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THE ECONOMICS OF TROPICAL DEFORESTATION

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ABSTRACT

Two factors have elevated recent academic and policy interest in tropical deforestation: first, the realization that it is a major contributor to climate change; and second, a revolution in satellite-based measurement that has revealed that it is proceeding at a rapid rate. We begin by reviewing the methodological advances that have enabled measurement of forest loss at a fine spatial resolution across the globe. We then develop a simple benchmark model of deforestation based on classic models of natural resource extraction. Extending this approach to incorporate features that characterize deforestation in developing countries---pressure for land use change, significant local and global externalities, weak property rights, and political economy constraints---provides us with a framework for reviewing the fast-growing empirical literature on the economics of deforestation in the tropics. This combination of theory and empirics provides insights not only into the economic drivers and impacts of tropical deforestation but also into policies that may affect its progression. We conclude by identifying areas where more work is needed in this important body of research.

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1 Introduction

Over the 20-year period from 2001–2020, 1.48 million km^2 in the tropics was deforested—an area larger than France, Spain, and Germany combined. The tropics accounted for more than half of all global forest loss over this period, and in recent years tropical forest loss has increased more sharply than in the rest of the world, as shown in Figure 1.[1] Much of the remaining tropical forest is also at risk: due to increasing fragmentation by roads and other infrastructure, half of all tropical forest area is predicted to be within 100m of a forest edge by 2100 (Fischer et al., 2021; Taubert et al., 2018). Already, approximately 10% of remaining moist tropical forest areas were considered degraded in 2020 (Vancutsem et al., 2021).

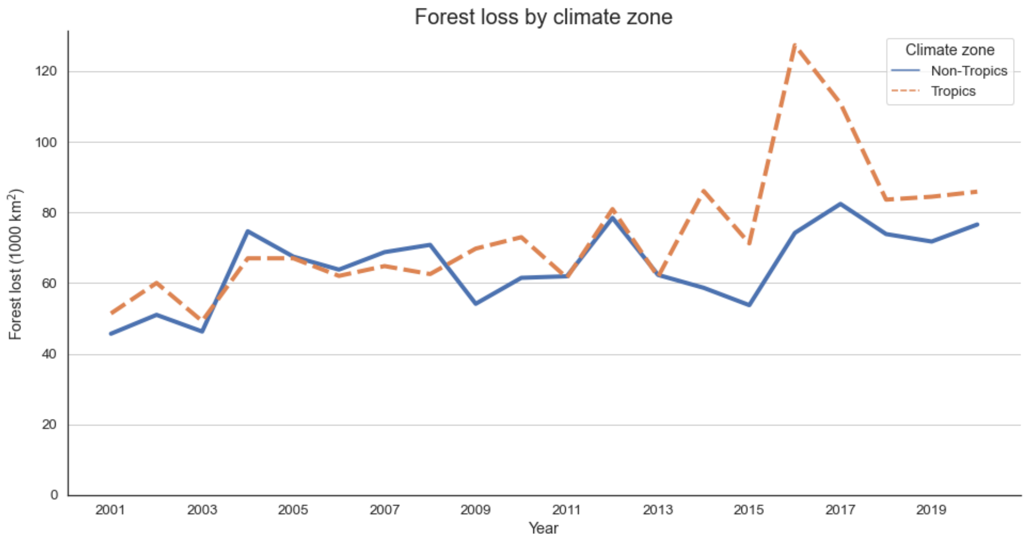
These trends have far-reaching ramifications. Globally, deforestation is a major driver of climate change, contributing an estimated 6–17% of total anthropogenic greenhouse gas emissions (van der Werf et al., 2009), up to two-thirds of which are attributed specifically to tropical humid regions (Achard et al., 2014). Biodiversity loss is also a first-order concern, given that tropical regions support over two-thirds of known species (Bradshaw et al., 2009). Moreover, recent studies have documented tropical deforestation's impact on a wide range of human health outcomes, including increased malaria prevalence (Berazneva and Byker, 2017; MacDonald and Mordecai, 2019; Garg, 2019) and infant mortality due to smoke from forest fires (Jayachandran, 2009).

How should we think about tropical deforestation? Do recent trends in tropical forest loss reflect optimal changes given increased demand for forest products and alternative uses of land? If not, what combination of challenges gives rise to the wedge between actual changes in tropical forest cover and the socially optimal level of forest extraction?

The goal of this article is to review the tools and evidence economics can provide to help answer these questions—and, to the extent that forest extraction is too high, to investigate what can be done about it. To do so, we organize our discussion of tropical deforestation following a set of theoretical frameworks describing natural resource extraction and review the empirical literature relating to each. We hope not only to leave the reader with a picture of where the evidence stands on the drivers of tropical deforestation, but also to illustrate how simple economic models can help understand these drivers, which can in turn highlight policies that may be effective in aligning deforestation decisions with their true social costs.

[1] Hansen et al. (2013) separately captures loss and gain for each pixel, so that pixels may experience both forest loss and forest gain over the study period. However, data on forest gain is available only as a 12-year total for 2001–2012 and therefore cannot be disaggregated by year or combined with loss data for 2013–2020 to cover the full study period. As a result, we report gross rather than net forest loss here.

Panel A: Loss by climate zone



Panel B: Loss by country (tropical regions only)

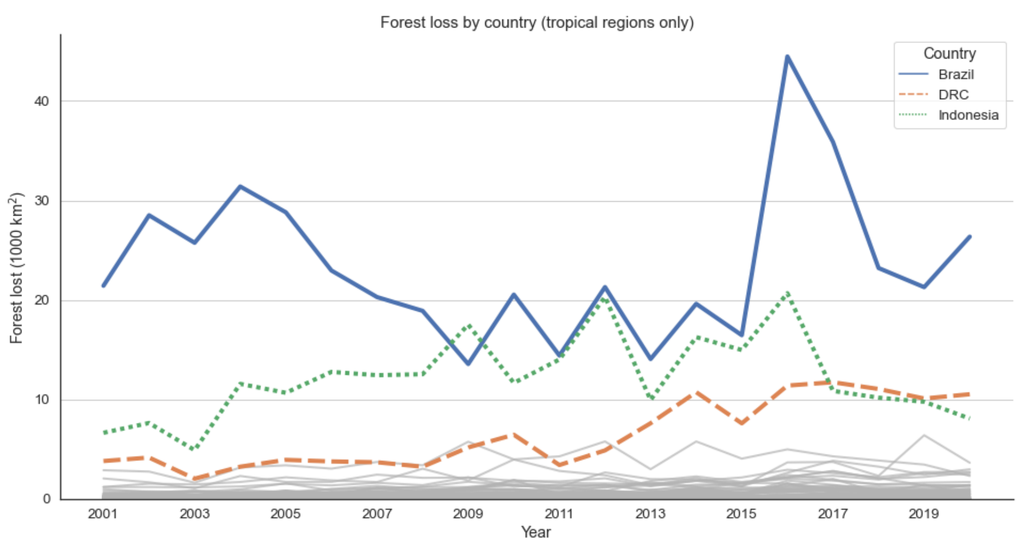


Figure 1. Forest loss by climate zone and by country within the tropics. Data on tree canopy cover in 2000 and gross forest cover loss from 2001–2020 are from Hansen et al. (2013). Forest is defined as 50% tree cover. Loss data indicate binary occurrence of a forest loss event in a given pixel and the year in which the event primarily occurred. We multiply binary forest loss occurrence by 2000 tree cover to calculate the extent of forest loss by year, and then aggregate by climate zones defined inside or outside the region between the Tropics of Cancer and Capricorn.

Much of the early empirical economics research on tropical deforestation—reviewed by Barbier et al. (1991), Kaimowitz and Angelsen (1998) and Barbier and Burgess (2001), among others—centered on cross-country regression analyses. This early wave of papers built an evidence base on how country-level factors including GDP, population growth, agricultural expansion, agricultural productivity, commodity trade, and access to infrastructure related to tropical deforestation levels. This literature also made clear that the non-market existence or conservation value of tropical forests was not being internalized by governments in major forest countries such as Zaïre (now the Democratic Republic of the Congo), Brazil, and Indonesia, where pro-exploitation policies tended to prevail during the 1980s and 1990s.

This "first wave" of studies was followed by a "second wave" that narrowed in on the decision processes of deforesting agents (households and firms), and examined their responsiveness to local economic features such as agricultural prices, road access, and land tenure security (Brown, 1994; Barbier and Burgess, 2001). This newer wave of papers derived insights from economic theory and delved into within-country microdata to better understand how market forces and government policies affect incentives to deforest. This literature also began to consider how to design policies to slow the rate of deforestation, as the threat of climate change and other negative externalities from tropical deforestation became more apparent.

In recent years, what might be labeled a "third wave" of economic research on tropical deforestation has surged. This wave has been triggered by the widespread availability of high-resolution, high-frequency satellite data that has allowed researchers and policymakers to monitor land use—and hence deforestation—across the whole planet, as illustrated in Figure 2. Therefore, we begin our review in Section 2 with a survey of recent developments in measurement and data availability. These developments have revolutionized our understanding of what is happening to the world's tropical forests. They form the basis for new empirical work analyzing the economic drivers of tropical deforestation, with a strong focus on microeconometric causal inference and quantitative modeling of deforestation decisions to estimate policy-relevant structural parameters. Our review focuses on this third wave of research, building on the important insights provided by the first and second waves and prior reviews of these earlier studies.

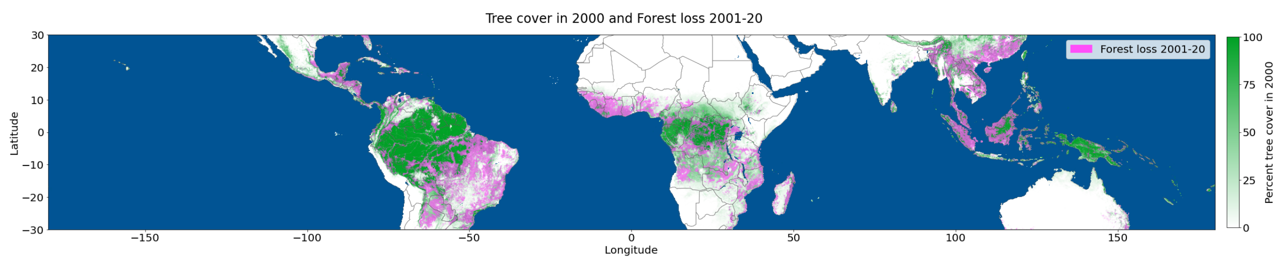


Figure 2. Tree cover in 2000 and forest loss from 2001–2020. Data on tree canopy cover in 2000 and gross forest cover loss from 2001–2020 are from Hansen et al. (2013). Forest is defined as 50% tree cover. Loss data indicate binary occurrence of a forest loss event in a given pixel and the year in which the event primarily occurred. We multiply binary forest loss occurrence by 2000 tree cover to calculate the extent of the loss by year.

We then develop our economic analysis in Section 3 with benchmark models of deforestation with a single forest owner and no externalities. To fix ideas, we begin with the simplest case: a single agent controlling an exhaustible natural resource. Think here of an untouched, 'old growth' forest, managed by a single entity such as a national forest service. Within a reasonable span of time, the forest can be felled only once: it might take hundreds of years to return to old-growth stature (and, factoring in the loss of biodiversity, it may never do so). The decision to deforest a tract of land is thus once and forever. As such, we begin in Section 3.1 with a review of classical models of exhaustible natural resources and draw out their implications for forestry.

Of course, forests *do* regrow. In Section 3.2, we therefore focus on a standard model of forest management with a renewable resource as our benchmark case. It is worth reflecting on this model because it forms the benchmark for traditional forest management policy throughout the world, and it is the primary way many economists thought about forestry before concerns about climate change and other global externalities associated with tropical forest loss emerged (e.g., Samuelson, 1976). In such a model, the primary source of forests' value is the timber they provide, rather than carbon sequestration or other amenity services that have come into focus more recently.

While this basic model of optimal forest management may apply neatly to how a firm like Weyerhaeuser manages its North American forests for paper or plywood production, it misses many of the realities and challenges that are driving tropical forest change in the 21st century. In Section 4, we consider these in turn.

First, in Section 4.1, we discuss the fact that a main driver of deforestation is not the timber being extracted; instead, in many cases, the property owner's value of deforestation comes from an alternative land use such as raising cattle (as in Brazil) or growing oil palm (as in Indonesia). The property owner simultaneously considers the value of timber and the alternative value of the land applied to other uses. We then review the empirical evidence that changes in values of alternative land uses—as a result of changes in market access, prices of agricultural commodities, or increased demand from expanded international trade—are, indeed, an important driver of tropical deforestation.

Compared to when classical forest management theory was developed in the 19th and 20th centuries, perhaps the most important change in economic thinking about deforestation is the widespread recognition that it imposes a sizeable global externality in the form of carbon emissions (in addition to other externalities such as particulate pollution, soil erosion, and loss of biodiversity). It is precisely this externality issue that has made tropical deforestation a major international policy concern. In Section 4.2, therefore, we extend the benchmark model to include these types of externalities and analyze their implications for the socially optimal level of forest extraction.

