.2). In other instances, two elements can be achieved simultaneously (see the *Goal–Goal* spillover effects; Section 8.6) or must occur in sequence (Policy 1 and Policy 2 occur in sequence; Section 8.4). These findings

Table 8.2 Characteristics of the 15 selected cases[a]

Case name	*Case ID*	*Country/continent*	*Population (area)*	*Urban–rural*	*Development level*	*Funder type*[b]	*Concurrent payment names*[c]
USA	1	USA/N. America	332,639,102 (9,147,593 km^2)	Urban and rural	Developed	Gov	EQIP and CRP
Jordan Lake	2	USA/N. America	719,888 (4,367 km^2)	Urban and rural	Developed	Non-gov	Nitrogen credits vs. phosphorus credits
Neuse	3	USA/N. America	1,687,462 (15,700 km^2)	Urban and rural	Developed	Gov	Wetland credits vs. nutrient credits
Yucatán and Chiapas	4	Mexico/N. America	7,315,083 (112,835 km^2)	Rural	Developing to developed	Gov	PSA-H vs. PROCOREF
Páramo	5	Ecuador/S. America	N/A (18 km^2)	Rural	Developing	Gov	PROFAFOR vs. SocioPáramo program
Rio Grande catchment	6	Bolivia/S. America	N/A (7,339 km^2)	Rural	Developing	Gov	Level 1 payments vs. Levels 2 and 3 payments
New world[d] & Great Britain	7	Americas–Europe	63,200,000 (219,949 km^2)	Urban and rural	Developing to developed	Gov	Carbon only vs. carbon-biodiversity payments
Baltic Sea	8	Denmark etc./Europe	85,000,000 (1,720,270 km^2)	Urban and rural	Developed	Gov	Nitrogen payment vs. phosphorus payment
Marecchia & Foglia	9	Italy/Europe	404,800 (1,310 km^2)	Urban and rural	Developed	Un-specified	Water retention vs. drinking water supply
Fanjingshan	10	China/Asia	21,000 (419 km^2)	Rural	Developing	Gov	GTGP vs. FEBC
Tianma	11	China/Asia	17,295 (289 km^2)	Rural	Developing	Gov	GTGP vs. FEBC
Wolong	12	China/Asia	5,950 (2,000 km^2)	Rural	Developing	Gov	GTGB vs. NFCP

PVPF-KPWS	13	Cambodia/Asia	249,304+(5,925 km^2)[e]	Rural	Developing	Gov, Non-gov	Bird-nest program vs. ecotourism and agri-environment programs
Australia	14	Australia/Oceania	23,401,892 (7,741,220 km^2)	Urban and rural	Developed	Gov	Carbon only vs. carbon-biodiversity payments
Nepal	15	Nepal/Asia	26,490,000 (147,181 km^2)	Rural	Developing	Gov, Non-gov	REDD+vs. CFM

Notes:
[a] For sources of information, see Tables 8.3 and 8.4.
[b] "Gov" and "Non-gov" refer to governmental and non-governmental sponsors, respectively.
[c] For information about acronyms, see the following text.
[d] New World refers to the Americas and Oceania; the numbers 63,200,000 (219,949 km^2) are only about Great Britain (the UK) based on the source.
[e] The data for Population in Kulen Promtep (KP) Wildlife Sanctuary is unknown; the numbers are only for Preah Vihear (PV).

