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# Impacts of Chilean forest subsidies on forest cover, carbon and biodiversity

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**In response to the important benefits forests provide, there is a growing effort to reforest the world. Past policies and current commitments indicate that many of these forests will be plantations. Since plantations often replace more carbon-rich or biodiverse land covers, this approach to forest expansion may undermine objectives of increased carbon storage and biodiversity. We use an econometric land use change model to simulate the carbon and biodiversity impacts of subsidy driven plantation expansion in Chile between 1986 and 2011. A comparison of simulations with and without subsidies indicates that payments for afforestation increased tree cover through expansion of plantations of exotic species but decreased the area of native forests. Chile's forest subsidies probably decreased biodiversity without increasing total carbon stored in aboveground biomass. Carefully enforced safeguards on the conversion of natural ecosystems can improve both the carbon and biodiversity outcomes of reforestation policies.**

Forests will play a critical role in humanity's efforts to address the interconnected, existential challenges posed by climate change[1–5] and biodiversity loss[6,7]. In recognition of the multiple benefits provided by forests, policymakers have adopted ambitious commitments to expand the world's forests in the coming decades. For example, the Bonn Challenge seeks to restore $350\times10^6$ ha of forests by 2030, an area equivalent to 30% of the world's remaining primary forests[8–10]. As countries outline their commitments, however, it has become clear that governments are prioritizing the planting of commercially valuable species over the restoration of natural forests. Forest plantations and agroforestry systems represent 79% of the $197\times10^6$ ha of forest restoration committed by 24 countries through 2019[8,11].

Public commitments to expand the area of forest plantations represent a continuation of decades of aggressive government support for forest plantations throughout the world[12]. Most of the world's plantation forests were established with a mix of public incentives including subsidies, tax benefits and preferential access to credit[12,13]. Although passive regeneration of natural forests may be an inexpensive and technically simple approach to reforestation[8], government reforestation incentives have often favoured the expansion of commercial plantations to generate economic benefits[14,15]. However, plantations may perform less well than natural forests for a variety of desired ecosystem services including carbon sequestration, erosion control and support of biodiversity[16–18].

The carbon and biodiversity impacts of afforestation incentives will depend upon the specific land use changes they precipitate[19]. For example, widespread afforestation in grasslands and savannahs would endanger the biodiversity of these grassy biomes[20–22]. Similarly, many countries with afforestation incentives continue to clear native forests[23,24]. Even with strong safeguards against direct conversion, afforestation incentives could reduce native forest regeneration on abandoned, agricultural lands if those marginal agricultural lands are increasingly converted to plantations. As the world seeks to rapidly expand the area of planted and restored forests, clear assessments of forest subsidies are necessary to design policies that achieve desired carbon and biodiversity gains[25].

In this study, we assess the carbon and biodiversity impacts of state-sponsored afforestation by considering one of the world's longest operating afforestation subsidies, Chile's Decree Law 701 (DL 701). In effect, between 1974 and 2012, these subsidies underscored a long-term policy of government support for forest expansion to achieve both environmental and economic objectives. In 2019, the reintroduction of DL 701 was an important topic of environmental debate, serving as the planned implementation mechanism to achieve reforestation commitments outlined in Chile's Intended Nationally Determined Contribution under the 2015 Paris Agreement on climate[26,27]. In addition, the structure of DL 701 has been widely replicated, serving as the model for national policies in Argentina, Ecuador, Colombia, Uruguay and Paraguay and for a variety of international development projects[28,29]. Despite the relevance of the subsidy to national and international policy, no rigorous, comprehensive analyses of the subsidy's environmental impacts have been conducted. Here, we quantify the land use, carbon and biodiversity impacts of DL 701 to guide future national, regional and global afforestation and reforestation policies. Our simulations show that the subsidy increased the area covered by trees but failed to increase carbon storage and reduced native forest extent and biodiversity.

## Chile's afforestation subsidies

For nearly a century, the Chilean government has provided strong and consistent policy support for afforestation. Motivated by both a desire to reduce widespread erosion on marginal agricultural lands[30,31] and an economic policy of import substitution in the wake of the great depression[32], Chilean policymakers passed the first Forest Law in 1931. The law incentivized domestic timber production through tariffs and domestic content requirements, and

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encouraged afforestation through property and inheritance tax exemptions. In the following decades, government steadily claimed a more active role in the forest sector. Under the leadership of Salvador Allende's Unidad Popular, forestry became the most heavily socialized sector of the economy[32].

Although Chile's 1973 coup led to dramatic changes in the structure of forest policies, state support of industrial forestry remained a priority as evidenced by the 1974 adoption of DL 701. This law provided two strong incentives for further afforestation: (1) permanent protection of afforested lands from expropriation and (2) subsidies covering 75% of the cost of afforestation as well as additional support for ongoing plantation management. While DL 701 included some de jure prohibitions on the use of subsidies on lands that were originally forested, de facto protections were limited due to lax enforcement and budgetary limitations[33–35]. Multiple documented cases indicate that forestry companies and forest owners used temporary conversion of native forests to alternate land uses such as agriculture to circumvent the native forest protections of DL 701[33,36,37].

Previous estimates of the impact of the DL 701 subsidies on plantation areas have varied dramatically. The simplest assessments have assumed that every hectare subsidized by the law was planted as a result of the law. Such analyses tend to credit the law with the creation of over a million hectares of plantation forests[38]. However, these results ignore the broader political and socioeconomic conditions that encouraged expansion of forest plantations over the past 40 years. Indeed, some observers have noted that trade reforms alone were a bigger driver of plantation expansion than the subsidies[39]. More rigorous analyses of the impact of the subsidies have compared rates of plantation establishment under the programme to robust counterfactuals, through either quasi-experimental methods or simulation. These analyses have generated more conservative estimates of the impact of afforestation subsidies, ranging from an insignificant effect[40] to modest increases in the area of plantation forestry[41,42].