Once we recognize that tropical deforestation imposes externalities, the natural question is whether conventional approaches to combating externalities can address them. For example, the global REDD+ framework is built around the idea that a donor can reduce deforestation by appropriately compensating forest owners for foregone extraction in a Coasean fashion. However, doing so involves a number of theoretical challenges, from additionality concerns to commitment problems. We therefore review the empirical evidence on whether, in practice, such payments for ecosystem services (PES) approaches are effective in addressing deforestation's externalities.

Third, in Section 4.3, we relax the assumption that there is a single property owner with well defined property rights. Instead, in many forest settings, particularly in low- and middle-income countries, forest rights are less clearly delineated. While in this case classical theory predicts a tragedy of the commons and over-extraction relative to single ownership, we discuss how the empirical literature is surprisingly ambiguous on this point.

Finally, in Section 4.4, we discuss how to extend the benchmark framework to include political economy considerations. Given the importance of deforestation's externalities, as well as direct state ownership of forests in many contexts, the state is actively involved in forest management, both *de jure* and *de facto*. As such, political economy considerations—from 'electoral logging cycles' to Cournot competition between jurisdictions in setting extraction quantities—are often of first-order importance for determining overall deforestation levels.

We conclude by looking forward to key areas of future research needed to better understand what drives tropical deforestation, how it affects human welfare, and which policies can effectively align the incentives of those deciding whether to deforest with the appropriate social costs.

2 Measurement and Data

Remote sensing, now primarily performed via satellite-based observation, has revolutionized our ability to track what is happening to forests at a fine temporal and spatial resolution. Prior to the advent of remote sensing technologies, monitoring forests required on-site human observation.

While this approach to measurement can provide detailed information on local forest conditions, it is prohibitively expensive to implement repeatedly and at scale, especially in tropical regions where state capacity is weak and diverse landscapes make it difficult to achieve adequate sampling densities.

Instead, most at-scale forest measurements now rely on remote sensing techniques. These began with aerial photography in the 1940s and subsequently evolved to satellite imagery beginning in the 1970s. Satellite-based imagery permits broad swaths of land to be scanned in a consistent manner, providing high-frequency, high-resolution data on land usage. The temporal frequency of satellite imagery has enabled large methodological advances in economic modeling of deforestation, leading in particular to the rapid growth of a literature that estimates dynamic discrete-choice models of land use decisions. In addition to revolutionizing academic research on deforestation, such datasets are increasingly being used to aid enforcement activities relating to illegal deforestation: Brazil's Real-Time Deforestation Detection System (DETER) (Assunção et al., 2017) and the Global Forest Watch application (Moffette et al., 2021) are two notable examples.

This section provides a brief overview of current remote sensing techniques, key datasets used in analyses of tropical forests, and a discussion of measurement challenges.

2.1 Principles of remote sensing and measurement

Remote sensing techniques exploit the fact that different materials on the ground absorb, reflect, and emit electromagnetic energy to varying extents and at different wavelengths (Kennedy et al., 2009). Optical sensors on satellites track these differences in reflectivity, and various statistical and machine learning techniques are then used to predict tree cover, deforestation, and even vegetation type as a function of these underlying reflectivity measurements.

In recent decades, new generations of satellites have allowed for higher resolution and sampling frequency. Images collected by Landsat satellites serve as the basis for many forest monitoring products. These images provide consistent global coverage from 1999 onward, with a 16-day revisit cycle and spatial resolution of up to 30 meters (data with less consistent regional coverage back to 1972). Advanced Very High-Resolution Radiometer (AVHRR) sensors provide global daily data dating back to 1979 at a coarser resolution of 1.1 kilometers. Daily global coverage with improved image quality has been available at 250 meter spatial resolution from Moderate Resolution Imaging Spectroradiometer (MODIS) sensors since 2000, and Sentinel 2 satellites have captured 10 meter-resolution global images with a 5-day revisit cycle since 2015. Commercial Earth observation systems such as the Planet constellation, QuickBird, WorldView, and SPOT also provide very high-resolution ($< 10m$) imagery for the monitoring of forest areas.

There are several challenges associated with using remotely sensed data. First, pre-processing of the raw images is necessary to reduce unwanted variation due to solar angle, sensor viewing angle, or atmospheric conditions, and also to accentuate useful features (see, e.g., Hansen and Loveland, 2012). While some algorithms have been developed to do so, it remains challenging to correct for cloud and haze cover, which can exceed 80% on average in some areas (Jain, 2020). Satellites with frequent revisits are often useful in tropical areas to improve the number of cloud-free images.

Second, one needs to classify the observations in each pixel in order to convert a vector of radiances in different spectra into forest cover measurements. While for small areas one can use visual inspection to classify areas as forest or non-forest, to do so at scale, one needs a statistical decision rule, such as from supervised machine learning. These approaches use secondary data for a limited area to train a classification algorithm, which is then used to provide estimates of forest cover and deforestation over the entire area of interest. Validation using a different secondary dataset is often used to check the accuracy of this output (see, e.g., Olofsson et al., 2014). Forest detection, in particular, raises additional challenges. Different definitions of 'forest'—for instance, using minimum vegetation height or land use criteria—can be challenging to measure (Tropek et al., 2014). More subtle changes in forest structure, such as selective logging or fire-induced degradation, are particularly hard to detect without very high spatial and temporal resolution imagery (Gao et al., 2020). The same is also true for reforestation, since it does not produce changes in radiance as quickly as deforestation (Hansen et al., 2013; Garcia and Heilmayr, 2021), as a result of which deforestation is often measured as an irreversible variable.

Recent work in empirical economics has highlighted the importance of understanding the data generating process underpinning remotely sensed data products and considering how it might affect common econometric research designs. At a broad level, detection of deforestation is known to be more accurate in temperate than tropical forests (Hansen et al., 2013) and in more homogeneous landscapes (Mitchard et al., 2015). Recent contributions in this literature have considered how this systematic variation in accuracy, among other issues, may result in non-classical measurement error, and have proposed a variety of potential solutions.

Alix-Garcia and Millimet (2021), for example, compare two commonly used datasets[2] and find that discrepancies between the two sources are correlated with geographic features including slope, elevation, biome and the availability of cloud-free images. They consider biases that may result from the measurement of deforestation as a binary outcome variable and propose a correction. Related work by Garcia and Heilmayr (2021) highlights the potential for bias in common econometric analyses of deforestation using two-way fixed effects regressions with pixel unit fixed effects, given that the measurement of deforestation as a binary, irreversible outcome variable renders it impossible to detect repeated deforestation events in the same location. Torchiana et al. (2020) propose a correction for transition rate estimates based on a hidden Markov model, and find that estimates without this correction for misclassification are severely biased. Avelino et al. (2016) consider how the pixel-based nature of remotely sensed data can bias estimates: when the unit of measurement is smaller than the unit of economic decision-making, attenuation bias is introduced through unnecessary noise in the independent variables, while aggregation may lead to bias when it is too large. Ratledge et al. (2021) highlight that inference may be undermined if machine learning uses the treatment of interest for prediction and discuss possible correction techniques using a 'tailored loss function' at the prediction stage that penalizes bias across the distribution of the remotely-sensed variable. Carleton et al. (2022) review a range of techniques for calibrating the measurement error structure, and argue that multiple imputation can perform effectively at reducing these biases. Jain (2020) reviews a broad range of challenges associated with using satellite data for causal inference in environmental applications.

Advances in radar and lidar technologies, which pass through clouds and yield more detailed imagery, are helping to further improve remote sensing capabilities and address some of the challenges of existing data. These 'active' remote sensing approaches supply their own energy, rather than relying on the sun's illumination, and sample the signal scattered back. These technologies are especially useful for detecting forest degradation and estimating biomass change (Gao et al., 2020; Dupuis et al., 2020), and may be especially well suited to forest monitoring in tropical regions. High quality open-source radar and lidar time series data for tropical countries has recently become available from the Advanced Land Observing Satellite missions, launched in 2006 and 2014; the Sentinel 1 satellites, launched in 2014; and NASA's Global Ecosystem Dynamics Investigation (GEDI), launched in 2018 (Tarazona et al., 2021).

[2] The Global Forest Change product (Hansen et al., 2013) and the Government of Mexico's Land Use and Vegetation Series V dataset (Government of Mexico, 2014).

2.2 Satellite-based datasets

Several remotely sensed datasets, which produce directly usable, pixel-level estimates of deforestation based on satellite data, have been used in economic analyses of deforestation in tropical regions. Perhaps the most commonly used dataset is the Global Forest Change (GFC) product based on Landsat satellite images (Hansen et al., 2013), which provides annual global maps of tree cover, gain, and loss since 2000 at a spatial resolution of 30 meters.[3] Figure 1 uses this dataset to summarize patterns of deforestation, country by country, over the past 20 years. The data reveal the predominance of Brazil, Indonesia and the Democratic Republic of the Congo in driving forest loss. They also reveal dramatic changes in deforestation rates over time, such as the decline in deforestation in Brazil in the mid-2000s and the uptick beginning in the late 2010s.

Song et al. (2018) provide global coverage of deforestation back to 1982 by combining images from Landsat, MODIS, AVHRR and other high resolution sensors. The dataset has a coarser resolution of approximately five kilometers. Importantly, both datasets capture *tree* cover rather than *forest* cover (Tropek et al., 2014), and may be used in combination with secondary data (e.g., Potapov et al., 2017) to measure deforestation specifically. Moreover, these datasets do not capture the important distinction between rotational forestry (i.e., repeated cutting and replanting of trees) and forest clearing for agricultural conversion.

Other datasets focused on tropical forests have also been used in economic studies. For example, Vancutsem et al. (2021) provide data on tropical moist forests at a spatial resolution of 30 meters from 1990 to 2019, using Landsat satellite data and an algorithm that is tailored to local varieties in order to map deforestation and degradation separately. Locally calibrated products are available in some areas, such as in Brazil (e.g., Hargrave and Kis-Katos, 2013; Assunção et al., 2015).

Satellite products are used to capture other forest-related attributes. For example, satellite measurements have also been used to detect the carbon density of tropical forests (Baccini et al., 2012)—an important input for understanding the emissions contribution of tropical deforestation—as well as forest fires, which lead to forest degradation and are an important environmental concern in their own right. Fires can also be detected; Giglio and Justice (2015)'s Thermal Anomalies and Fire Daily dataset builds on MODIS images to report the presence of fires in a one-kilometer grid from 2000 to the present, and has been used, for example, by Balboni et al. (2021b).

3 Benchmark Models of Optimal Resource Depletion

In order to systematically understand the drivers of tropical deforestation, we begin our analysis with classic models of optimal resource extraction. We start in Section 3.1 with a discussion of nonrenewable natural resource depletion, and then turn in Section 3.2 to the workhorse renewable resource models which take into account the fact that forests can, with some lag, regrow.

3.1 Forests as an Exhaustible Resource

The question of how optimally to manage scarce natural resources has occupied economists' attention for nearly a century. Modern research on the economics of natural resource extraction traces its roots to the seminal work of Hotelling (1931), but writing on the topic dates back at least to Faustmann (1849), who studied the optimal harvesting rotation period for trees within a forest owned by a sole manager.

[3]For use cases in economics, see, e.g., Ferraro and Simorangkir (2020); Berazneva and Byker (2017); Carlson et al. (2018); Balboni et al. (2021b); Burgess et al. (2019); and Leijten et al. (2021).

Hotelling's work gave rise to what is now well known as the "Hotelling rule": with costless extraction and perfect knowledge of the total resource stock, the price of a purely non-renewable resource (e.g., a mineral deposit or an old-growth forest) will rise proportionally at the rate of interest. An agent with sole ownership over the resource will extract it such that the final unit is consumed in the same period at which demand falls to zero. Price dynamics in Hotelling's model are independent of demand-side factors, and are purely a result of scarcity and an intertemporal no-arbitrage condition.

Many studies have built on Hotelling's foundational work and discussed the implications of natural resource scarcity. Smith (1968), extending the work of Gordon (1954) and Scott (1955), developed one of the first unified models of firm production from natural resources, encompassing both exhaustible and renewable resources and analyzing cases of private versus common ownership, the latter of which we discuss in more detail below. A flurry of theoretical work on optimal natural resource depletion arose in the late 1960s and 1970s, spurred by contemporary environmental crises (particularly the 1973 oil crisis) and by the famous "Club of Rome" study, *The Limits to Growth* (Meadows et al., 1972). One product of this sharpened attention to natural resource scarcity was a special *Review of Economic Studies* symposium issue on the economics of exhaustible resources in 1974, with contributions including by Solow (1974) on intergenerational equity, Stiglitz (1974) on optimal growth paths, and Dasgupta and Heal (1974) on technological change.

Other studies have attempted to reconcile the Hotelling rule result with data on the realized trajectory of natural resource prices. Although forests differ from the classic non-renewable resource in that forests regrow, there are some settings in which conceptualizing tropical forests as a non-renewable resource may be sensible. For example, 'old-growth' forests, which account for roughly one third of the world's forest land (FAO and UNEP, 2020), may take hundreds of years to regrow—far longer than the planning horizon of a finite-lived agent. They also provide important ecological services by harboring a high concentration of biodiversity, which may be impossible to restore even if trees themselves can regrow. Along these lines, Berck and Bentley (1997) and Livernois et al. (2006) conduct a direct test of the Hotelling rule, both examining old-growth, functionally nonrenewable forests in the Pacific Northwest. The advantage of this empirical setting relative to studies of other natural resources is that scarcity rents—the key object of interest in Hotelling's theory—are directly observable in the form of logging firms' stumpage bids over forest land. In both papers, the authors develop a modified version of Hotelling's theory and, in most of their specifications, fail to reject it.