Table 8.3 Sources of population information in Table 8.2

Site name	*Population reference*
Fanjingshan	Wandersee, S.M., An, L., López-Carr, D., & Yang, Y. (2012). Perception and decisions in modeling coupled human and natural systems: A case study from Fanjingshan National Nature Reserve, China. Ecological Modelling, 229, 37–49.
Wolong	Xu, J., Wei, J., & Liu, W. (2019). Escalating human–wildlife conflict in the Wolong Nature Reserve, China: A dynamic and paradoxical process. Ecology and Evolution, 9(12), 7273–7283.
Tianma	Zhang, Q., Bilsborrow, R. E., Song, C., Tao, S., & Huang, Q. (2019). Rural household income distribution and inequality in China: Effects of payments for ecosystem services policies and other factors. Ecological Economics, 160, 114–127.
Neuse	State of North Carolina, North Carolina Department of Environment and Natural Resources, & Office of Environmental Education and Public Affairs. (2013). *Neuse River basin*. NC: North Carolina Department of Environment and Natural Resources, & Office of Environmental Education and Public Affairs. Retrieved from https://files.nc.gov/deqee/documents/files/neuse.pdf
Jordan Lake	NC Geographic Information Coordinating Council (GICC). (2018). *2000_Census_Blocks* [Shapefile]. Retrieved from https://hub.arcgis.com/datasets/nconemap::2000-census-blocks
Páramo	n/a
Marecchia & Foglia	Morri, E., Pruscini, F., Scolozzi, R., & Santolini, R. (2014). A forest ecosystem services evaluation at the river basin scale: Supply and demand between coastal areas and upstream lands (Italy). Ecological indicators, 37, 210–219.
Baltic Sea	Lääne, A., Kraav, E., Titova, G., United Nations Environment Programme (UNEP). (2005). *Global International Waters Assessment: Baltic Sea, GIWA Regional assessment 17*. Kalmar, Sweden: University of Kalmar.
Australia	Australian Bureau of Statistics. (2019). *2016 Census QuickStats*. Retrieved from https://quickstats.censusdata.abs.gov.au/census_services/getproduct/census/2016/quickstat/036
New World & Great Britain	United Nations Department of Economic and Social Affairs. (2018). *2017 Demographic Yearbook*. NY: United Nations. Office for National Statistics. (2012). *2011 Census: Population Estimates for the United Kingdom, March 2011*. Retrieved from https://www.ons.gov.uk/peoplepopulationandcommunity/populationandmigration/populationestimates/bulletins/2011censuspopulationestimatesfortheunitedkingdom/2012-12-17
Preah Vihear & Kulen Promtep	OCHA ROAP. (2018). khm_pop_2016_adm3_v2 [csv]. Retrieved from https://data.humdata.org/dataset/cambodia-population-statistics

Rio Grande Valles	Instituto Nacional de Estadística. (2012). Resultados: Censo de Poblacion y Vivienda 2012. Retrieved from http://datos.ine.gob.bo/binbol/RpWebEngine.exe/Portal?BASE=CPV2012COM&lang=ESP
USA	Umited States Census Bureau. (2019). Population estimates, July 1, 2019, (V2019) – United States [data table]. QuickFacts. Retrieved from https://www.census.gov/quickfacts/fact/table/US
Yucatán and Chiapas	Instituto Nacional de Estadística, Geografía e Informática (INEGI). (2015). Encuesta Intercensal 2015: Principales Resultados. Retrieved from https://www.inegi.org.mx/contenidos/programas/intercensal/2015/doc/eic_2015_presentacion.pdf
Nepal	CBS, 2011. National Population and Housing Census 2011 (National Report). Central Bureau of Statistics (Nepal). June 22, 2011. https://web.archive.org/web/20130418041642/http://cbs.gov.np/wp-content/uploads/2012/11/National%20Report.pdf