Even less well understood is the impact of DL 701 on the extent of natural ecosystems. In changing the relative returns to different land uses, plantation subsidies could have multiple important effects on the area of native forests. By increasing the returns to plantation forestry, subsidies could accelerate direct conversion of native forests to plantations. In addition, plantation subsidies could indirectly reduce native forest cover by encouraging plantation establishment on shrublands or marginal agricultural lands where forests might naturally regenerate. Plantation expansion could have further indirect effects on land use allocation by changing the returns to forest products and, as a result, the profitability of native forests and native forest harvesting[43]. Several case studies have documented direct conversion of Chile's native forests and shrubs to plantation forests but these studies do not provide causal evidence linking this conversion to the subsidies of DL 701[23,44–47]. In contrast, Bopp et al. show that subsidized expansion of plantations led smallholder farmers to reduce the area of native forests on their properties[42]. However, the study does not generate landscape-scale estimates of the land use or environmental impacts of DL 701.

## Research design

To assess the biodiversity and carbon impacts of afforestation incentives, we evaluate historic land use patterns in Chile. The forest establishment subsidies of DL 701 have been identified as an important driver of plantation expansion[41] and, as a result, carbon storage[38]. However, local case studies have emphasized the ecological impacts of observed conversion from native forests to plantations[23,44,48,49]. Using detailed maps of Chilean land use change from 1986 to 2011, we assess the scale of changes in both native and plantation forests[23]. To assess the impact of afforestation subsidies, we estimate an econometric model of land use change (Fig. 1)[50–52]. We then rely upon comparisons of a policy baseline to simulated counterfactuals to carefully estimate the scale and uncertainty of the impacts of DL 701 on land use, carbon storage and biodiversity[53]. Specifically, we simulate three historical scenarios: (S1) a 'DL 701 baseline' scenario that represents historical subsidy patterns without enforcement of native forest protections; (S2) a 'no-subsidy counterfactual' scenario with no forest establishment subsidies; and (S3) a 'perfect enforcement counterfactual' scenario that includes subsidies but restricts their availability to lands that are not covered by native forest. We compare simulated land use change in the baseline scenario to the no-subsidy counterfactual to quantify the impact of DL 701; and compare the perfect enforcement counterfactual to the no-subsidy counterfactual to estimate the land use changes that could have been possible with more effective protections for natural ecosystems. The resulting land use outcomes are combined with a carbon bookkeeping model to quantify changes in aboveground carbon storage. Finally, we assess the biodiversity impacts of these land use changes using the 'area-weighted standardized species richness', a new landscape-scale metric of biodiversity.

## Land use impacts

Chilean plantation establishment between 1986 and 2011 was strongly correlated with potential returns to forestry (Table 1). By increasing returns to plantation forestry, Chile's afforestation subsidies increased plantation expansion into native forests, shrub and agricultural lands. Table 2 presents the net changes in land use simulated under our three policy scenarios (for example, net gain in plantation forests) and Table 3 divides these final areas into the specific land use transitions occurring under each scenario (for example, area of native forests converted to plantations). Between 1986 and 2011, the DL 701 baseline scenario experienced $1{,}266 \pm 6 \times 10^3$ ha of plantation expansion into areas formerly covered by native forest ($337 \pm 2 \times 10^3$ ha), shrub ($515 \pm 1 \times 10^3$ ha) and agriculture ($710 \pm 3 \times 10^3$ ha). Given the profitability of plantation forestry, most of this expansion would have occurred in the absence of any subsidies. By comparing the baseline scenario (S1) to the no-subsidy counterfactual (S2), we estimate the amount of additional plantation expansion that occurred as a result of the subsidy. Specifically, the DL 701 baseline scenario resulted in $42 \pm 1 \times 10^3$ additional hectares of plantation forests. On the basis of these simulations, we attribute 3.28% of plantation expansion occurring between 1986 and 2011 to the afforestation subsidies.

Of the $260 \times 10^3$ ha of net native forest loss that occurred between 1986 and 2011, our simulations indicate that $12 \pm 1 \times 10^3$ ha (4.70%) were the result of DL 701 subsidies. Direct conversion of native forest to plantation was the primary contributor to subsidy driven, native forest loss. Although individual forestry companies have acknowledged larger areas of native forest to plantation conversion[36], most of these conversions would probably have occurred without government subsidies. In addition, subsidy driven plantation establishment on shrub and marginal agricultural lands prevented the regeneration of $3 \pm 1 \times 10^3$ ha of native forests.

To understand the ways in which enforcement of restrictions on native forest substitution could have altered land use change dynamics, we modelled a restricted subsidy scenario in which plantation subsidies were only available on non-forested lands (S3). In this scenario, plantations expanded by $1{,}254 \pm 6 \times 10^3$ ha, which is $29 \pm 1 \times 10^3$ more hectares than under the no-subsidy scenario. In marked contrast to the subsidy scenario, the restricted subsidy scenario did not result in additional direct conversion of native forests to plantations. Nevertheless, the restricted subsidy did still generate a small ($2 \pm 1 \times 10^3$ ha) reduction in the total area of forests in 2011 due to reduced forest regeneration on shrub and marginal agricultural lands.

## Carbon impacts

On average, native forests are the most carbon-dense land cover in Chile, followed by plantation forests, shrublands and pastures