While Hotelling-style models may be useful for these settings, most recent research has explicitly captured the renewable nature of forest resources, which more accurately reflects the factors influencing deforestation decisions of forest owners in many settings of economic interest.

3.2 Renewable resource depletion under sole ownership

In order to bring our focus more directly to the forces that underlie tropical deforestation today, we consider a simple model of *renewable* resource extraction under sole ownership. This model allows us to highlight the key parameters that influence an owner's deforestation activity and thus suggest specific policy levers that may influence deforestation rates. Much of our exposition follows that of Peterson and Fisher (1977) and Fisher (1981).[4] We present here the key economic insights

[4] Our exposition differs somewhat from the famous "Faustmann result" discussed in depth in Samuelson (1976). Samuelson's analysis primarily describes the problem facing a forest owner who chooses the optimal rotation period T at which to harvest and replant the entire forest. Our focus is instead on a model in which the forest owner depletes the stock of forest by a certain increment in each period and the forest naturally regenerates according to a known growth law. While the two models yield similar insights, we focus on the latter, as it lends itself more naturally to

from this model, especially those that we will modify in Section 4 to capture features relevant to tropical deforestation. A full exposition of the model can be found in Appendix A.

The problem facing a sole owner of forest land is to choose a path of extraction over time that maximizes the net present value of extracted timber, taking into account that the forest will regrow in each period according to a known growth law that may depend on the remaining forest stock. A key insight of renewable resource theory is that the optimal extraction path equates the market price of timber with the marginal cost of extraction plus a correction term that reflects the intertemporal effect of extraction today on the availability of timber in future periods.

One main conclusion from the model is that the economically optimal path of extraction generically does not coincide with the notion of "maximum sustained yield," i.e., the maximum growth rate of the renewable resource that can be sustained in equilibrium. Samuelson (1976) highlights this divergence, which is important because many ecologists and environmental decisionmakers at the time had advocated for the idea of maximum sustained yield in setting policy.[5] Intuitively, economic discounting implies that the owner prefers to cut more trees today rather than to wait for the forest to grow further; the higher the discount rate, the higher the steady-state level of extraction and hence the larger the divergence from maximum sustained yield. In the context of tropical forests, agents' discount rates may be especially high (Barbier et al., 1991), due in part to insecure property rights and regulatory uncertainty (we return to this point below).

4 Beyond the Classical Model: Extensions and Empirical Evidence

The benchmark model described above is useful for deriving general principles of renewable resource management, but it misses several key features that characterize forestry in general and tropical deforestation in particular. First, forestry is heavily land-intensive, and the land on which the forest sits may have lucrative alternative uses. Second, the management of forests in tropical areas is potentially subject to a number of optimization failures that are absent from the classical model. In this section, we first describe theoretical issues surrounding alternative land uses and several empirical approaches that have been developed in the literature to estimate landholders' responsiveness to the value of such uses. Then, we discuss three types of optimization failures that may drive a wedge between realized tropical deforestation and the socially optimal level: unpriced externalities, common-property access regimes, and political economy constraints. Considering each in turn, we consider how the benchmark model of renewable resource extraction might be extended to accommodate them and review the empirical evidence on their importance.

4.1 Modeling land use choice

One feature that distinguishes forests from other types of renewable resources is the opportunity cost of land use. Keeping tropical forests intact necessarily precludes the use of forested land for other purposes, such as for agricultural cultivation or cattle grazing. Note the contrast with the models discussed above: some of the early models of natural resource extraction were motivated primarily by the depletion of non-renewable resources such as oil and mineral deposits (e.g., Hotelling, 1931) or, in the case of renewable resources, by the exploitation of fisheries (e.g., Gordon, 1954). Alternative lucrative uses of land (or ocean) in these settings are unlikely to be of first-order importance for resource owners, and as such, opportunity cost does not typically feature in these

an analysis of the intensive margin of deforestation (i.e., how much forest area is depleted) and will facilitate our discussion of externalities and common-property resources in subsequent sections.

[5] Indeed, Peterson and Fisher (1977) and Goundrey (1960) note that the concept of maximum sustained yield was, at the time of writing, codified in US and Canadian forestry regulations.

models. Even the benchmark forest-regrowth model discussed in Section 3.2 implicitly assumed that all the land would be used for forest; the only question was *when* to harvest the trees relative to their rate of re-growth. By contrast, alternative land uses appear to be first-order when thinking about tropical deforestation because most value accrues to the land owner *after* the trees are cut down.

4.1.1 Theoretical issues

Work in this area often features discrete-choice models of land use in which farmers, taking as given agricultural and timber prices and other exogenous factors such as market access, decide whether and how much forest to clear for agriculture. A notable early example is Pfaff (1999), who was among the first to exploit satellite-based measures of forest loss. Angelsen and Kaimowitz (1999) provide an overview of over 140 early economic models of the drivers of deforestation, including models of alternative land uses. In almost all models they review, an increase in agricultural output prices tends to increase deforestation, all else equal; only under subsistence farming is there potentially an inverse relationship between agricultural prices and deforestation, as higher prices allow farmers to substitute toward leisure activities rather than further harvesting.

Market access. A key determinant of land use decisions is the ease with which timber and agricultural products can be brought to market. Transportation costs lower the net economic returns of different land uses, potentially at different rates—an idea that dates back at least to von Thünen (1826)'s theory of land rent and underlies many modern studies of land use choice.

Openness to international trade is another form of market access that has played a particularly salient role in academic and policy discussions. In addition to trade's effects on timber markets *per se*, Abman and Lundberg (2020) summarize several possible channels by which trade may affect deforestation through agricultural markets. First, trade openness may have an effect through changes in agricultural prices, which alter the value of agricultural land uses relative to forest. Second, trade may reduce the cost of imported agricultural inputs, which increases agricultural productivity; such productivity increases have an ambiguous effect on deforestation for reasons we discuss below. Finally, trade openness may lower the cost of forest-clearing capital ("cheaper chainsaws"), thereby increasing deforestation.

Dynamics. Much early work on land use choice as a driver of deforestation treated the owner's decision as static: given prevailing prices at a particular time, landowners choose the highest-return use for their land. However, land use change is fundamentally a dynamic process, subject to adjustment frictions as well as landowners' expectations over the long-run path of future prices. An early analysis by Albers et al. (1996) considers the option value of conservation when land development is irreversible and future returns are uncertain in a three-period model. Recent methodological advances in the empirical industrial organization literature have provided tools to specify and estimate sophisticated discrete-choice models of agricultural land use, including those that incorporate dynamics over long horizons. Scott (2014) provides an important contribution, developing an Euler equation-based empirical framework that can incorporate unobservable heterogeneity across plots. Using this framework, he illustrates that long-run elasticities of land use with respect to price changes are roughly ten times as large as those estimated from static models.

4.1.2 Empirical approaches

Output prices. Many papers have examined the impact of agricultural output prices on deforestation and related land use. The general empirical idea is to use time series in the national or global price of a relevant agricultural commodity (e.g., cattle, soybeans, palm oil), interacted with some cross-sectional measure of exposure, in order to determine how much those price changes would affect demand in a particular location. For example, Assunção et al. (2015) examine the impacts of crop and cattle prices on deforestation in Brazil and find that deforestation increases with agricultural output prices. Similarly, several analyses have documented that increases in the global price of palm oil led to substantial increases in oil palm cultivation in Indonesia. Edwards (2019) applies a strategy similar to Assunção et al. (2015). Hsiao (2021) takes a more structural approach, modeling palm mill owners' decision making with an intertemporal Euler equation. Because the expansion of oil palm cultivation is a large driver of deforestation in Indonesia (Gaveau et al., 2016; Austin et al., 2019), higher global demand for palm oil increases mill construction and hence deforestation considerably.

Foster and Rosenzweig (2003) focus on demand for timber products as a potential explanation for the observed *increase* in aggregate forest cover in India since 1961. They observe a positive correlation between forest cover and income growth from 1980–1995 in developing countries, but only among those that were relatively closed to trade, as well as a substantial increase in the consumption of forest products. They argue that India was essentially closed to global trade in timber products over this period, and that the observed increase in India's aggregate forest cover was attributable to increases in the value of forest output relative to agriculture.

Market access. Several papers have examined the impact of road infrastructure on deforestation in various settings (e.g., Chomitz and Gray, 1996; Pfaff et al., 2007), finding that higher road density is associated with higher deforestation rates in most cases. In such studies, endogeneity of road placement often presents a challenge to causal interpretation. Asher et al. (2020) employ a variety of microeconometric approaches using panel data to surmount this challenge and find that the construction of new highways in India led to large increases in deforestation in nearby areas, driven by heightened demand for timber, but the construction of smaller rural roads caused only a small increase in deforestation limited to the road construction period.

Souza-Rodrigues (2019) takes an alternative approach based on market access. He uses cross-sectional variation in access to the Brazilian transportation network to proxy for the returns to agriculture in a von Thünen-like structural model. After controlling for input prices and observable land characteristics, market access is akin to a price shifter: the return that farmers receive for agricultural output is the market price net of input and transportation costs. He shows that greater road access leads to more deforestation and farming, which he interprets through his model as a responsiveness to net agricultrual prices received. Observed deforestation thus allows him to back out farmers' value of agricultural (i.e., deforested) land. Having estimated the structural parameters of landowners' decision making in his model, he then simulates the deforestation effects of counterfactual policies, including conservation subsidies and taxes on agricultural land.

Direct empirical studies of the effect of trade openness on deforestation are relatively scarce, in part because rigorous causal identification at the country level has proven difficult. Ferreira (2004) performs a cross-sectional, cross-country analysis and finds some evidence that lower trade barriers may have increased deforestation from 1990–2000, but only when interacted with baseline measures of institutional strength. Ferreira's results suggest that the effects of trade through timber markets may be mediated through a country's property rights regime, a topic to which we return in more detail in Section 4.3.

Abman and Lundberg (2020) surmount the difficulties posed by the endogeneity of trade openness by exploiting the staggered and idiosyncratic timing of regional trade agreements (RTAs). They estimate event study regressions, using a panel of 189 countries spanning 2001–2012, and find that deforestation rates cumulatively increased by between 19-26% over the three years following ratification of an RTA, driven almost entirely by tropical developing countries. Moreover, they demonstrate that agricultural land conversion increased in the years following an RTA while timber output remained constant, providing evidence that agricultural trade rather than trade in timber *per se* was the primary driver of deforestation increases.

Productivity. While most models predict that increases in output prices should lead to increases in deforestation, it is less clear *ex ante* how changes in agricultural productivity will affect deforestation. The so-called "Borlaug hypothesis" states that agricultural technological improvement can decrease deforestation by reducing the total land area needed for agricultural production (Borlaug, 2007). But of course, this claim is not obvious: if demand is very elastic, productivity improvements could lead to more deforestation, not less. Angelsen and Kaimowitz (2001) present a series of case studies from countries around the world and suggest that, while the Borlaug hypothesis plausibly holds at the global level, it is less clear that it is relevant at the level of regions or specific agricultural products.

One recent study looking at this question is Szerman et al. (2022), who use electrification in Brazil as an instrument for agricultural productivity. They show that electrification increased productivity in agriculture – for example, by enabling temperature-and-humidity controlled storage facilities and the use of electrical pumps for irrigation – but had little impact on livestock productivity, which led farmers to switch from livestock to crops. Because livestock is much more land-intensive than crops, deforestation ultimately declined in some of their measures. Similarly, Abman et al. (2020) find that the introduction of high-yield variety seeds and agricultural training by an NGO in Uganda reduced deforestation, and Abman and Carney (2020) find that ethnic patronage in a fertilizer subsidy program in Malawi led some areas to receive more fertilizer, which subsequently reduced deforestation levels in those areas. On the other hand, Hess et al. (2021) experimentally evaluate a community-driven development program in the Gambia and find that, in treated communities that spent the grant on infrastructure and agricultural inputs, deforestation increased by roughly 12%, potentially due to income-driven changes in deforestation behavior. Similarly, Carreira et al. (2022) find that the introduction of genetically engineered soy seeds led to increased deforestation via cropland expansion in the Brazilian Amazon 2000–2017.

Credit constraints and cash transfers. In the presence of credit constraints, switching costs between land uses may be particularly salient: even if an alternative land use is more profitable, landowners may not be able to borrow against this future value in order to pay the static cost of switching today. Assunção et al. (2020) illustrate the importance of credit constraints for agricultural land conversion in the Brazilian Amazon: they use a difference-in-differences design to study the impact of a 2008 policy change that tightened access to agricultural credit by requiring credit recipients to comply with land titling requirements and environmental regulations. They show that the policy change reduced the cumulative deforested land by up to 60% over a nine-year period, although it is important to note that this policy explicitly bundled credit with a set of environmental compliance requirements.