Table 8.4 Sources of area information in Table 8.2

Site name	*Area reference*
Fanjingshan	Tsai, Y.H., Stow, D., Chen, H.L., Lewison, R., An, L., & Shi, L. (2018). Mapping vegetation and land-use types in Fanjingshan National Nature Reserve using Google Earth Engine. Remote Sensing, 10(6), 927.
Wolong	An, L., He, G., Liang, Z., & Liu, J. (2006). Impacts of demographic and socioeconomic factors on spatio-temporal dynamics of panda habitat. Biodiversity & Conservation, 15(8), 2343–2363.
Tianma	Xu, J.L., Zhang, Z.W., Zheng, G.M., Zhang, X.H., Sun, Q.H., & McGowan, P. (2007). Home range and habitat use of Reeves's Pheasant Syrmaticus reevesii in the protected areas created from forest farms in the Dabie Mountains, central China. Bird Conservation International, 17(4), 319–330.
Neuse	State of North Carolina, North Carolina Department of Environment and Natural Resources, & Office of Environmental Education and Public Affairs. (2013). *Neuse River basin*. NC: North Carolina Department of Environment and Natural Resources, & Office of Environmental Education and Public Affairs. Retrieved from https://files.nc.gov/deqee/documents/files/neuse.pdf
Páramo	We only used data of Mazar Wildlife Reserve within Páramo. *Mazar Wildlife Reserve (MWR) general information*. Retrieved from https://docplayer.net/217990-Mazar-wildlife-reserve-mwr-general-information.html
Marecchia & Foglia	Morri, E., Pruscini, F., Scolozzi, R., & Santolini, R. (2014). A forest ecosystem services evaluation at the river basin scale: Supply and demand between coastal areas and upstream lands (Italy). Ecological indicators, 37, 210–219.
Baltic Sea	Lääne, A., Kraav, E., Titova, G., United Nations Environment Programme (UNEP). (2005). *Global International Waters Assessment: Baltic Sea, GIWA Regional assessment 17*. Kalmar, Sweden: University of Kalmar.
Australia	United Nations Department of Economic and Social Affairs. (2018). *2017 Demographic Yearbook*. NY: United Nations.
New World	United Nations Department of Economic and Social Affairs. (2018). *2017 Demographic Yearbook*. NY: United Nations.
Great Britain	United Nations Environment Programme (UNEP). (1998). *Island directory: Islands of United Kingdom*. Retrieved from http://islands.unep.ch/ICJ.htm
Preah Vihear & Kulen Promtep	Clements, T., John, A., Nielsen, K., An, D., Tan, S., Milner-Gulland, E.J. (2010). Payments for biodiversity conservation in the context of weak institutions: Comparison of three programs from Cambodia. Ecological Economics, 69(6), 1283–1291.
Rio Grande catchment	Pynegar, E.L., Jones, J.P., Gibbons, J.M., & Asquith, N.M. (2018). The effectiveness of Payments for Ecosystem Services at delivering improvements in water quality: lessons for experiments at the landscape scale. PeerJ, 6, e5753.

USA	Umited States Census Bureau. (2018). State Area Measurements and Internal Point Coordinates [data table]. Geographies, Reference Files. Retrieved from https://www.census.gov/geographies/reference-files/2010/geo/state-area.html
Yucatán and Chiapas	Instituto Nacional de Estadística, Geografía e Informática (INEGI). (2013). Información por entidad: Yucatán. Retrieved from https://web.archive.org/web/20130531052605/http://cuentame.inegi.gob.mx/monografias/informacion/yuc/default.aspx?tema=me&e=31 (for Yucatán)
	Instituto Nacional de Estadística, Geografía e Informática (INEGI). (2013). Información por entidad: Chiapas. Retrieved from https://web.archive.org/web/20130602020035/http://cuentame.inegi.gob.mx/monografias/informacion/chis/default.aspx?tema=me&e=07 (for Chiapas)
Nepal	Anup, K. C., Rijal, K., & Sapkota, R. P. (2015). Role of ecotourism in environmental conservation and socioeconomic development in Annapurna conservation area, Nepal. International Journal of Sustainable Development & World Ecology, 22(3), 251-258. https://doi.org/10.1080/13504509.2015.1005721

are exciting, yet no systematic research has been devoted to such spillover effects and the underlying mechanisms.

Appendix: Policy-mix examples

This appendix contains seven cases where two policies or more have spillover effects among one another. Given the text length and detail, we list them as cases rather than put them in a table. In each case, we identify concurrent PES policies (Policy A, Policy B, and Policy C if any), point out the spillover type, present some detail, and give the reference(s).

Case 1: Australia policy-mix

Policy “A”—The New South Wales (NSW) BioBanking (2008) is a habitat banking scheme that uses provisions from other acts to ensure that scheme and management actions are enforceable and consistent with the property vegetation planning process.

Policy “B”—Australian Threatened Species Legislation Amendment Act 2004 incorporates a biodiversity certification process in order to protect threatened species.