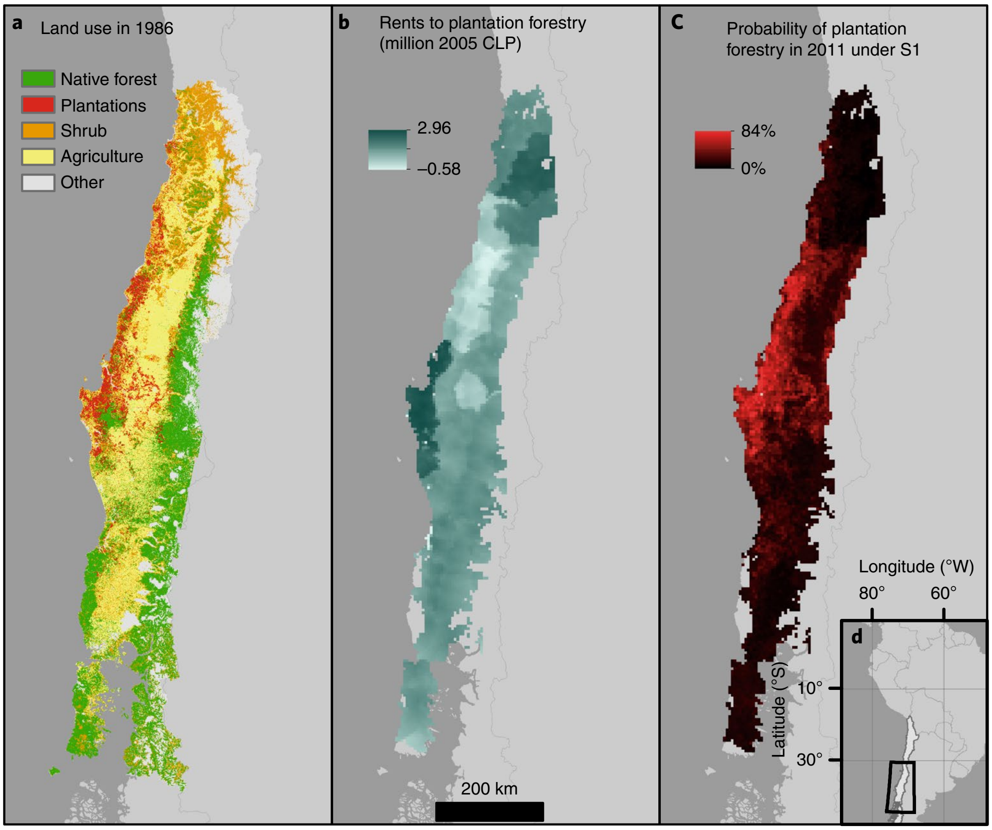


**Fig. 1 | Overview of research approach.** Remotely sensed maps of land use (**a**) were used to identify land use change between 1986 and 2011. Observed land use change was combined with modelled rents to different uses (**b**) to estimate an empirical model of land use change. Using this model, we estimated the probability that each pixel would be dedicated to a specific land use under different policy scenarios (**c**). The study region included portions of Central and Southern Chile (**d**).

(Supplementary Fig. 1). However, these national patterns obscure regional differences. In Chile's central regions such as Valparaíso and O'Higgins, intensively managed plantation forests achieve higher carbon densities than the region's natural sclerophyllous forests. In more southern regions including Los Lagos, Araucanía and Los Ríos, native forests store many times the average aboveground carbon of a typical plantation. For example, the aboveground biomass in temperate Valdivian rainforests of the Los Ríos region contains, on average, 146 $t\,ha^{-1}$ of carbon while aboveground biomass in nearby plantations contains 37 $t\,ha^{-1}$ of carbon when averaged across a plantation's rotation (data sources and calculations described in Supplementary Methods).

Two trends with considerable carbon impacts were evident over the study period: rapid expansion of plantation forests (100.52%) and net losses of native forests (−12.86%). With these two trends counteracting each other, the net change in aboveground carbon storage was relatively modest. Between 1986 and 2011, the carbon stored in aboveground vegetation increased by 5,652 kilotonnes of carbon (ktC) or 1.98%.

Comparison of the DL 701 baseline scenario to the no-subsidy counterfactual provides an indication of the impact of the subsidy on total carbon storage in aboveground biomass. Additional storage from subsidy driven plantation expansion was outweighed by losses in storage in native forests and shrublands. As a result, we estimate that afforestation subsidies decreased total carbon storage in aboveground biomass by $46 \pm 87$ ktC. In contrast, by prohibiting plantation subsidies on previously forested lands, the perfect enforcement counterfactual scenario achieved a $573 \pm 84$ ktC increase in carbon storage. Assuming a modest social cost of carbon of US\$31 per tonne of $CO_2$, we estimate that perfect enforcement of restrictions on native forest conversion could have increased the climate benefits of the subsidy from a loss of US\$$0.39 \times 10^6$ to a gain of US\$$4.84 \times 10^6$.

## Biodiversity impacts

Our simulations indicate that plantation subsidies accelerated biodiversity losses in Chile by encouraging the expansion of plantations into more biodiverse forests. Our meta-analysis of field studies shows that Chilean native forests have $1.323 \pm 0.655$ s.d. higher species richness than paired plantation forests (See Supplementary Methods for more details on the meta-analysis protocol). On the basis of this, we estimate that a randomly selected site in a native forest would contain 14–41% more vascular plant species than a comparable site in a plantation. While these pairwise comparisons help highlight the relative biodiversity in specific land covers, additional aggregation is required to assess subsidy induced biodiversity changes in the broader landscape[54,55]. To consistently summarize landscape-scale biodiversity, we introduce the area-weighted standardized species richness metric, which measures biodiversity as a function of land use shares (see Methods). Under our DL 701 baseline scenario, we find that the area-weighted standardized species richness in 2011 was $-1.017 \pm 0.028$ s.d., which is $-0.043 \pm 0.005$ s.d. lower than in 1986. This can be interpreted as indicating that the mean species richness of a randomly selected sample of points from the landscape would have had 1.017 s.d. lower mean species richness than a comparable sample of native forest points. In contrast, simulations of our no-subsidy counterfactual yield a change in the area-weighted standardized species richness of $-0.042 \pm 0.005$ s.d. Comparing these results, we estimate that DL 701 decreased Chile's area-weighted standardized species richness by $0.0016 \pm 0.00024$ s.d.