Ferraro and Simorangkir (2020) present evidence from the rollout of a conditional cash transfer program in Indonesia that deforestation is indeed an important source of liquidity for otherwise credit-constrained households: the cash transfer program reduced village-level forest loss by about

30%, driven mainly by periods of negative rainfall shocks when agricultural income was otherwise low. On the other hand, cash transfer programs may induce deforestation effects through income-driven consumption changes. For example, Alix-Garcia et al. (2013) study the rollout of the *Oportunidades* conditional cash transfer program in Mexico using a regression discontinuity design around the locality-level poverty threshold for eligibility. In localities that were eligible for the transfer, households began consuming more milk and meat, which the authors argue drove up the return to cattle cultivation and hence increased deforestation. These countervailing effects suggest that the link between local incomes and deforestation depends on whether the forest provides an alternative, less attractive income source (as in the Indonesian case) or whether it is a source of supplies of local goods whose demand increases with local incomes.

Value of conservation. These approaches focus on the changing value of the land's alternative uses. However, in a context where there may be future payments for conservation (as discussed in Section 4.2 below) and land use investments may be irreversible, the option value of not deforesting—i.e., of holding the land as forest which can itself generate potential future returns—is of first-order importance for landowners' decision making. Stavins (1999) is among the first to discuss these concerns systematically, developing a revealed preference-based discrete-choice approach to estimate heterogeneous costs of carbon sequestration with a county-level panel of land use choices in the American South.

Araujo et al. (2020) consider these issues in a dynamic discrete-choice model of land use in the Brazilian Amazon. In their model, there is a fixed conversion cost from one land use type to another, and the authors apply a dynamic Euler-equation approach that incorporates the costs of adjusting from forested to deforested land and vice versa. They use both cross-sectional differences in the returns to cropping and pasturing, given by land use differences and transportation costs, the value of existing forest stocks, as well as time-series differences in prices of cropland and pastures to estimate the model.

It is worth noting that these discrete-choice models of deforestation are typically formulated differently from the optimal control approach used in the models discussed Section 3 and Appendix A; instead, discrete-choice models often specify a Bellman equation to reflect the dynamic nature of the landowner's decision-making process. This discrete-choice formulation does not allow for analytically calculating an optimum as in the model of Section 3.2, but on the other hand allows researchers to explicitly model the types of switching costs that would be hard to incorporate into general optimal control models, as well as to estimate structural parameters of landowners' profit functions.

Forest-wide complementarities. One particular area in which the dynamic discrete-choice framework is likely to be unwieldy, however, is in situations where deforestation decisions in one part of the forest affect growth rates and other payoff-relevant parameters in other land parcels. Such spatial complementarities are integral in the analysis of "tipping points", as discussed by Franklin and Pindyck (2018). A growing body of ecological research highlights the possibility that, after aggregate deforestation levels cross a certain threshold, the entire tropical forest ecosystem may be so disrupted as to enter a functionally irreversible transition to open savanna. Additionally, agents' value of conserving a given parcel of land may depend on the conservation status of neighboring parcels (Albers et al., 2008) which, as noted by Scott (2014), have not yet been tackled in the agricultural discrete-choice literature.

Having considered the optimization problem facing a single agent with sole property rights over a tract of forest land, the question remains as to whether a social planner might want to further decrease the amount of deforestation in equilibrium. In particular, one reason a social planner might further decrease deforestation is that it entails meaningful social costs that are external to the agents engaged in forestry. We next turn to consider these externalities in detail.

4.2 Deforestation externalities

The negative externalities of tropical deforestation are substantial and global in scope, as discussed in Section 1. Such externalities, particularly as they relate to carbon emissions and climate change, were not well documented empirically when the classic theory of natural resource extraction was developed in the mid-20th century. Another key distinction between tropical forests and other natural resources is that forests provide amenity services in addition to extractive benefits.[6] Krutilla (1967) observes that the traditional economic rationale for conservation does not directly address this "pure" amenity value, nor do markets make adequate provisions for it. He further argues that forest reserves have "serendipity value:" they sustain important genetic resources and biodiversity that give rise to welfare-improving scientific discovery and which a single owner would not internalize.[7] These observations all point to the conclusion that deforestation may carry large costs that are external to the agents who possess rights over the forested land, and that the socially optimal level of deforestation may be much lower than the single-agent solution derived above.

4.2.1 Theoretical issues

In Appendix B, we extend our benchmark model of optimal renewable resource extraction to incorporate deforestation externalities. We capture such externalities by introducing a level shift in the static costs of forest extraction, and illustrate that the socially optimal level of deforestation in steady-state falls below that of the sole owner's optimum.

Policy instruments. Economic theory proposes several levers by which the negative externalities of deforestation may be corrected. One suggestion that arises from canonical theories of externalities is a Pigouvian tax on deforestation activity or, relatedly, on agricultural land when the latter is the primary alternative to forest conservation. Levying a Pigouvian tax on deforestation is challenging, however: an implementing government needs to identify the owner of the land, measure incremental deforestation, and then actually collect the tax. In the countries where many tropical forests are located, credit constraints, limited state capacity, and unclear land ownership can complicate these processes substantially.

A much more common policy suggestion is the converse: subsidies for *not* deforesting, often labeled payments for ecosystem services (also known as payments for environmental services, or PES). These types of payments are the cornerstone of the REDD+ (reducing emissions from deforestation and forest degradation) strategy as part of the UN-sponsored global climate change frameworks. One reason why PES subsidies are more common than Pigouvian taxes may be that they are politically more patalable, and do not face the same collection difficulties. However, PES programs entail the challenge of identifying marginal landowners—those who would not conserve the forest

[6] Samuelson (1976) notes that the presence of large externalities in forestry could provide justification for the optimal forest rotation period being closer to the "foresters' optimum" concept of maximum sustained yield, and speculates that the consumption value of forest may vastly outweigh its existence value at low levels of income (which can be seen as an early precursor of the Environmental Kuznets Curve hypothesis).

[7] A distinct body of research has developed attempting to monetize the value of the various services that tropical forests provide. Doing so, however, presents a host of theoretical and methodological issues outside the scope of our review. Carson (2012) presents a concise overview of contingent valuation methods for non-market services. For one recent meta-analysis of papers valuing the services provided by the Brazilian Amazon, see Brouwer et al. (2022).

in the absence of payments. We review empirical evidence surrounding this design challenge (also termed "additionality") below. Jayachandran (2013) studies another theoretical difficulty, arguing that when landowners are credit-constrained and timber is valuable, their opportunity cost of conservation is especially high because they can deforest in order to cover unexpected lump-sum expenses, which a stream of smaller PES payments could not cover. As a result, these landowners may refuse to opt in to PES contracts, even if the overall net present value of the PES transfers is higher than the net present value of the opportunity costs of maintaining forested land.

Harstad (2016) develops a dynamic game theoretic model of the market for so-called "conservation goods" (such as tropical forests) which helps explain the prevalence and structure of PES-like contracts, including across countries. The owner of forest property values consumption of the forest less than an outside party values its conservation, but Harstad shows that within a broad class of equilibrium concepts, there exists no pure strategy equilibrium in which conservation is achieved. His model predicts that forest "rental" contracts, rather than purchases, will be common in situations in which monitoring and protection of the forest is costly after sale, which reflects the actual structure of many REDD+ contracts between developed and developing countries.

Trade policy. Trade policy has been considered as an alternative tool for correcting international deforestation externalities. In early theoretical work, Barbier and Rauscher (1994) develop a model in which timber is extracted for domestic consumption or export. Importing countries can impose tariffs or import bans in order to increase the equilibrium stock of intact tropical forest, but such policies may, under certain circumstances, increase deforestation. By contrast, international transfers such as PES unambiguously increase the stock of conserved forest. In recent work, Harstad (2022) develops a model that formalizes the conditions under which trade agreements between developed and developing countries can reduce the level of tropical deforestation among exporters. His main insight is that a "contingent trade agreement"—whereby the timber-importing North sets tariffs as a function of deforestation levels—can reverse the negative relationship between free trade and deforestation. The scope of contingent trade agreements to reduce deforestation is limited by the fact that tariffs must be renegotiation-proof; if the timber-exporting South's tariffs on goods imported from the North are also allowed to be contingent on conservation levels, then greater reductions in tropical deforestation can be achieved.

4.2.2 Empirical approaches

Payments for environmental services. As described above, PES have emerged since the 1990s as a common policy suggestion for the prevention of tropical deforestation. These policies carry a Coasean flavor, as landholding agents receive compensation in exchange for not engaging in a behavior that imposes negative externalities on the "donor." While some early empirical evidence on the effectiveness of PES was mixed—see, e.g., Pattanayak et al. (2010) for a review—some recent evaluations have shown more promising results. In particular, Jayachandran et al. (2017) conduct a randomized evaluation of PES contracts among 121 villages in Uganda. In the 60 villages randomly assigned to treatment, enrollees receive approximately \$28 USD per hectare of forest conserved annually over a two-year study period. Despite the fact that only 32% of eligible forest owners enrolled in the program, 88% of those who did enroll ultimately complied with the conservation requirement. As a result, the PES contracts reduced deforestation in treatment villages from 9.1% to 4.2% of baseline forest cover. Moreover, the PES contracts did not induce "leakage" of deforestation activity to neighboring areas not covered by the contracts.

On the other hand, Edwards et al. (2022) provide less sanguine evidence from a randomized evaluation of a village-level PES program that paid villages in Indonesia conditional on experiencing zero forest fires in the 2018 fire season. The authors find that the program mobilized village-level efforts to prevent forest fires, but ultimately did not have any detectable effect on fire incidence. The authors hypothesize that the program was ineffective because the size of the transfer may not have been large enough and because fire prevention efforts suffered from a collective action problem. These results point to key considerations for the design of PES contracts: the amount of the payment—which must accurately reflect the landowner's opportunity cost of conservation—and the identity of the payment recipient. A recent evaluation by Wong et al. (2022), however, suggests that community-level PES programs may be effective in some contexts: they illustrate that the Bolsa Verde cash transfer program in Brazil, which incentivized rural communities to maintain at least 80 percent forest cover, reduced deforestation in treated areas, driven by recipients' increased reporting of illegal deforestation activity by others.

Even if successful, PES programs may be expensive to implement, especially given additionality concerns. As highlighted by Jayachandran (2022), PES transfers can be decomposed into the component that compensates landowners for this compliance cost and the component that is a pure transfer, and both may be substantial. Souza-Rodrigues (2019)'s structural estimates of the landowners' "demand for deforestation" can be interpreted as estimates of this compliance cost. In part because of heterogeneity in agricultural productivity across land parcels, he finds that achieving the Brazilian government's stated policy goal of 80% forest cover in each parcel through PES subsidies would require immense government expenditures (roughly 1.5% of the Brazilian federal budget annually). Moreover, the less additionality among enrolled parcels, the greater the pure transfer cost.

The pure transfer component of PES contracts may nonetheless be valuable as a tool for alleviating poverty, as evaluated by Sims and Alix-Garcia (2017), among others. Alix-Garcia et al. (2015) highlight, however, that the extent to which PES programs can achieve anti-poverty and conservation aims simultaneously depends on the correlation between a land parcel's deforestation risk—i.e., the observable characteristics of the land that make it suitable for agricultural conversion—and the wealth of the landowner.[8] In the setting of a Mexican PES program, the authors show that deforestation risk is *positively* correlated with wealth, inducing a tradeoff between the additionality of PES contracts and the extent to which they reduce poverty.

The optimal design of PES contracts in the face of such heterogeneity remains an active area of research. Mason and Plantinga (2013), studying the design of carbon offsets, illustrate theoretically that a fiscally optimal scheme would offer landowners an incentive-compatible menu of two-part contracts on the amount of land held as forest and the amount transferred to the landowner. Moreover, because landowners' compliance costs fluctuate over time in response to changing agricultural output prices, Assunção et al. (2015) highlight that PES contracts should (but currently typically do not) respond to agricultural prices. Jack and Jayachandran (2019) demonstrate that introducing small "hassle" costs of enrollment in PES contracts may improve the cost-effectiveness of such contracts if enrollment costs are correlated with the landowner's status quo likelihood of conservation. Similarly, Jack (2013) illustrates that a tree-planting subsidy program in Malawi was cheaper per surviving tree when contracts were allocated through an auction rather than by random assignment. Additional rigorous evidence on whether PES programs work, and how to design them to be more cost-effective, is a useful dimension for future research.

Protected areas. Quantity restrictions on deforestation, typically in the form of protected areas in which all logging is made illegal, present an alternative policy instrument that is less infor-

[8] An earlier, more general framework for evaluating tradeoffs in the targeting of land conservation policies was put forth by Babcock et al. (1996).

mationally intensive than Pigouvian taxation. Indeed, protected areas are extremely common in practice: more than 16% of the Earth's land is covered by a protected area (UNEP-WCMC and IUCN, 2020). However, similar additionality issues arise when evaluating the effectiveness of protected areas: it is crucial to know whether protected land would have actually been deforested in the absence of the restriction. Andam et al. (2008) note that the intuitively appealing approach of comparing deforestation rates in protected areas to surrounding unprotected lands—such as via a regression discontinuity design—may provide biased estimates if establishing a protected area induces "leakage" of deforestation activity to nearby areas. They instead employ a matching approach, leveraging detailed microdata from Costa Rica to compare protected areas to observably similar tracts between 1960–1997. They find that accounting for observable differences substantially attenuates (but does not eliminate) the estimated effect of protected areas on deforestation rates. Börner et al. (2020) provides a review of forest conservation policy and corroborates that the effects of protected areas estimated in the literature are generally modest.