Spillover type: Policy “A” *supports* and *complements* Policy “B”.

Details: The banking scheme was designed for that purpose. Developers can choose between adopting the habitat banking scheme or negotiating an offset with the NSW government. The latter was their sole option prior to introducing the BioBanking scheme. Developers are thus free to choose between offsetting the impact themselves and purchasing the required credits. This kind of overlap heightens the flexibility and cost effectiveness of the overall policy-mix to achieve conservation goals.

Source: Santos, R., Schröter-Schlaack, C., Antunes, P., Ring, I. & PEDRO Clemente, P. (2015). Reviewing the role of habitat banking and tradable development rights in the conservation policy-mix. Environmental Conservation. FirstView Article 1-12.

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Case 2: The USA policy-mix

Policy “A”—US EPA’s Project XL—1995 initiative designed to promote cleaner technologies by giving firms flexibility by adopting less prescriptive, performance-oriented regulations.

Policies “B”—The Clean Air Act—1963 federal law designed to control air pollution on a national level.

Policy “C”—The Clean Water Act—1977 primary federal law in the USA governing water pollution.

Spillover type: Project XL is a major *contradiction* to the Clean Air and Clean Water Acts.

Details: Even if firms participated in Project XL, which calls for more relaxed standards, they might still be prosecuted for failing to comply with the best available technology standards established by the Clean Air and Clean Water Acts. Ultimately, Project XL failed.

Source: Gunningham, N. & Sinclair, D. (1999). Regulatory Pluralism: Designing Policy Mixes for Environmental Protection. *Law & Policy*, Vol, 21. No.1. 49–75.

Case 3: The USA policy-mix

Policy "A"—historical regulatory instruments that established protected areas such as Trillemarka Nature Reserve in Norway created on December 13, 2002 (date found on Wikipedia). This type of command and control regulation was met with opposition and conflict.

Policy "B"—Voluntary Conservation Approach—an economic instrument was proposed in 2000. Since 2003, nearly all new conservation processes have been in this voluntary form. Forest owners with biodiversity hotspots on their property can receive compensation for protecting areas as a nature reserve.

Policy "C"—Norway Nature Diversity Act established in 2009—oversees all previous and current laws related to land use and biodiversity and is a legal framework for all future *regulatory* and *economic instruments* in forest and biodiversity.

Spillover type: The voluntary conservation approach ("B") has a *sequential complementary relationship* with the historical regulatory instrument ("A").

Details: Policy "sequencing" occurs when certain instruments are applied when another instrument fails or has shortcomings. The Nature Diversity Act ("C") *complements* the previous policies because it *coordinates* these and other policies.

Source: Barton, D.N., Lindhjem, H., Rusch, G.M., Sverdrup-Thygeson, A., Blumentrath, S., Sørheim, M.D., Svarstad, H., & Gundersen, V. (2012). *Assessment of Existing and Proposed Policy Instruments for Biodiversity Conservation in Norway*. POLICYMIX Report Issue No 1/2012. Oslo, Norway.

Case 4: Nepal policy-mix

Policy "A"—In Nepal, Chitwan National Park (CNP) was established in 1973 as a protected, regulated area managed by the Department of National Parks and Wildlife Conservation. Aside from a 3-day grass collecting period, resource collection is prohibited in the park.

Policy "B"—Incentive-based programs (IBPs). These programs can empower and provide skill training for local people, while developing revenue sharing programs, sustainable extraction programs, and tourist markets.

Spillover type: IBPs (Policy "B") was set up to *complement* the CNP. However, villagers' actions do not always coincide with the views that they express about the importance of conservation.

Details: Residents surrounding CNP continue to disregard legal restrictions on resource collection. One goal is to create a link or association between the social/economic benefits and conservation efforts. Through surveys, it was determined that there had been some successes as a result due to the combination of these programs and policies. However, there is an inability to deliver benefits to the population surrounding the park. Villagers far from the entry point get fewer benefits than the gateway village.