**Table 1 | Primary econometric model estimation results**

| Outcome land use | Variable | Starting land use | | | |
|---|---|---|---|---|---|
| | | Native forest | Plantation | Shrub | Agriculture |
| Plantation | Plantation rents (high land quality) | 0.549** (0.238) | −0.0238 (0.142) | 0.529*** (0.167) | 0.400*** (0.136) |
| | Plantation rents (moderate land quality) | 0.599** (0.299) | 0.102 (0.154) | 0.412*** (0.159) | 0.483*** (0.117) |
| | Plantation rents (low land quality) | 0.506*** (0.143) | −0.0724 (0.0790) | 0.111 (0.111) | 0.422*** (0.0862) |
| | Plantation dummy (high land quality) | −3.614*** (0.618) | 1.494*** (0.379) | −3.972*** (0.497) | −1.457*** (0.362) |
| | Plantation dummy (moderate land quality) | −3.541*** (0.758) | 1.676*** (0.398) | −3.611*** (0.440) | −1.205*** (0.339) |
| | Plantation dummy (low land quality) | −2.749*** (0.459) | 1.740***(0.300) | −2.965*** (0.352) | −0.794*** (0.300) |
| | Central | 2.828*** (0.283) | 1.429*** (0.212) | 1.716*** (0.224) | 0.165 (0.216) |
| | South | 1.671*** (0.314) | 1.622*** (0.228) | 0.870*** (0.288) | 0.191 (0.211) |
| Agriculture | Agriculture rents (high land quality) | 0.0541 (0.0448) | 0.111*** (0.0300) | 0.0510*** (0.0192) | 0.109*** (0.0310) |
| | Agriculture rents (moderate land quality) | 0.0397 0.0784) | 0.107*** (0.0367) | −0.0135 (0.0236) | 0.0957** (0.0397) |
| | Agriculture rents (low land quality) | −0.100 (0.0790) | 0.109*** (0.0291) | −0.0944*** (0.0229) | 0.0753** (0.0382) |
| | Agriculture dummy (high land quality) | −1.090*** (0.371) | 0.220 (0.153) | −0.862*** (0.165) | 2.395*** (0.190) |
| | Agriculture dummy (moderate land quality) | −2.124*** (0.485) | −0.190 (0.190) | −1.681*** (0.175) | 1.901*** (0.220) |
| | Agriculture dummy (low land quality) | −2.362*** (0.444) | −1.100*** (0.141) | −2.303*** (0.130) | 1.545*** (0.188) |
| | Central | 1.564*** (0.340) | 0.246* (0.128) | 0.350** (0.140) | −0.562*** (0.166) |
| | South | 1.947*** (0.306) | 1.289*** (0.151) | 0.826*** (0.132) | −0.129 (0.160) |
| Native forest | Native forest rents (high land quality) | −1.097** (0.438) | −0.0581 (0.569) | −1.881*** (0.510) | −1.718*** (0.638) |
| | Native forest rents (moderate land quality) | −0.805* (0.477) | −0.539 (0.529) | −1.327*** (0.437) | −1.244 (0.781) |
| | Native forest rents (low land quality) | −0.290 (0.396) | 0.267 (0.430) | −0.724* (0.376) | −0.676 (0.544) |
| | Native forest dummy (high land quality) | 2.439** (1.062) | −1.524 (1.294) | 1.082 (1.122) | 0.515 (1.510) |
| | Native forest dummy (moderate land quality) | 2.286** (1.154) | 0.000903 (1.231) | 0.306 (1.059) | −0.0136 (1.737) |
| | Native forest dummy (low land quality) | 2.128** (1.022) | −1.310 (1.113) | −0.725 (0.980) | −0.542 (1.446) |
| | Central | 1.089*** (0.228) | 0.244 (0.246) | −0.541** (0.248) | −0.0440 (0.423) |
| | South | 0.855* (0.467) | 1.725*** (0.536) | 0.237 (0.468) | 1.497** (0.713) |

$n = 100{,}948$ points. Comuna-cluster robust standard errors in parentheses. ***$P < 0.01$, **$P < 0.05$, *$P < 0.1$.

Plantation expansion has been a dominant cause of natural ecosystem loss in Chile but only a fraction of these plantations were the result of DL 701 subsidies[23,44]. Our simulations indicate that between 1986 and 2011 DL 701 subsidies alone were responsible for 3.76% of Chile's biodiversity declines related to habitat-loss[56]. However, stronger enforcement of restrictions on the availability of subsidies for previously forested lands could have reduced the subsidy's negative biodiversity impacts. Our perfect enforcement counterfactual simulations experienced $0.0013 \pm 0.00018$ s.d. (78%) less of a decline in the area-weighted standardized species richness than our DL 701 baseline scenario.

## Conclusions

Prominent initiatives such as the Bonn Challenge and the Trillion Trees Initiative hope to address the intertwined challenges of rural poverty, climate change and biodiversity loss through large-scale afforestation and reforestation. Initial national plans indicate that many countries will follow Chile's model for tree cover expansion, relying heavily upon plantation forests to achieve their commitments[8,28,29]. Chile itself has conditioned its international commitment to expand forests by 100,000 ha by 2030 on a renewal of DL 701 subsidies[26]. However, our analysis of DL 701 provides several cautionary lessons to guide the future adoption, design and implementation of afforestation and reforestation policies.

First, planting forests may be more expensive than many advocates hope. Government records indicate that Chile paid US$$408 \times 10^6$ to encourage private landowners to plant $912 \times 10^3$ ha of plantation forests between 1986 and 2011, an average payment of US$447 ha$^{-1}$ (ref. [57]). Careful comparisons to a counterfactual allow us to estimate the share of these plantations that are additional, that is, would not have been established in the absence of subsidies. Consistent with previous simulation models and recent quasi-experimental analyses, we find that only 3.28% of plantations were additional[41], and that the cost for each additional hectare of plantation forest was 30 times higher than the average per-hectare payment made to programme participants. These results reinforce the need to target payments towards actors that would not otherwise reforest their land, rather than dividing payments across all actors who engage in reforestation[58].