Coasean bargaining. Coase (1960)'s theory of social cost suggests that the socially optimal level of deforestation might also be attained, in the absence of significant transaction costs, through a system of transfers between agents. Dasgupta (1996) highlights, however, that in most developing country settings, Coasean solutions to deforestation may be unlikely to emerge given that common law institutions often favor polluters' rights by default, so that those harmed by deforestation—who are likely to be economically vulnerable—must compensate those engaged in deforestation not to do so. Balboni et al. (2021b), examining the strategic illegal use of fire for land clearing in Indonesia's forest estate, provide a modern example of a setting with suggestive evidence for Coasean arrangements among private firms. Using MODIS satellite data to identify fires, the authors document that strategic fire-setting is less likely to occur on days when the weather is more conducive to fire spread in pixels that are surrounded by the landowner's own land. Their results suggest, however, that this tendency is attenuated when the surrounding area consists of land in a single other concession, raising the prospect that firms may treat risks to nearby concessions similarly to risks to their own land when transaction costs are low, consistent with Coasean bargaining between firms.

Trade policy. Hsiao (2021) quantitatively explores the conditions under which import tariffs can function as a Pigouvian tax, and hence lower deforestation abroad. The conditions he identifies are that a) importing countries coordinate on their tariffs, and b) these importers can commit to upholding tariffs even when doing so is not statically optimal. He develops a dynamic model that allows for palm oil producers to be forward-looking in their decisions to construct oil palm mills and over how much land to deforest. Incorporating dynamics into the model is important because imposing tariffs on exports is sub-optimal in a purely static framework once palm oil manufacturers have already engaged in deforestation and the associated carbon emissions are sunk. Estimation of the model suggests that coordinated and committed tariffs can nearly replicate the deforestation reductions achieved by a first-best domestic tax. Dominguez-Iino (2021) examines deforestation in South America driven by demand for agricultural products, estimating a structural model in which farmers choose both how much land to deforest and which products they produce on deforested land. On the demand side, he models an agricultural supply chain in which monopsonistic intermediaries purchase from farmers and sell to consumers. Akin to Hsiao's finding, he concludes that tariffs would largely be ineffective due to leakage of trade to countries without regulation. Moreover, such trade interventions may be regressive because poorer regions feature less elastic supply of agricultural products and hence bear greater incidence of the tariff.

Non-state interventions. Non-state actors such as firms and non-governmental organizations may also undertake measures to stem deforestation. A recent and growing empirical literature examines the effects of such non-state commitments, but finds relatively limited effects on forest loss. For example, Alix-Garcia and Gibbs (2017) evaluate the impacts of "zero-deforestation cattle agreements" among meatpacking companies in Brazil from 2007 to 2015 not to purchase cattle from properties that had deforested above legal limits. The authors employ a difference-in-differences strategy exploiting the staggered rollout of such agreements and find that, despite widespread adoption, they had no aggregate effects on deforestation in their sample, potentially as a result of "leakage" of deforestation activity to unmonitored properties. Similarly, Blackman et al. (2018) use a matched difference-in-differences design to study the deforestation impacts of Forest Stewardship Council certification in Mexico and find no significant effects. One notable exception to these null results is Heilmayr et al. (2020), who evaluate the impact of the Amazon Soy Moratorium (ASM) in Brazil. As part of this agreement, grain traders—who accounted for 90% of purchases of soy produced in the Brazilian Amazon—committed not to purchase soy grown on deforested land. The authors provide evidence that the Moratorium effectively reduced deforestation and did not lead to leakage. In light of these mixed results, identifying conditions under which such non-state deforestation commitments may effectively curb deforestation activity remains a potentially fertile area for future research.

4.3 Tropical forests as common-property resources

The results derived in Section 3 rely on the assumption of sole, well defined ownership of forest property. While this assumption may be reasonable in some settings, it is more tenuous when considering tropical forests, where property rights are often imperfectly defined and enforced (Amacher et al., 2009, Araujo et al., 2009). In this section, we derive predictions for the pattern of deforestation when forests are treated as common-property resources and discuss solutions to common-property issues that have been proposed in theory and studied empirically.

4.3.1 Theoretical issues

A large body of theoretical work emphasizes that when property rights over natural resources are weak or nonexistent, these resources will tend to be over-exploited relative to the single-agent optimum discussed in Section 3. Hardin (1968) famously labeled this phenomenon the "tragedy of the commons;" Gordon (1954)'s foundational work formalizes this conclusion in a model of fisheries. In our setting, consider a stylized example consisting of a continuum of forest parcels, each with its own level of fertility, and a continuum of identical potential entrants into forestry. The planner's optimal allocation of firms is such that the marginal yield to effort in each forest parcel is equalized. Under a common property regime, however, this allocation is not an equilibrium, because any individual firm could do better by instead moving to a parcel with higher *average* yield. When agents cannot expect to appropriate profits in any future period, profits in each time period are thus competed down to zero. In Appendix C, we illustrate mathematically and graphically how the free-entry equilibrium leads to over-exploitation relative to the sole-ownership benchmark.

Weakly enforced property rights raise a distinct but related set of theoretical issues with nuanced implications for the design of policy.[9] Mendelsohn (1994) provides two early models of the link between insecure land tenure and deforestation in developing countries, emphasizing that weak property rights can discourage sustainable management of resources, either because doing so can

[9] Although many of the papers discussed here take the strength of property rights as given, property rights regimes are, of course, not set exogenously, which also has implications for forests (Copeland and Taylor, 2009).

be a way of securing ownership rights (Angelsen, 1999) or because the probability of eviction leads squatters to exploit land for shorter-term "destructive" uses. Bohn and Deacon (2000) develop a model of investment and natural resource use under different levels of ownership risk. Their model predicts that for resources such as tropical forests, whose extraction is no more capital-intensive than general production, weaker ownership rights will result in greater deforestation.

Policy instruments. The discussion above suggests that assigning clearer property rights is likely to move extraction closer to privately optimal levels. Such approaches are discussed in the early work of Gordon (1954) and Scott (1955) and synthesized (and critiqued) by Ostrom (1990). If there are externalities associated with extraction, however, assigning property rights may not be a panacea. Although reductions in deforestation through secure property rights are a typical feature of renewable resource models, a relatively smaller set of models focuses on the possibility of offsetting "investment effects," whereby land security increases agricultural investment and hence the productivity of agricultural (but not forest) land. If the magnitude of this investment effect outweighs the conservation effect of increased land security, deforestation may increase (Liscow, 2013), potentially exceeding the 'optimal' level in the presence of externalities. Other authors have derived theoretical results highlighting potential adverse political economy and redistributive implications of conversion from common to private property. For instance, Weitzman (1974) shows that a variable factor (e.g., labor) without property rights will be weakly worse off under private ownership, and Dasgupta (1996) cautions that "the privatization of village commons and forest lands, while hallowed at the altar of economic efficiency, can have disastrous distributional consequences, disenfranchising entire classes of people from economic citizenship."

Trade openness. Weak property rights feature centrally in the literature examining the effect of trade openness on natural resource extraction in developing countries. In particular, differences in property rights regimes across countries may be interpreted as a potential source of comparative advantage in the production of natural resource products (i.e, timber), so that an increase in trade openness may exacerbate (or ameliorate) the over-exploitation of forest resources that theory predicts under an open-access regime. Copeland et al. (2022) review the theoretical literature on trade openness and renewable resource depletion, and empirical applications outside of the forestry setting. We focus here on particular features that are relevant for understanding trade's influence on tropical deforestation.

The canonical model marrying Ricardian trade theory with the theory of optimal renewable resource extraction is developed in Brander and Taylor (1997) and several subsequent papers. In this model, production is divided into two sectors—harvesting and manufacturing—where productivity in the harvesting sector is directly proportional to the current stock of the resource. In a two-country version of their model, Brander and Taylor (1997) consider countries that differ in their ability to enforce property rights over a renewable resource stock. In the country with weak property rights and open access, the forest is over-exploited relative to the optimum in autarky. If the autarky level of over-exploitation is high enough, then the incentive to export helps correct the productivity losses from over-extraction in autarky and both countries can gain from trade. Otherwise, the country with weak property rights suffers long-term losses from trade.

4.3.2 Empirical approaches

Reforms to property rights regimes. Several policy reforms in countries with large stands of tropical forest have offered the opportunity to test the predictions of common-pool resource theory relating to property rights allocation and formalization, with mixed findings.[10] Wren-Lewis et al. (2020) conduct a village-level randomized evaluation of a land registration and demarcation program in Benin, finding that treated villages experienced a 20% decrease in forest cover loss and a 5% decrease in forest fires. On the other hand, Probst et al. (2020) examine a large land titling program in the Brazilian Amazon, exploiting the staggered timing of title allocations, and find that the program led to increases in deforestation among small and medium landholders (deforestation among large landholders remained unchanged), potentially driven by increased market integration and hence responsiveness to agricultural price increases. Similarly, Liscow (2013) finds that insecure property rights resulting from Nicaragua's 1981 agrarian reform law resulted in 14% higher forest cover levels in 2001, consistent with "investment effects" for long-term agricultural uses.

Local resource governance. An important dimension of common-property resources which was absent from original common property theory but has since been shown to have implications for optimal usage, is the extent to which they are *local*. Dasgupta (1996) emphasizes that the notion of "tragedy" may be misleading for local common-property resources because, in these settings, users of the commons are known to each other, interact strategically, and can achieve efficient outcomes even in the absence of a formal regulatory structure. Ostrom (1990), among others, provides empirical case study evidence for such local resource governance. Baland and Platteau (1996) similarly catalog examples of local common-property management and analyze both successes and failures of optimal usage. Seabright (1993) underscores that the time horizon of the repeated game that local commons users play is important for sustaining informal cooperation. Introducing formal private property rights, which can be traded at will, may undermine informal cooperation by reducing the time horizon of agents' interaction with one another, for instance by removing the threat of retaliation and the ability to build a reputation for collaboration.

More recent empirical evidence on local resource governance and deforestation is again mixed, and appears highly context-dependent. Alix-Garcia (2007) studies common-property forestry among small communities (*ejidos*) in Mexico. She finds that communities with more members exhibit higher deforestation levels, but also that inequality in the distribution of land ownership has an offsetting effect on deforestation. Baland et al. (2010) evaluate the impact of local forest councils (Van Panchayats) in the Indian Himalayas and find that, relative to state-managed forests, this community management was successful in regulating firewood and fodder extraction. In the Brazilian Amazon, Baragwanath and Bayi (2020) find that the demarcation and approval of full indigenous property rights from 1982 reduced deforestation within demarcated areas (which were common property among these indigenous communities) by up to 75%. On the other hand, BenYishay et al. (2017), using alternative satellite outcome measures, do not find significant reductions resulting from separate aspects of the same land titling process. Similarly, Kraus et al. (2021) do not find aggregate reductions in deforestation within a year of the recent rollout of community titling to forest land in Indonesia, and Eisenbarth et al. (2021)'s recent experimental evaluation of a village-level community monitoring program in Uganda finds that this may simply displace deforestation activity to neighboring, unmonitored areas. Given the heterogeneity in responses, determining more clearly what explains the heterogeneity in the effects of more secure land titling is an important area for future research.

[10] For a systematic review of the environmental effects of land tenure interventions, see, e.g., Tseng et al. (2021).

4.4 Political economy determinants of tropical deforestation

Our theoretical discussion to this point has focused on the levels of forest extraction that natural resource theory predicts under private ownership and open-access regimes, and has highlighted how these levels may differ from what a social planner would implement. However, political economy considerations may result in a divergence between the social planner's optimum and the forest management policies that governments enact in practice. Moreover, a growing body of research has highlighted that the economic and electoral incentives of local politicians, as well as constraints on enforcement capacity in remote areas, may drive a wedge between the *de jure* design of forest management policies and their *de facto* implementation.

4.4.1 Theoretical issues

National policy dynamics. National governments are subject to lobbying pressures from both pro-exploitation and pro-conservation interests, which can lead to sizeable changes in policy orientation with respect to deforestation. Harstad (2020) develops a theoretical model that explains why such policy reversals may occur in practice, highlighting the fundamental asymmetry between lobbying by pro-exploitation interests (e.g., logging firms) and pro-conservation interests (e.g., external donors) in the dynamic game played by consecutive administrations presiding over an exhaustible resource. The conservationist lobby must pay the government in perpetuity in order for the forest to be conserved, while the deforestation-oriented lobby need pay only once to deforest in the present. When deforestation becomes relatively more valuable, the current government will extract more not only for the value it receives from extraction today, but also because it expects future governments to extract more, a phenomenon Harstad refers to as the "conservation multiplier." Due to the multiplier, even small changes in these relative returns may lead to large changes in deforestation levels.