Source: Nepal, S., & Spiteri, A. (2011). Linking Livelihoods and Conservation: An Examination of Local Residents' Perceived Linkages Between Conservation and Livelihood Benefits Around Nepal's Chitwan National Park. *Environmental Management*. 47:727–738.

Case 5: European Union policy-mix

Policy "A"—The European Union's Common Agricultural Policy (specifical provisions under more recent CAP reforms). This policy provides essential economic support to farmers sustainably managing wood pastures through direct payments to low-intensity livestock farmers for the variety of ecosystem services they provide.

Policy "B"—The EU's Rural Development Policy provides payments to wood-pasture farmers and others who go above and beyond environmental standards established under the CAP.

Policy "C"—Natura 2000—at the core of EU Habitats Directive—maintains and restores natural habitats.

Spillover type: The Rural Development Policy *supplements* the CAP provisions, yet *contradicts* the CAP because it establishes agro-forestry systems on agricultural land, some of which could be woody pastures. The Natura 2000 and EU Habitat Directive seem to directly *contradict* the CAP.

Details: Of the 233 natural habitat types included in this directive, 65 have some relationship to wood pastures, yet many are referred to as forest habitats. The criteria for forest habitats under Natura 2000 call for the restoration of tall, ungrazed, dense forests which do not allow sustainable livestock grazing in forests and do not safeguard wood pastures.

Source: Plieninger, T., Hartel, T., Martín-López, B., Beaufoy, G., Bergmeier, E., Kirby, K., Jesús Monterog, M., Moreno, G., Oteros-Rozas, E., and Van Uytvanck, J. (2015). Wood-pastures of Europe: Geographic coverage, social–ecological values, conservation management, and policy implications. *Biological Conservation*. 190: 201570–79.

Case 6: Brazilian policy-mix

Policy "A"—The Brazilian Forest Code, a federal law, establishes a percentage of the area of rural properties that are to be maintained as a permanent forest reserve. As of 1996, deforestation was prohibited on 80% of private landholdings in the "Legal Amazon" region.

Policy "B"—Ecological-Economic Zone (EEZ)—provides allocation of credit and other public incentives and allows the reserved area to be reduced to 50% in designated, productive-use areas that are involved in the EEZ. Forests may be managed for timber and non-timber production/extraction.

Spillover type: While the Brazilian Forest Code maintains a baseline for conservation requirements, the EEZ *supplements* the Code by creating allocations of credit and public incentives in productive-use areas.

Details: Other policies that complement these efforts are the 1998 Environmental Crime Law that enforces conservation efforts and streamlines court proceedings and Integrated System for Monitoring and Licensing (SIMLAM), an environmental monitoring system that integrates satellite images and forest inspections. A rural credit system is offered to landowners who register with SIMLAM, and some government banks require a declaration of compliance with the Forest Code to be screened for credit.

Source: May, P.H., Andrade, J., Vivan, J.L., Kaechele, K, Fernanda Gebara, M., and Abad, R. (2012). *Assessment of the role of economic and regulatory instruments in the conservation policymix for the Brazilian Amazon – a coarse grain analysis.* POLICYMIX Report. Issue No 5/2012. Oslo, Norway.

Case 7: Indonesia policy-mix

Policy "A"—Reducing Emissions from Deforestation and Forest Degradation (REDD+)—adopted in 2010 by the United Nations Framework Convention on Climate Change (UNFCC)—conserves forests, enhances forest carbon stocks, and sustainably manages forests.

Policy "B"—Kyoto Protocol—agroforest projects.

Spillover type: These two policies did not support one another. There were trade-offs between carbon sequestration and biodiversity.

Details: The above results were based on a carbon and biodiversity management study in Sulawesi, Indonesia. Contradictions between REDD+ projects and Kyoto Protocol agroforest projects need to be explored further.

Source: Kessler, M., Hertel, D., Jungkunst, H., Kluge, J., Abrahamczyk, S., et al. (2012). Can joint carbon and biodiversity management in tropical agroforestry landscapes be optimized? *PLOS ONE*, 7(10), e47192–e47196.

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