Second, reforestation policies will fail to meet their biodiversity and carbon objectives if they do not carefully safeguard existing biodiverse and carbon-rich land covers. Our simulations indicate that, between 1986 and 2011, DL 701 subsidies resulted in the net loss of $12 \pm 1 \times 10^3$ ha of native forests, underscoring concerns raised in

**Table 2 | Simulated changes in land use, biodiversity and carbon storage occurring between 1986 and 2011**

| Outcome | Units | Change between 1986 and 2011 | | | Difference between simulations | |
|---|---|---|---|---|---|---|
| | | DL 701 baseline (S1) | No-subsidy counterfactual (S2) | Perfect enforcement counterfactual (S3) | Impact of subsidy (S1 – S2) | Impact of perfectly enforced subsidy (S3 – S2) |
| Native forest | ×10³ ha | −259.76 ± 3.91 | −247.54 ± 3.89 | −249.21 ± 3.88 | −12.22 ± 0.95 | −1.67 ± 0.91 |
| Plantation | ×10³ ha | 1,266.49 ± 5.89 | 1,224.95 ± 5.76 | 1,254.26 ± 5.83 | 41.54 ± 1.23 | 29.30 ± 1.17 |
| Shrub | ×10³ ha | −395.80 ± 4.66 | −387.23 ± 4.70 | −395.39 ± 4.59 | −8.57 ± 1.15 | −8.16 ± 1.11 |
| Agriculture | ×10³ ha | −610.94 ± 4.55 | −590.18 ± 4.45 | −609.66 ± 4.51 | −20.76 ± 1.19 | −19.48 ± 1.16 |
| Carbon sequestration | ktC | 9,398.73 ± 380.49 | 9,444.90 ± 378.74 | 10,017.70 ± 377.97 | −46.17 ± 86.61 | 572.79 ± 84.25 |
| Biodiversity | Area-weighted standardized species richness | −0.0432 ± 0.00469 | −0.0416 ± 0.00454 | −0.0419 ± 0.00467 | −0.00163 ± 0.000242 | −0.000353 ± 0.00022 |

Each cell presents the mean value and the 95% confidence interval of the mean based on 1,000 simulations of each scenario.

**Table 3 | Comparison of simulated land use transitions, 1986–2011**

| Starting land use | Final land use | Change between 1986 and 2011 | | | Difference between simulations | |
|---|---|---|---|---|---|---|
| | | DL 701 baseline (S1) | No-subsidy counterfactual (S2) | Perfect enforcement counterfactual (S3) | Impact of subsidy (S1 – S2) | Impact of perfectly enforced subsidy (S3 – S2) |
| Native forest | Native forest | 1,350.55 ± 3.03 | 1,360.26 ± 3.01 | 1,360.33 ± 3.03 | −9.70 ± 0.66 | −0.07 ± 0.61 |
| | Plantation | 336.89 ± 1.98 | 324.28 ± 1.94 | 325.03 ± 1.97 | 12.61 ± 0.50 | 0.75 ± 0.44 |
| | Shrub | 298.59 ± 0.90 | 300.02 ± 0.94 | 299.89 ± 0.91 | −1.44 ± 0.46 | −0.14 ± 0.45 |
| | Agriculture | 278.57 ± 1.32 | 280.04 ± 1.33 | 279.36 ± 1.32 | −1.47 ± 0.41 | −0.68 ± 0.40 |
| Plantation | Native forest | 76.07 ± 0.42 | 75.95 ± 0.42 | 76.24 ± 0.42 | 0.12 ± 0.22 | 0.29 ± 0.23 |
| | Plantation | 935.98 ± 0.87 | 936.27 ± 0.86 | 935.94 ± 0.86 | −0.29 ± 0.41 | −0.33 ± 0.41 |
| | Shrub | 110.70 ± 0.44 | 110.63 ± 0.44 | 110.63 ± 0.43 | 0.07 ± 0.27 | −0.00 ± 0.27 |
| | Agriculture | 108.65 ± 0.41 | 108.55 ± 0.42 | 108.59 ± 0.42 | −0.10 ± 0.26 | 0.04 ± 0.26 |
| Shrub | Native forest | 303.20 ± 1.05 | 304.55 ± 1.03 | 303.88 ± 1.03 | −1.35 ± 0.44 | −0.67 ± 0.43 |
| | Plantation | 514.67 ± 1.85 | 506.48 ± 1.84 | 514.54 ± 1.82 | 8.20 ± 0.57 | 8.07 ± 0.57 |
| | Shrub | 1,418.60 ± 2.64 | 1,422.33 ± 2.62 | 1,418.14 ± 2.58 | −3.73 ± 0.74 | −4.19 ± 0.70 |
| | Agriculture | 637.32 ± 1.23 | 640.44 ± 1.22 | 637.23 ± 1.23 | −3.12 ± 0.56 | −3.21 ± 0.54 |
| Agriculture | Native forest | 275.01 ± 1.39 | 276.30 ± 1.40 | 274.93 ± 1.39 | −1.28 ± 0.43 | −1.36 ± 0.42 |
| | Plantation | 710.35 ± 2.80 | 689.33 ± 2.67 | 710.14 ± 2.82 | 21.02 ± 0.71 | 20.82 ± 0.71 |
| | Shrub | 650.12 ± 2.02 | 653.59 ± 2.03 | 649.76 ± 2.03 | −3.47 ± 0.63 | −3.83 ± 0.67 |
| | Agriculture | 3,216.42 ± 2.97 | 3,232.69 ± 2.90 | 3,217.06 ± 2.98 | −16.27 ± 0.88 | −15.63 ± 0.89 |

Each cell presents the mean area (×10³ ha) and the 95% confidence interval of the mean based on 1,000 simulations of each scenario.

local case studies[48]. This was driven primarily through direct conversion of native forests to plantations but also included reductions in forest regeneration on shrub and marginal agricultural lands. The carbon benefit of subsidized plantation expansion was offset by the associated decline in more carbon-dense native forests, resulting in a modest, net decrease (46 ± 87 ktC) in carbon stored in aboveground biomass. In addition, given that plantation forests contain 1.323 ± 0.655 s.d. lower species richness than native forests, we find that the subsidies resulted in an 0.0016 ± 0.00024 s.d. decline in Chile's area-weighted standardized species richness. However, perfectly enforced restrictions on subsidy payments for previously forested lands could have increased carbon storage by 618.97 ktC while mitigating 78% of the subsidy's biodiversity losses. These results emphasize that strong, well-enforced safeguards for natural ecosystems can improve climate and biodiversity benefits of afforestation incentives, while reducing their costs.