Harstad's theory also implies that greater political instability—when current administrations face a higher probability of being unseated in the future—will lead to higher deforestation levels because the current administration places lower value on future conservation payments. Other studies, such as Robinson et al. (2006) and van der Ploeg (2018), have developed related models which predict that a lower probability of being in power in the future will lead a government to extract more of an exhaustible natural resource today.

Illegality and electoral accountability. When logging firms face high costs of compliance with conservation regulations, lobbying to influence policy decisions is not their only possible recourse: firms may instead pay bribes for local officials to 'look the other way' when deforestation is *de jure* illegal. Harstad and Svensson (2011) develop a theory that formally distinguishes when firms will bribe to deforest illegally rather than lobby to change laws entirely. Intuitively, their model predicts that firms with low levels of capital will pay bribes to circumvent the rules; as firms grow, the level of bribes demanded by local bureaucrats will rise because firms' cost of compliance with the law and the bureaucrat's probability of detection both increase. Eventually, bribes rise to a level at which firms would rather lobby to change the rules entirely. However, anticipating future increases in bribes demanded, firms may avoid investing in growing their capital stock today, and this hold-up problem may lead to a poverty trap.

Other studies have focused on optimal design of forestry policy taking into account the possibility that harvesting firms may bribe local officials. Amacher et al. (2012), for example, present a model in which a central government can determine concession sizes, the royalty rate that harvesting firms must pay in order to log, and the wages of local inspectors. Harvesting firms may in turn bribe the local inspector, who faces an exogenous probability of detection, in order to harvest illegally. While the numerically optimal policy depends delicately on the interactions between these parameters, one insight that emerges is that when local inspectors are corruptible, the central government may optimally create smaller forest concessions relative to a no-bribery scenario in order to limit the harms of illegal deforestation.

Finally, given the rents associated with public office, electoral incentives may play an additional role in determining when and how often local officials facilitate illegal deforestation. Following the foundational work of Nordhaus (1975) on the political business cycle, a growing body of literature has documented electoral cycles in environmental protection—whereby more or less deforestation activity occurs in the year leading up to a local election—with the implication that officials intentionally exert more or less control over environmental outcomes during election years in order to curry favor with voters.

Cross-jurisdiction interactions. Given the importance of local officials' economic and electoral incentives for *de facto* implementation of conservation policies, pecuniary externalities between neighboring jurisdictions within a single timber market may have important implications for deforestation levels in equilibrium. Harstad and Mideksa (2017) develop a theoretical framework for understanding these forces with a particular focus on the strength of local institutions. Deforestation in one district imposes a pecuniary externality on neighboring districts in the same timber market because it lowers the price of timber products. Whether this externality is positive or negative, however, depends on the degree to which district officials can appropriate revenues from logging and how enforcement costs vary as the price increases. If institutions are strong and district governments can appropriate revenues from logging, and if enforcement costs are small or inelastic, then additional logging in one district imposes a negative externality on neighboring governments, who receive lower timber revenues. On the other hand, if enforcement costs are large and elastic, then deforestation in one district imposes a positive pecuniary externality on its neighbors by making illegal logging less lucrative, thus reducing enforcement costs for neighboring districts.

4.4.2 Empirical approaches

National conservation policies. Burgess et al. (2019) provide evidence of the importance of national forest management policy in the Brazilian Amazon, implementing a spatial regression discontinuity design that exploits the discrete shift in policy regime occurring at the national border between Brazil and its Amazonian neighbors. The authors find that between 2001–2005, deforestation rates on the Brazilian side of the border were three to four times higher than on observably similar land located just across the border. This jump in deforestation activity at the border disappeared in 2006, coinciding with the enactment of several conservation policies at the national level, but returned in 2014 during a time of weakened environmental regulation. Although this evidence suggests that national policies may effectively reduce deforestation, developing country governments may still have only limited resources to devote to implementation and enforcement. Assunção et al. (2019) analyze how optimally to allocate limited government resources in an environment characterized by weak institutions. They first evaluate the impacts of the Brazilian government's "Priority List," formulated in 2008, which designated 36 municipalities to receive more intensive deforestation monitoring. They find that the list led to a 43% reduction in deforestation in directly targeted municipalities, and also had spillover effects on neighbors and municipalities with high historical deforestation rates. They then develop a general framework for calculating the optimal priority list of municipalities, and find that the priority list that was actually implemented led to roughly 12 percent higher carbon emissions than the optimal priority list.

Mangonnet et al. (2022) focus on the national processes that influence which land is actually conserved under such policies in practice. The Brazilian government can legally designate protected areas in the Amazon via executive order of the president; at the same time, forest conservation entails meaningful economic costs at the local level, as communities are deprived of the opportunity to profit from forest extraction.[11] Consequently, political calculations at the national level may shape the spatial distribution of protected areas as the executive branch avoids inflicting such costs on political allies. The authors document that municipalities with mayors from opposition parties were 26–32% more likely to be designated as protected areas over the period spanning 1997–2012, perhaps driven by an attempt to preserve the economic rents of local elites in politically aligned municipalities.

Enforcement ability. Theoretical studies have emphasized the potential of bribery and corruption by local officials to undercut the effectiveness of national conservation policy. Moreover, weak state capacity in remote areas implies that logging firms may be able to engage in illegal deforestation without detection—i.e., without the need to pay any bribes whatsoever. In light of these enforcement difficulties, Assunção et al. (2017) evaluate the effectiveness of an innovative satellite-based monitoring initiative in the Brazilian Amazon. Brazil's Real-Time System for Detection of Deforestation (DETER), which came into operation in 2004. The authors exploit the fact that on cloudy days, DETER's satellite-based technology was relatively less effective at detecting changes in forest cover. They therefore use yearly cloud cover as an instrument for environmental enforcement activity in a municipality, as proxied by the amount of fines levied. The authors' 2SLS estimates suggest that a 50% increase in annual enforcement mediated by DETER led to a 25% reduction in annual municipality-level deforestation over the period 2006–2016.

Interactions with elections. As discussed above, local officials may become relatively more permissive of illegal deforestation activity in years leading up to a local election if doing so improves their electoral prospects. Studies testing for electoral deforestation cycles typically employ locality-level panel data and difference-in-differences specifications to compare trends in deforestation within localities and their coincidence with election timing. Pailler (2018) adopts this strategy to estimate the effect on deforestation rates in Brazil when a municipality's mayor ran for re-election relative to municipalities where the incumbent did not seek re-election. She finds that deforestation rates were 8–10% higher in election years in municipalities in which the incumbent mayor ran for re-election between 2002–2012, and notes that deforestation may even be a mechanism for funding electoral campaigns directly.

Of course, an empirical difficulty is the potential endogeneity of incumbents deciding to seek re-election. Other studies have surmounted this challenge by exploiting idiosyncratic variation in the timing of local elections. For example, Indonesia's post-Soeharto decentralization in the late 1990s induced variation in the timing of district head elections on the basis of when sitting district heads' terms expired. Burgess et al. (2011) and Cisneros et al. (2021) exploit this variation and look for deforestation impacts. Both papers find increased deforestation rates in the year leading up to a district head election. Cisneros et al. (2021) further argues that the political and economic drivers of deforestation interact: the electoral deforestation cycle was amplified in districts that were more exposed to fluctuations in the global price of palm oil. Balboni et al. (2021a), on the other hand, find that the incidence and physical scope of forest fires used for land clearing declined significantly in election years from 2005–2014. This divergence with previous results perhaps stemming from the

[11] On the other hand, Sims (2010) documents in Thailand that protected areas may also carry local development benefits in the form of increased tourism income and lower local poverty rates.

conspicuous nature of fire-setting via air pollution and other salient, short-run negative externalities.

Cross-jurisdiction interactions. Burgess et al. (2012) provide direct empirical evidence of local officials' incentives to permit deforestation activity and of pecuniary externalities across districts. They exploit the occurrence of administrative splits in local district governments in Indonesia from 2000 to 2008. They find that the creation of an additional district within a province led to an 8.2% increase in the deforestation rate within that province, both in legal and illegal logging areas, accompanied by a decrease of approximately 3.3% in the local price of timber. The authors show that their results are consistent with a model of Cournot competition between districts in which district heads choose the quantity of legal and illegal logging permits to sell. They also show that, if the probability of detection of illegal permits is increasing in the quantity of timber extracted, then external sources of rents from office will substitute for rents from logging permits, and provide evidence for this prediction by showing that district-level oil and gas revenue shocks lead to less deforestation in the short run.

These findings underscore that the incentives of local officials are important determinants of deforestation activity, holding fixed the national policy regime. Moreover, decentralization of forest management to local communities may not necessarily lead to lower deforestation rates in settings where local elites can collect rents from forest resources. On the other hand, Alesina et al. (2019) exploit the same natural experiment to present countervailing evidence: administrative splits led to an increase in ethnic homogeneity within many of the resulting districts, which they argue facilitated voter coordination to "control" politicians and punish them for perceived corruption and the degradation of public goods. As a result, district splits that resulted in greater ethnic homogeneity led to a decrease in deforestation within those districts. However, the magnitude of this effect is ultimately outweighed by the positive deforestation effects of increased Cournot competition discussed by Burgess et al. (2012).

5 Conclusion

The greenhouse gases that drive climate change can emanate from any national jurisdiction but affect the climate globally. A realization that tropical deforestation is both proceeding at a rapid pace and will significantly affect the earth's warming trajectory has elevated global interest in deforestation: what had been, until relatively recently, a domestic natural resource extraction issue has morphed into an international policy concern.

The measurement revolution which allows us to monitor forest loss at a granular level does not, in and of itself, allow us to understand what drives it. For this, we need to focus on the economics of tropical deforestation. In many ways, economic analysis, both theoretical and empirical, is trying to catch up with the complex and fast-moving land use changes revealed by the data and to quantify the relative magnitudes of the forces driving them. In the tropics, this requires moving beyond classical models of optimal resource extraction and forest yield management to encompass growing pressures for land use change that accompany development, national and international externalities, insecure property rights, and political economy challenges. This is the scaffolding that we use to build our review of the recent literature.

There are many fronts on which further progress can be made. To conclude our review, we point to some key areas where we believe the need for additional evidence is most pressing.

Perhaps most obvious is that we need to get much better at measuring the value of conservation. Growing awareness of the externalities imposed by deforestation has brought into focus the considerable value that might be derived from conserving the vast tracts of forests in the tropics. We are, however, in the infancy of quantifying these values precisely enough to serve as a useful guide for policy. Making progress here requires accurate, highly disaggregated measures of the carbon stored in different stands of forest, as well as conservation value extending beyond carbon to encompass, for instance, biodiversity, soil, and watershed protection values. It also requires improvements in measurement of the social costs of deforestation at the local, national, and international levels, as well as frameworks for aggregating these costs. Weighing the value of conservation with that of extracting timber and converting forests to alternative uses needs to form the bedrock of policies to confront tropical deforestation. With proper estimates of the social value of conservation, policymakers can determine how much landowners must be paid for conservation, or taxed for deforestation, to properly include the true social costs of deforestation in their calculus.

But of course, knowing the magnitude of the external costs of deforestation is only the first step. Implementing policies to address them at scale is much harder. While a growing literature on payments for ecosystem services considers how to calibrate, finance, and structure these types of payments, there is still much to learn about which schemes work, whether and how they can be taken to scale, and how to think about these contracts dynamically over long time periods. There is also limited empirical evidence on how to structure, let alone implement, Pigouvian taxes that penalize firms and individuals for deforesting. Moreover, given that tropical deforestation is, in many contexts, tied to financial precarity and land insecurity, the implementation of PES transfers and Pigouvian taxes carry important equity and anti-poverty considerations that economists have not yet explored thoroughly. There is considerable scope to develop a broader program evaluation literature on the use of payments and penalties to constrain deforestation, drawing on a range of methods from randomized trials to structural models which take general equilibrium effects into account.

Another potential policy route is regulation—controlling quantities of deforestation rather than setting prices. Governments could bring even more forest under national ownership or protected status—disallowing other uses—or better enforce regulations on the amount of deforestation that are already on the books. Such approaches may be particularly important in situations where property rights are insecure or ill-defined. But as we have discussed above, enforcement remains a challenge in many contexts, and so understanding how to do so effectively, particularly in areas with limited state capacity, remains an important area for future research.

National policies, whether price-based or quantity-based, require the buy-in of local government agents to make sure they are enforced, which highlights another key area for further work: how to tilt the incentives of politicians and civil servants in favor of conserving rather than degrading tropical forests. Politicians and civil servants control policies, such as building infrastructure or openness to trade, which encourage land use change. They also design and implement the environmental regulations which govern the use of forested land. There is clearly a need for more research into how government representatives are captured via corruption and lobbying by firms who want to convert forest to other uses, and which reforms may make them more accountable to domestic and international citizens who favor conservation.

An additional key challenge is the disconnect between national policy jurisdictions and the international incidence of impacts. What happens to the vast stands of forest in the Amazon, Congo Basin, and Indonesia will affect citizens everywhere, yet those outside the countries which contain tropical forests have limited means of influencing their rate of extraction. The design of international policies that can align local incentives with global costs is thus an agenda of global interest. Recent work on whether trade policy and other cross-national instruments can be used to encourage conservation in countries which otherwise might deforest their territories in order to promote development has begun to explore these issues.

Finally, failures of accountability are often most striking in the forested areas of the tropics that are afflicted by conflict. This points to the particular need for more work on drivers of deforestation in the Democratic Republic of the Congo and sub-Saharan Africa more generally, which are poorly represented in the literature on tropical deforestation and where different types of policies may be needed to constrain rampant deforestation.

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Appendix for "The Economics of Tropical Deforestation"

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June 29, 2023

A Renewable resource depletion under sole ownership

We assume that forest property owners are price-takers in the market for forest products.[1] The owner's payoff in period t is $py_t - c(y_t, X_t)$, where p is the market price of timber, y_t is the quantity extracted, X_t is the current size of the forest (i.e., the timber stock), and $c(y_t, X_t)$ is the cost to the owner of extracting y_t given current stock X_t.[2] The owner's problem, then, is to choose a path of extraction $\{y_t\}$, given some initial forest stock X_0, that maximizes the net present value of extracted timber:

$$\max_{\{y_t\}} \int_0^\infty [py_t - c(y_t, X_t)]e^{-rt}\, dt$$

subject to

$$\frac{dX_t}{dt} = g(X_t) - y_t$$

where $g(\cdot)$ is known as the *natural growth law* of the forest. Typically, $g(\cdot)$ is assumed to be strictly concave—most commonly using a quadratic function in X_t, which yields the familiar logistic evolution of stock over time.[3] We assume for simplicity that the interest rate r is constant and that extraction costs $c(\cdot, X_t)$ are convex in y_t.[4]

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[1] To fix ideas, we simply use "timber" to capture a potentially broader category of products in what follows.

[2] The payoff can be equivalently represented by substituting in place of $c(y_t, X_t)$ a term equal to the total wage paid for extraction effort (as in Peterson and Fisher, 1977). In other words, the process of extraction can be represented using a production function with labor as an input rather than using a cost function. We choose to present the cost function formulation here as it will allow us more clearly to highlight the role of external social costs of deforestation.

[3] Bioeconomic models sometimes assume a growth law that depends on other factors such as inter-species competition and age structure of the resource stock. For the sake of simplicity, we abstract from such forces here.

[4] Note that the problems facing the owner of an exhaustible natural resource and the owner of a renewable resource are quite similar: their objective functions are identical, and in the former case, the owner's flow constraint instead reflects the purely exhaustible nature of the resource. In particular, their flow constraint is simply $\frac{dX_t}{dt} = -y_t$. They also face the constraint that $\int_0^\infty y_t\, dt \leq X_0$.

In most expositions of the owner's renewable resource extraction problem, the optimal path of extraction is derived using the Pontryagin maximum principle. The Hamiltonian for the owner's maximization problem is:

$$H = py_t - c(y_t, X_t) + q_t[g(X_t) - y_t]$$

where the co-state variable q_t represents the shadow price of timber: it is the amount by which the net present value of the forest decreases when one unit of timber is extracted today. Peterson and Fisher call this co-state variable the "marginal user cost" of resource extraction. As we will discuss below, a key distinction between sole-ownership and common-property depletion is that in the latter case, agents do not take this marginal user cost into account.

The Hamiltonian above yields the necessary conditions:

$$\frac{\partial H}{\partial y_t} = 0$$

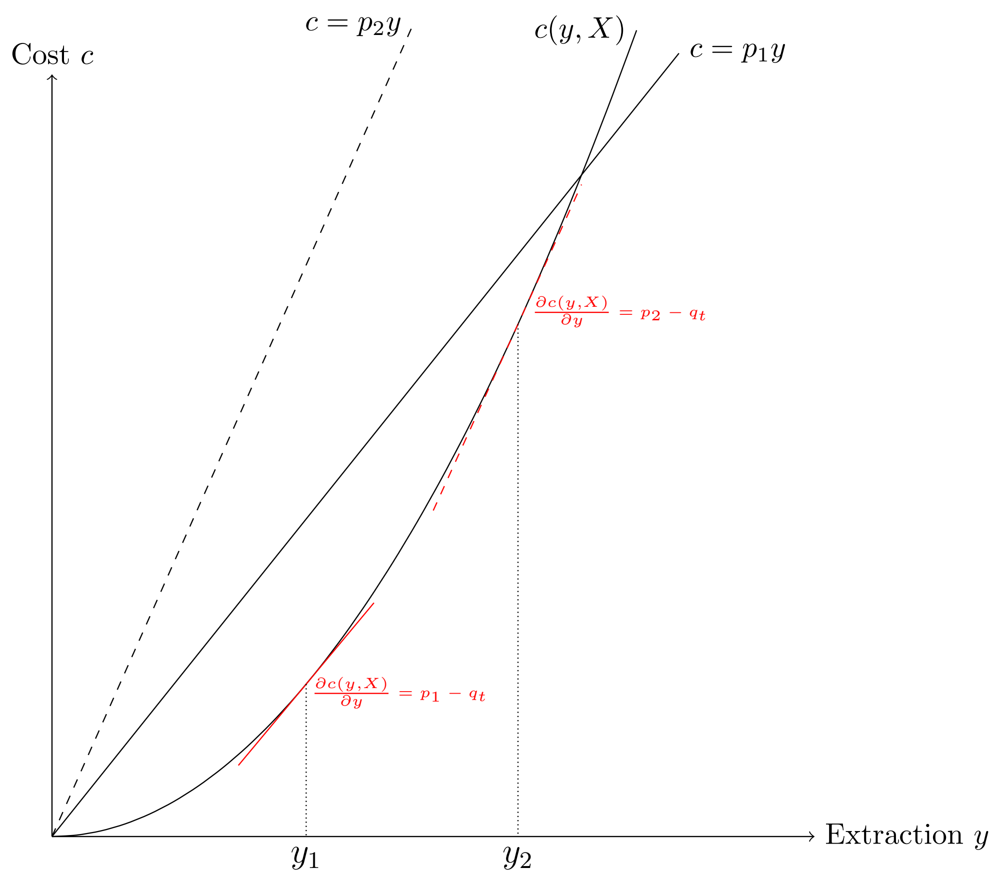
$$\implies p = \frac{\partial c(y_t, X_t)}{\partial y_t} + q_t \tag{1}$$

$$\frac{dq_t}{dt} = rq_t - \frac{\partial H}{\partial X}$$

$$\implies \frac{dq_t}{dt} = rq_t + \frac{\partial c(y_t, X_t)}{\partial X} - q_t \frac{dg(X_t)}{dX} \tag{2}$$

Equation (1) indicates that the optimal extraction path equates price with the marginal cost of extraction plus the marginal user cost of timber in each period. One insight that emerges from this analysis is that, even in the absence of any market imperfection (e.g., market power on the part of the timber-extracting agent), price does not equal marginal cost—the wedge between the two reflects the rivalrous nature of natural resource consumption, where rivalry in the single-agent case refers to consumption of the resource at different points in time.

Figure A1 presents this optimality condition graphically and illustrates how price changes for timber affect a sole owner's extraction decision within a given period. The convex curve represents the owner's extraction cost as a function of extraction levels y, assuming a given forest stock X. The solid straight line represents the owner's zero-profit curve along which revenues given a price p_1 are exactly equal to extraction costs. Given price p_1, the owner will extract an amount y_1 such that the marginal cost of extraction is equal to the price of output minus the marginal user cost q_t, which is assumed here to be small. Holding q_t constant, increasing the price of output from p_1 to p_2 (and moving from the solid to the dashed zero-profit line) leads the sole owner to extract a greater amount y_2 within that period.



Appendix Figure A1. Sole owner's single-period extraction decision

Equations (1) and (2) define the dynamics of optimal extraction y given any current stock X, and specifying the initial stock X_0 pins down the level of extraction and the size of remaining forest in each period along the equilibrium path. These conditions allow us to analyze the resulting steady state and conduct comparative statics. Let y^*, X^*, and q^* denote the steady-state levels of extraction, forest stock, and marginal user cost, respectively. In this model, a steady state is such that equation (1) and the following additional conditions hold:

$$\frac{dq_t}{dt} = rq^* + \frac{\partial c(y^*, X^*)}{\partial X} - q^* \frac{dg(X^*)}{dX} = 0 \tag{3}$$

$$\frac{dX_t}{dt} = g(X^*) - y^* = 0 \tag{4}$$

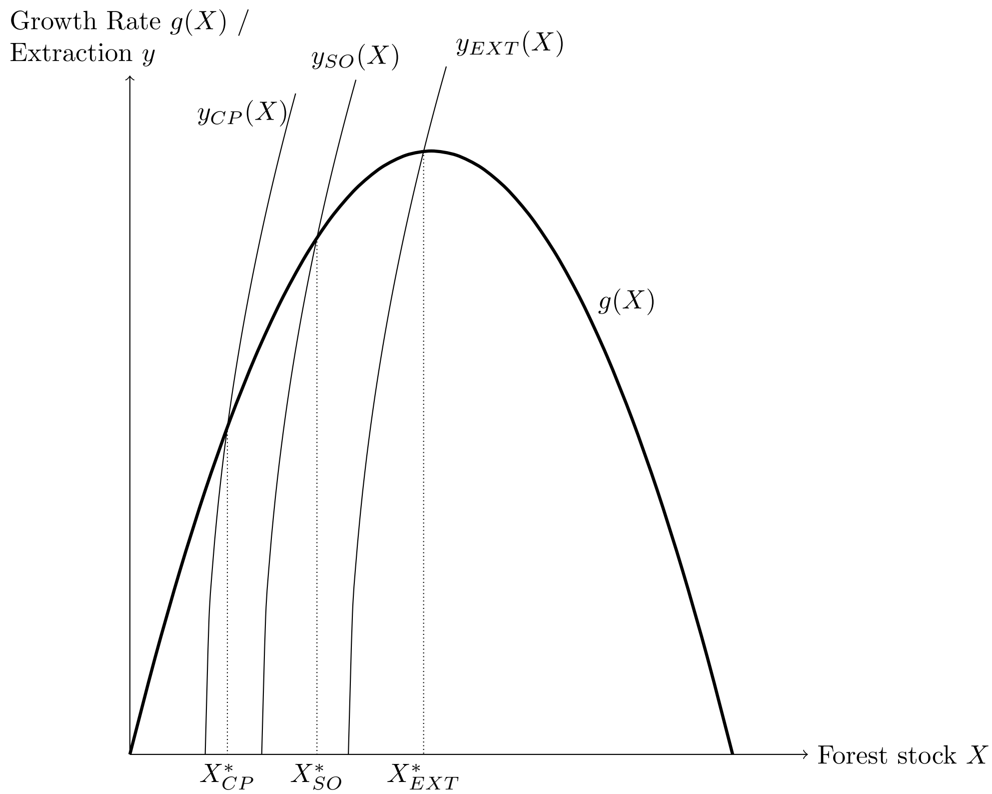
Note that Equation (1) implies that y^*, the steady-state level of extraction, will be such that the marginal cost of extraction equals $p - q^*$. Clearly, higher timber prices will lead to a higher steady-state level of extraction, as examined graphically above. Furthermore, given the convexity of $c(\cdot, X_t)$ in y_t, a negative level shift in marginal extraction costs will lead to higher y^*.

We first examine the case in which an interior solution (a steady state in which $y^* > 0$ and hence $X^* > 0$) exists. Equation (3) then illustrates an important conclusion emerging from the theoretical literature on forest management: the economically optimal path of extraction generically does not coincide with the notion of "maximum sustained yield," i.e., the maximum growth rate of the renewable resource—and hence the maximum rate of extraction—that can be sustained in equilibrium. Given a positive interest rate, $\frac{dg(X^*)}{dX} = 0$ may not be be optimal, i.e., the point of maximum forest growth may be below or above the owner's optimal level of extraction. If we further assume that $\frac{\partial c}{\partial X} = 0$ (the cost of cutting a given number of trees does not depend on the size of the remaining forest), then maximum sustained yield *cannot* be optimal from the standpoint of the owner; instead the steady-state optimum lies below the point of maximum sustained yield. Samuelson (1976) and Peterson and Fisher (1977), in particular, highlight this divergence, which is important because many ecologists and environmental policymakers at the time had tended to advocate for the maximum sustained yield notion by default.[5] However, only with an effective interest rate of zero will the "economists' optimum" coincide with the "foresters' optimum" of maximum sustained yield. Intuitively, economic discounting implies that the agent prefers to cut more trees today rather than to wait for the forest to grow further; the higher the discount rate, the higher the steady-state level of extraction and hence the larger the divergence from maximum sustained yield. In the specific context of tropical forests, agents' discount rates may be especially high (Barbier et al., 1991), due in part to insecure property rights and regulatory uncertainty (we return to this point below).[6]

Figure A2 illustrates one possible steady-state of the model graphically. The natural growth law $g(X)$ is depicted as the inverted parabola in bold. The line labeled $y_{SO}(X)$ represents the locus of single-owner optimal extraction levels as a function of the forest stock. This locus can be traced out by varying the forest stock X (and hence shifting the cost curve $c(y, X)$ outward) in Figure A1. For each stock level X, the optimal level of extraction $y_{SO}(X)$ can be determined according to the optimality condition in equation (1). A steady state X^*_{SO} then occurs when the level of extraction $y_{SO}(X^*_{SO})$ exactly balances the forest's natural rate of growth $g(X^*_{SO})$.