## Methods

We quantified the land use, carbon and biodiversity impacts of Chile's afforestation subsidies using an integrated model consisting of four parts: (1) an econometric model quantifying the sensitivity of land use change to the profitability of different land uses; (2) Monte Carlo simulations of 1986–2011 land use change under two policy scenarios and a no-policy counterfactual; (3) a carbon bookkeeping model to quantify the carbon impacts of simulated land use change; and (4) a meta-analysis of biodiversity studies to quantify the impacts of simulated land use change on landscape-scale species richness. We describe each of these components in greater detail below and in the Supplementary Methods. Analysis and production of tables and figures were completed using a combination of Python, R, Stata and ArcGIS Desktop[59–62].

**Study area and unit of observation.** We drew an evenly spaced 1 × 1 km grid of points spanning the Chilean regions (Chile's largest administrative division) between Valparaíso in the north and Los Lagos in the south. Collectively, these regions contain 98% of all of Chile's plantation forests[63]. We removed points that intersected water bodies, urban areas, permanent bare soils (for example, rocky terrain in the Andes) or protected areas. In addition, we removed points that were missing from our land use or land capability class datasets. Land use was not observed in some locations due to cloud cover or errors from the underlying satellite sensors. Land capability classes were not observed in urban areas, military zones and remote portions of the Andes. The resulting sample of 112,217 points represent 112,217 $km^2$ spanning central–southern Chile (Fig. 1c).

**Econometric model of land use change.** Structural economic models of land use change typically consider land use as the revealed preference of a profit-maximizing landowner[64,65]. The fundamental assumption is that each landowner will use their property to generate the highest rents possible. We follow Lubowski et al.[66] and use a discrete-choice multinomial logit specification to model land use changes. To do so, we assume that each plot of land is controlled by a risk-neutral and price-taking landowner[66]. This landowner is assumed to convert her plot of land to a different land use if the net present value of the expected future stream of rents from the new use net of conversion costs exceeds the net present value of the expected future stream of rents for the existing land use. Thus, her decision rule is to select the land use ($k$) that maximizes the expected single-period return at time $t$, net of the expected single-period opportunity cost of converting the land use[50]:

$$\underset{k}{\arg\max}\left(\mathbf{U}_{i,t,j,k}\right),\ \text{where}\ \mathbf{U}_{i,t,j,k}=\boldsymbol{\pi}_{i,t,k}-\delta\mathbf{T}_{i,t,j,k} \quad (1)$$

$\mathbf{U}_{i,t,j,k}$ is the net present value of profits from use $k$ given initial use $j$ for point $i$ at time $t$, $\boldsymbol{\pi}_{i,t,k}$ is the single-period rent generated from use $k$ in point $i$ at time $t$, $\delta$ is the discount rate and $\mathbf{T}_{i,t,j,k}$ is the cost of converting point $i$ at time $t$ from use $j$ to use $k$ and is equal to 0 for all $j = k$. However, we do not perfectly observe each parcel's conversion costs or land rents to different uses. As a result, we further separate $\mathbf{U}_{i,t,j,k}$ into a function of observable and unobservable components:

$$\mathbf{U}_{i,t,j,k}=\mathbf{V}_{i,t,j,k}+\boldsymbol{\varepsilon}_{i,t,j,k}=\boldsymbol{\beta}'_{j,k}X_{i,t}+\boldsymbol{\varepsilon}_{i,t,j,k} \quad (2)$$

where $\mathbf{V}_{i,t,j,k}$ is the observable component, $\boldsymbol{\varepsilon}_{i,t,j,k}$ is the unobservable component, $\boldsymbol{\beta}'_{j,k}$ is an $m \times 1$ vector of parameters varying over initial land use ($j$) and final land use ($k$), $X_{i,t}$ is an $m \times i$ matrix of location ($i$) and time ($t$) specific independent variables. Given $L$ different land uses, the probability that land will be converted from initial use $j$ to use $k$ in period $t$ is $\mathrm{P}\left(\mathbf{U}_{i,t,j,k}\geq\mathbf{U}_{i,t,j,l}\right)\forall l\in L=\mathrm{P}\left(\boldsymbol{\varepsilon}_{i,t,j,k}-\boldsymbol{\varepsilon}_{i,t,j,l}\geq\boldsymbol{\beta}'_{j,k}X_{i,t}-\boldsymbol{\beta}'_{j,l}X_{i,t}\right)\forall l\in L$.

Assuming $\epsilon$ is an independent and identically distributed random variable from an extreme values distribution, we can generate a function of the probability that any point will undergo a specific land use transition[67]:

$$\mathrm{P}(i\ \text{converted from}\ j\ \text{to}\ k\ \text{at time}\ t)=\frac{e^{\boldsymbol{\beta}'_{j,k}X_{i,t}}}{\sum_l \boldsymbol{\beta}'_{j,l}X_{i,t}} \quad (3)$$

We used maximum likelihood estimation to estimate the reduced form coefficients ($\boldsymbol{\beta}'_{j,k}$) of this pooled, discrete-choice multinomial logit model. We clustered all standard errors at the level of the comuna, Chile's smallest administrative subdivision.

*Outcome variable.* We followed the reduced form structure proposed by Lubowski et al. in structuring our design matrix[66]. Each observation in this design matrix represents one possible ending land use (shrubland, native forest, plantation forest or agriculture) for each point ($i$) in each time period ($t$). Our outcome variable is a binary variable indicating the observed choice of land use allocation across the final land use options. Observed land use transitions were derived from maps of land use change from Heilmayr et al.[23]. This dataset uses a combination of supervised classification and spectral change detection of Landsat imagery to identify land use change at 30-m resolution during two periods: 1986–2001 and 2001–2011. Overall accuracies of classified maps are 87% for 1986, 86% for 2001 and 86% for 2011.

*Explanatory variables.* Our matrix of independent variables $X_{i,t}$ represents interactions between five types of data:

(1) $\mathbf{J}_{i,t}$ indicating the observed land use of location $i$ at the start of period $t$, as mapped by Heilmayr et al.[23].
(2) $\mathbf{K}_{u,i,t}$ representing each of the four possible land use options $u$ for point $i$ at the end of period $t$.
(3) $\mathbf{R}_{u,i,t}$ representing the rents to possible land use $u$ at point $i$ during period $t$. We generated spatially explicit maps of rents to plantation forestry, agricultural production and native forest fuelwood harvesting using a combination of Chilean agricultural and forestry statistics. Rents to plantation forestry included the average subsidy payments made to new plantations in each region. The calculation of these rents is described in the Supplementary Methods.
(4) $\mathbf{LCC}_i$ indicating the land capability class of location $i$. High-resolution maps of land capability classes are available for much of Chile[68]. These maps aggregate a variety of variables including slope, aspect, elevation, climate and soil quality into the standard eight-category index ranging from lands that have minimal limitations for agricultural production to those that are unsuitable for any cropping, grazing or forest production. To simplify the model, we recategorized the eight land capability classes into arable lands ($\mathbf{LCC} < 5$), those suited for pasture ($4 < \mathbf{LCC} < 7$) and low-quality lands suitable only for forestry or recreation ($\mathbf{LCC} > 6$).
(5) $\mathbf{G}_i$ indicating the geographic region. Chile's history and latitudinal climatic gradient have led to regional specialization in land use and rural production. To allow for such specialization, we include dummies to differentiate land use trends across three broad regions. The northern region includes the states of Valparaíso, O'Higgins and Santiago, and spans the most populous portion of the country. The central regions include Maule, Ñuble and Bíobío, the historic heart of Chile's forestry industry. Finally, the southern region includes La Araucanía, Los Ríos and Los Lagos, areas with extensive pasturelands and a growing forestry sector.

Summary statistics for each variable in vectors $\mathbf{R}_{k,i,t}$, $\mathbf{LCC}_i$ and $\mathbf{G}_i$ are presented in Supplementary Table 1.

*Econometric model results.* Regression coefficients from our econometric land use model are presented in Table 1. For each of the three rent variables (rents to plantation forestry, agriculture and fuelwood harvesting in native forests), we estimated 12 coefficients representing the sensitivity to rents for each of the 12 possible combinations of four starting land uses and three land quality classes. Each coefficient can be interpreted as measuring the sensitivity of landowners with land in a given starting land use and quality class to allocate their land towards a use in the next time period, in response to the rents to that use. The interpretation of these coefficients as well as robustness tests of these results are discussed in further detail in the Supplementary Methods.

**Simulations of land use change.** *Policy scenarios.* To estimate the impact of Chile's afforestation subsidies, we simulated three policy scenarios:

(S1) Subsidy scenario: this historical baseline scenario sought to most closely represent the actual implementation of the subsidies under DL 701. Plantation rents used in this scenario were identical to the rents that were used to estimate our econometric land use model. The subsidies were assumed to be available on any starting land use.
(S2) No-subsidy scenario: this scenario sought to simulate a counterfactual with no afforestation subsidies. Plantation rents were calculated as the net present value from a forestry rotation without any additional subsidies.
(S3) Restricted subsidy scenario: to explore the potential impact of restrictions on the incentivization of substitution of native forests for plantations, we developed a scenario in which no afforestation subsidies were available to locations covered by native forest. In this scenario, pixels that started a time period in a non-forest class were exposed to the same plantation rent as S1, while pixels that started a time period in the native forest class were exposed to the same plantation rent as S2.

*Monte Carlo simulation.* To predict land use change across the landscape, we developed a simulation model using our econometrically estimated coefficients. The two-period simulation model consists of five steps. (1) We took a random draw of coefficients using the coefficient vector and covariance matrix estimated using our econometric model of land use change. (2) We substituted this draw of coefficients ($\boldsymbol{\beta}_{j,k}$) into equation (3) to create a function predicting the probability that a point will undergo a transition from land use $j$ to land use $k$ as a function of the point's rents to different possible uses. (3) We combined this function with the scenario-specific rents facing each point to predict the probability of each possible land use transition. (4) On the basis of each point's transition probabilities, we randomly assigned points to an intermediate land use class representing their simulated land use in the year 2001. (5) Beginning with the intermediate, 2001 land use class, we repeated steps (2)–(4) to predict transitions occurring during the second time period. This process predicted a final, year-2011 land use class for each of our points, under each of the three scenarios.

To characterize the distribution of predicted land use, we repeated the above process for each of $n = 1{,}000$ iterations. We then compared these distributions under our policy scenarios (S1 and S3) to our no-policy counterfactual (S2) to quantify the impact of each policy on final land use shares (Table 2) and specific land use transitions (Table 3). We describe all outcomes as the mean and 95%

confidence interval of the mean calculated using the estimated standard error of the mean from 1,000 simulations. In the Supplementary Methods we discuss the results of a model validation procedure that demonstrates that our historical baseline scenario (S1) replicates aggregate historical trends in land use change with a high degree of accuracy.

**Estimating carbon impacts.** We developed a carbon bookkeeping model to convert simulated land use into an estimate of total aboveground carbon stored in biomass (AGC). This model relies upon region-specific, average AGC in each land use (native forest, plantation, shrub and agriculture), expressed as $\mathrm{tC\,ha^{-1}}$. Our region- and land-use-specific estimates of AGC are based upon a combination of field studies and forestry models, as described in the Supplementary Methods.

To estimate the total AGC across the entire study area, we multiplied the region- and land-use-specific carbon factors (Supplementary Fig. 1) by the land use shares generated under each of our three scenarios and 1,000 Monte Carlo iterations. This gave estimates and confidence intervals for the total AGC under each scenario. We quantified the net impact of afforestation subsidies on AGC by comparing each policy scenario (S1 and S3) to our no-policy counterfactual (S2).

**Estimating biodiversity impacts.** We estimated the impact of subsidized land use change using a new metric of landscape-scale biodiversity, the area-weighted standardized species richness. Tasser et al. propose the use of the area-weighted mean species richness ($S_m$) as an indicator of the impact of habitat quality on biodiversity[54]. This metric is calculated by taking the average of habitat-specific species richness weighted by the relative proportion of land in each habitat type. Zimmermann et al. apply $S_m$ to monitor the impact of land use change on landscape-scale biodiversity[55]. They note that $S_m$ can be interpreted as the mean $\alpha$ diversity of the landscape. As a result, it presents an intuitive metric to monitor changes in biodiversity resulting from changes in land use. Past efforts to monitor changes in landscape biodiversity made use of a comprehensive database of comparable phytosociological studies. However, such databases are generally unavailable for most parts of the world, including within our study region.