[5] Indeed, Peterson and Fisher (1977) and Goundrey (1960) note that the concept of maximum sustained yield was, at the time of writing, codified in US and Canadian forestry policies.

[6] For the purposes of considering social optimality, Samuelson (1976) relates his analysis to several other works (e.g., Ramsey, 1928; Diamond and Mirrlees, 1971) on the appropriate social discount rate.



Appendix Figure A2. Steady-state extraction and forest stocks

Equations (1) and (4) yield another important insight: if the price of timber is very high, or if the level of extraction costs are very low, then there may exist no positive X^* such that $g(X^*) = y^*$: the forest cannot grow at a rate that compensates for the high economic returns to extraction that the agent faces today. In this case, the only steady state is complete extinction of the forest: $y^* = X^* = g(X^*) = 0$. An analogous insight arises from Equation (3): if the interest rate r is very high, then again there may be no positive X^* such that $\frac{dg(X^*)}{dX}$ is large enough to sustain an equilibrium. As Peterson and Fisher (1977) note, forest stocks in this case "do not grow fast enough to justify waiting around for them." Extinction can occur even under sole ownership of the forest, a point developed rigorously in the work of Smith (1968), Clark (1973), and Neher (1974), among others. In Figure A2, such a scenario would be represented by an extraction locus $y_{SO}(X)$ that is shifted so far leftward that it intersects the growth curve only at $X = 0$. This insight has important implications for policy solutions to common-property resource issues we discuss below.

Similarly, if timber is valued at a very low price or if costs of extraction are high, then there may exist no stock X^* such that a positive level of extraction $y^* > 0$ is optimal: in this case, $y^* = g(X^*) = 0$, while the equilibrium stock equals the "carrying capacity" of the forest, i.e., $X^* = X_{max}$ such that $g(X_{max}) = 0$ and $g(X_t) < 0$ for any $X_t > X_{max}$. Timber production is not lucrative, so there will be no extraction whatsoever, and the forest will remain at the maximum size that natural growth constraints allow.

A.1 Alternative land uses

Directly incorporating alternative uses for forested land into the above model of resource depletion complicates the analysis substantially. One parsimonious approach taken by Bohn and Deacon

(2000) is to assume that demand for alternative land uses (e.g., agriculture) is perfectly inelastic and exogenously determined by food needs. Permanent shifts in demand for agricultural land can then be reflected in shifts in the carrying capacity of a given forest: greater food demand reduces the amount of forest land by re-designating a portion for agricultural production. In order to relax this assumption, one can additionally incorporate the quantity of cleared forest land (i.e., land that has been converted for agricultural use) as another state variable in the above dynamic system.

B Optimal resource depletion with deforestation externalities

There are several possible avenues through which the externalities of deforestation may be incorporated into our simple theoretical framework of resource extraction. Insofar as externalities arise from static misallocation in each period, the external costs of deforestation can simply be incorporated into the term $c(y_t, X_t)$ representing the costs of extraction (Fisher, 1981).[7]

To see how the presence of externalities can shift the socially optimal steady-state level of deforestation, we return to the set of equilibrium conditions derived in Section 4.1. Assume that, in addition to the private cost of extracting y_t units in period t, the owner also considers a static external cost $E(y_t)$ that does not depend on the current forest stock X_t.[8] The sole owner's maximization problem is then:

$$\max_{\{y_t\}} \int_0^\infty [py_t - c(y_t, X_t) - E(y_t)]e^{-rt}\, dt$$

subject to

$$\frac{dX_t}{dt} = g(X_t) - y_t$$

Equations (3) and (4) governing the resulting steady state are unchanged from above, but equation (1) now becomes

$$p = \frac{\partial c(y_t, X_t)}{\partial y_t} + \frac{dE(y_t)}{dy_t} + q_t \tag{5}$$

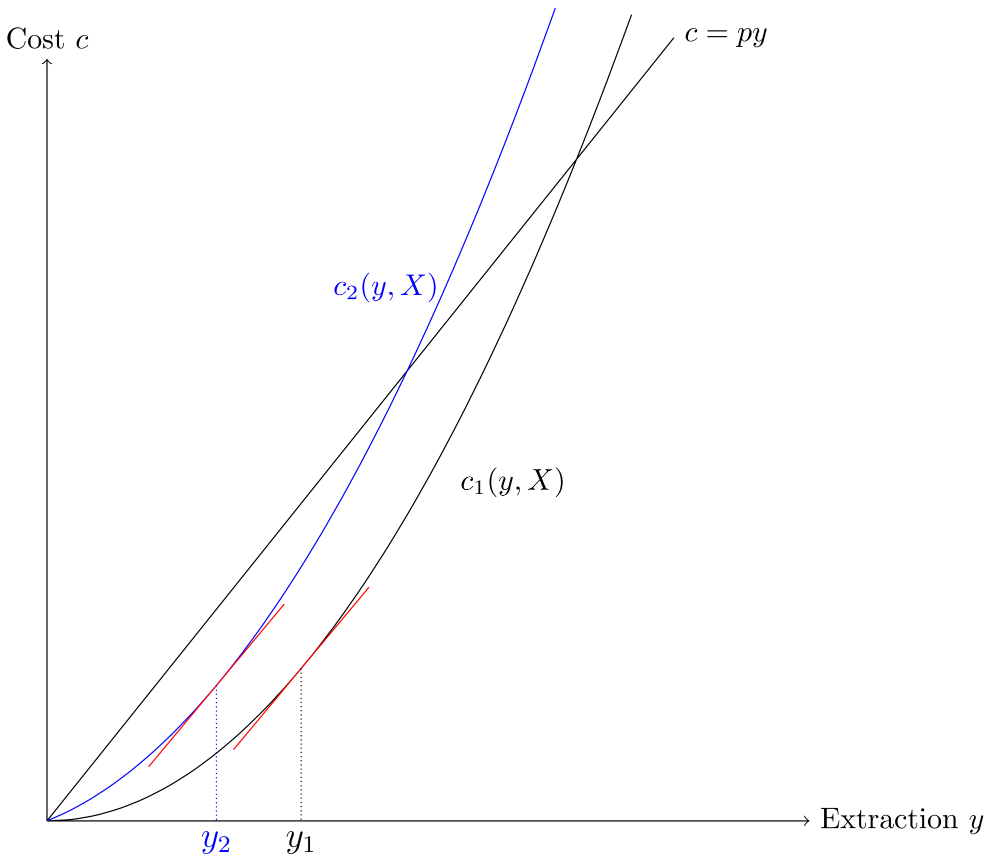
As long as the external cost of extraction is not decreasing in the amount extracted, the presence of deforestation externalities widens the wedge between the price of timber and the marginal cost of extraction. In the case of constant marginal external costs (such as when invoking the social cost of carbon), externalities represent a level shift upward in the marginal costs of extraction. The higher these external costs, the farther will be the socially optimal extraction level below the sole-ownership steady state level.

The influence of externalities on the socially optimal deforestation level is depicted graphically in Figure B1. The upward level shift in marginal costs of extraction are depicted as a shift from cost curve $c_1(y, X)$ (the cost curve faced by a sole owner depicted in Figure A1) to $c_2(y, X)$, drawn

[7] Alternatively, other models have explicitly accounted for the dynamics of waste accumulation as a byproduct of natural resource extraction (e.g., D'Arge and Kogiku, 1973; Rausser and Lapan, 1979) and have considered the optimal intertemporal control of pollution (e.g., Plourde, 1972; Keeler et al., 1971). Many of these models consider extractive natural resources as inputs into production of both goods and bads (e.g., factories use coal as an input and produce local pollutants in addition to consumer goods). To the extent that the main externalities of tropical deforestation are byproducts of the extraction process *per se*, rather than of production from forest resources, the "static" approach of incorporating external social costs in $c(y_t, X_t)$ captures the principal forces of interest while retaining tractability.

[8] Of course, a social planner might also consider the dynamics of carbon accumulation in the atmosphere and other environmental harm resulting from a given amount of extraction today. The assumption of static external costs, however, captures the qualitative insights of our model in a more straightforward fashion.

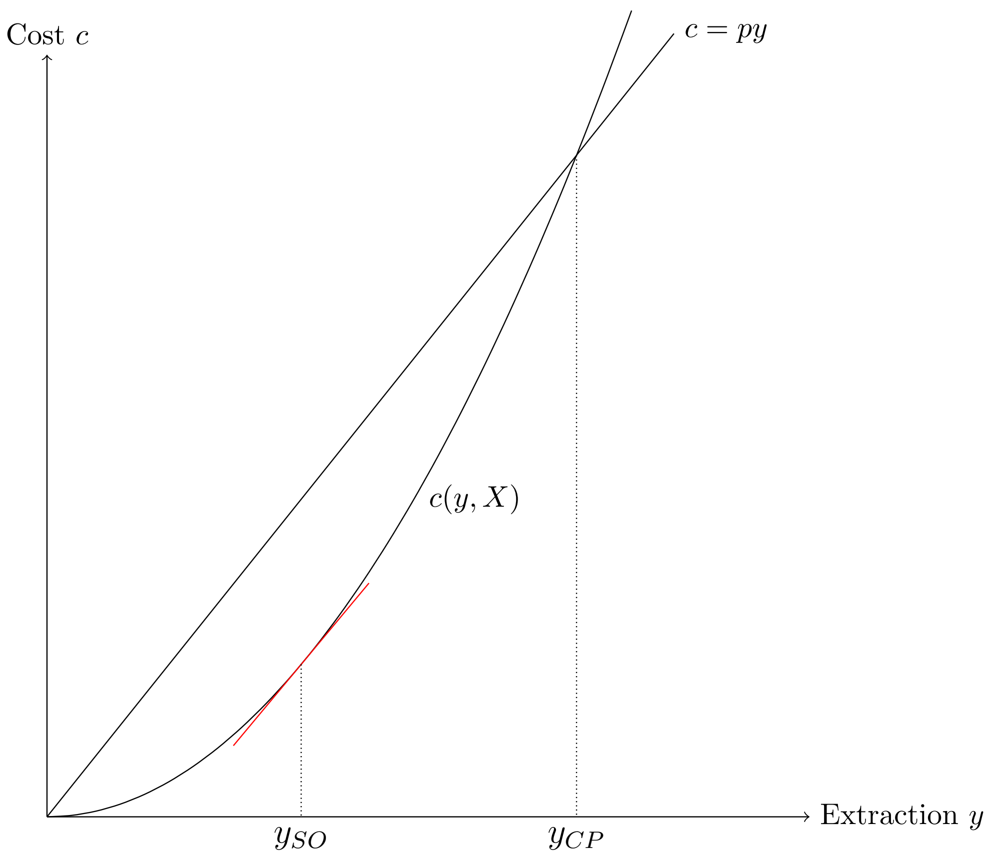
in blue. The optimal level of extraction within each period decreases accordingly, from y_1 to y_2.



Appendix Figure B1. Optimal extraction with externalities

C Resource depletion with common-property access

To illustrate how the free-entry equilibrium leads to over-exploitation relative to sole ownership, we revisit the model of renewable resource extraction developed in Appendix A. The zero-profit condition discussed in the previous paragraph requires that $py_t^{CP} - c(y_t^{CP}, X_t) = 0$, where y_t^{CP} denotes the agent's level of extraction in period t under the common-property regime. Note that extraction y and costs c are now aggregate quantities due to the free entry of foresting firms. The increase in aggregate deforestation is seen easily in Figure C1, as the common-property equilibrium occurs where the aggregate cost curve intersects the line $c = py$.



Appendix Figure C1. Common-property levels of aggregate extraction

A useful comparison arises from differentiating the zero-profit condition with respect to y, yielding:

$$\frac{\partial c(y_t^{CP}, X_t)}{\partial y} = p \tag{6}$$

Under a common property regime, deforesting firms enter until the marginal cost of extraction for any agent exactly offsets the economic return to extraction. By contrast, the single-agent condition (1) yields $\frac{\partial c(y_t^{SA}, X_t)}{\partial y} = p - q_t$, where y_t^{SA} denotes the sole-agent optimal extraction level. Because $q_t > 0$ unless the supply of the resource is truly unlimited, we have $\frac{\partial c(y_t^{SA}, X_t)}{\partial y} < \frac{\partial c(y_t^{CP}, X_t)}{\partial y}$, which, given the assumption of convex extraction costs, implies that $y_t^{SA} < y_t^{CP}$. Not only is aggregate extraction greater than under the sole-agent optimum, each individual agent also extracts more per period under a common property regime than they would under sole ownership of the forest.

Unlike "uni-directional" externalities studied in the previous section, common-property resources feature "reciprocal" externalities in which each agent's actions affect all other agents' yields, including their own. Such externalities arise because agents do not account for the marginal user cost q_t of extraction nor for their effect on the growth rate of a renewable resource through a depletion in stock. Compounding the fact that extraction in each period t is greater under common property than under sole ownership *given* a particular stock X_t, this higher level of extraction will lower the forest stock in the following period, making extinction even more likely than in the single-agent case (Smith, 1968; Peterson and Fisher, 1977).