In the absence of systematic, spatially explicit data on Chilean biodiversity, we conducted a meta-analysis to quantify how species richness varies as a function of land use. The meta-analysis protocol is described in detail in the Supplementary Methods. Our meta-analysis aggregates studies with pairwise comparisons of biodiversity in different land uses to yield an estimate of the standardized species richness in each land use type $h$ ($\alpha_h$)[6,69–71]. By converting each study's absolute differences in species richness to a standardized effect size, this meta-analysis enables the integration of studies conducted with different methods and, as a result, can integrate comparisons across a variety of taxa and geographic locations.

To integrate the results of our meta-analysis into $S_m$, we propose a new indicator of landscape-scale biodiversity—the area-weighted standardized species richness ($\alpha_s$). We define $\alpha_s$ as the standardized alpha diversity of each habitat weighted by the relative proportion of land in each habitat type (equation (4)):

$$\alpha_s = \sum_{h=1}^{H} \gamma_h \alpha_h \tag{4}$$

where $\gamma$ is the area proportion of land use $h$ and $H$ is the number of land use types. Whereas $S_m$ is measured in number of species, the unit of measurement of $\alpha_s$ is standard deviations from a baseline land use's mean species richness. The metric $\alpha_s$ can be converted to units of species by determining the species richness distribution for the baseline land use and identifying the species richness at $\alpha_s$ standard deviations from the mean.

In our implementation, we calculated $\alpha_s$ by combining the results of our biodiversity meta-analysis with our land use maps. First, we took a random draw of the parameter vector $\alpha_h$ from the effect sizes generated from our meta-analysis. We then used the results of our Monte Carlo simulation to calculate the final share of area within each habitat type ($\gamma_h$). Then we calculated the corresponding area-weighted standardized species richness using equation (4). We repeated this process for each of our 1,000 iterations and three scenarios. In the calculation of mean effect sizes, we compared all land uses to native forests. As a result, $\alpha_s$ can be interpreted as the mean species richness of the landscape as a whole, compared to the mean species richness of forests. A positive $\alpha_s$ would thus indicate that a randomly selected unit within the landscape would be expected to be $\alpha_s$ standard deviations more diverse than the average forest unit. In contrast, negative $\alpha_s$ values indicate that the landscape as a whole is less diverse than the average forest unit.

**Limitations.** Six primary limitations in the current study highlight specific opportunities for future research. First, our land use model treated land use rents as exogenous. However, subsidized plantation expansion may depress local timber and fuelwood prices, changing returns to native forests and plantations. Previous studies have drawn important insights from land use change models that endogenize such price dynamics[51]. Given the coefficient estimates from our land use model (Table 1), we anticipate that price feedbacks through commodity and land markets would moderate plantation expansion and native forest loss. As a result, our estimated impacts of the subsidy may overestimate plantation expansion and native forest loss. Nevertheless, given Chile's role as a relatively small and open actor in global timber markets, we believe that the scale of these endogenous price effects would be minimal.

Second, our analysis of land use change did not consider the impact of agglomeration economies in driving land use change[72]. Given the Chilean government's widespread support for the timber sector through a wide variety of policies, our emphasis on afforestation subsidies probably underestimates the broader impact that the government has had in creating an economic climate that encouraged timber sector expansion.

Third, we did not have access to parcel-specific data on subsidy eligibility or payments for our study period. Such data would enable the estimation of a property-level econometric model that more rigorously identifies the causal effect of Chile's subsidy programme[73–75]. In light of this data limitation, we chose to implement an econometrically estimated simulation model[5,51,76,77]. This approach provides a carefully constructed counterfactual for our quantification of the policy's land use impacts but relies upon the assumption that the subsidy functions similarly to any equivalently scaled increase in forestry profitability. Future work that uses quasi-experimental methods to empirically quantify the impact of the subsidies would be an important contribution.

Fourth, due to a paucity of land use data in the 1970s and early 1980s, we focus on the period 1986–2011. Since DL 701 subsidies were adopted in 1974, our analysis underestimates the aggregate effects of the subsidy programme over its entire life (1974–2012). It is worth keeping in mind that prior, local case studies have identified widespread native forest to plantation conversion in the 1970s and 1980s[44].

Fifth, although our carbon bookkeeping model provides a reasonably accurate model of changes in aboveground biomass, it relies upon several simplifying assumptions. Since the model doesn't include clear age structures for forest covers, or explicit analysis of management practices in pastures and agriculture, the model will not capture carbon fluxes within each land use over time. Instead, the model focuses on the impact of explicit conversions across land uses. In addition, the model only looks at aboveground biomass. As a result, the model will understate the carbon benefits of land uses with proportionally high levels of soil and root carbon such as grasslands. The model also does not track carbon sequestration in forest and agricultural products. If some biomass products are assumed to be long-term sinks, the sequestration effects of the corresponding land uses would be underestimated by this model.

Finally, our assessment of biodiversity impacts was constrained in several ways due to the limited phytosociological data available. For example, although the spatial configuration of land use is an important determinant of biodiversity, we were unable to explore the effects of configuration on species richness due to the limited variation in landscape configuration across the identified studies. In addition, detailed information on species composition within individual plots would enable more detailed analysis of species turnover across space and land use. These limitations of our biodiversity analysis highlight the potential benefits of central repositories for consistent phytosociological data[78].

## Data availability

All data needed to replicate our results are available on the Harvard Dataverse at https://dataverse.harvard.edu/dataset.xhtml?persistentId=doi:10.7910/DVN/6RDDQH.

## Code availability

All code needed to replicate our results are available on github at https://github.com/rheilmayr/chile_subsidies.

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## Author contributions

R.H., C.E. and E.F.L. conceived of the research. R.H. and C.E. collected data. R.H. conducted analysis. R.H., C.E. and E.F.L. wrote the manuscript.

## Competing interests

The authors declare no competing interests.

## Additional information

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