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.

Keywords: climate change, environmental economics, tropical deforestation, environmental degradation, biodiversity loss, natural resource management, land use change, remote sensing, externalities, common-property resources, political economy

JEL Codes: F18, F64, H23, O13, O44, Q23, Q56, Q57

1 Introduction

> "Surely, from the vantage point of the final third of the twentieth century, few will agree with the beginning-of-the-century claim... that wood is our most important necessity, second only to food in societal importance. Wood is only wood, just as coal is only coal, plastics are only plastics, and, some would say, as bubblegum is only bubblegum." (Samuelson, 1976)

Over the 20-year period from 2001–2020, 1.48 million km^2 of forest area in the tropics was deforested—an area larger than France, Spain, and Germany combined. Forest loss in the tropics accounted for more than half of all global forest loss over this period, and in recent years it has increased more sharply than forest loss in the rest of the world, as depicted in Figure 1.[1] Much of the remaining tropical forest is also at risk: due to increasing fragmentation by roads and other human 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 tropical moist forest areas were considered degraded in 2020 (Vancutsem et al., 2021).
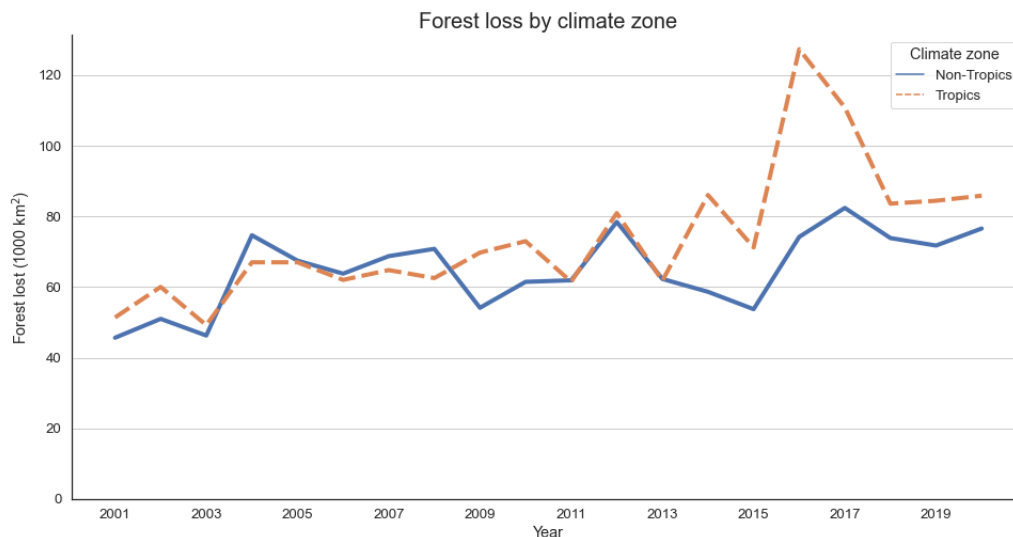

Figure 1. Forest loss by climate zone. Data on tree canopy cover in 2000 and gross forest cover loss from 2001–2020 are from (Hansen et al., 2013). Forest is defined as 50% tree cover. Loss data indicate binary occurrence of a forest loss event in a given pixel and the year in which the event primarily 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.

These trends have far-reaching ramifications. Globally, deforestation is a major driver of climate change, contributing an estimated 8–20% of total anthropogenic greenhouse gas emissions (Gullison et al., 2007; Tubiello et al., 2015; Van der Werf et al., 2009), up to two-thirds of which are attributed

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[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.

specifically to tropical humid regions (Achard et al., 2014). Lawrence and Vandecar (2015) project that complete loss of tropical forest cover would, on its own, increase mean global temperatures by 0.1-0.7°C.[2] Biodiversity loss associated with tropical deforestation 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 infant mortality due to smoke from forest fires in Indonesia (Jayachandran, 2009) and increased malaria prevalence in Nigeria (Berazneva and Byker, 2017) and the Amazon (MacDonald and Mordecai, 2019).

How should we think about tropical deforestation? Is "wood only wood," as Samuelson wrote in 1976, and do recent trends in tropical forest loss reflect 'optimal' land use changes given increased demand for forest products and the land on which the forest sits? And if not, what combination of challenges—unpriced externalities such as carbon emissions and biodiversity loss, ambiguous land rights, or political economy 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 basic tools and evidence economics can provide to help understand these questions—and, to the extent that forest extraction is 'too high,' what can be done about it. To do so, we organize our discussion of tropical deforestation following a set of simple theoretical frameworks of natural resource extraction, and we review the empirical literature relating to each. We hope not only to leave the reader with an understanding of where the evidence stands on the drivers of tropical deforestation, but also to illustrate how basic economic analysis can help understand these drivers, which can in turn clarify which policies may be effective in aligning deforestation decisions with their true social costs.

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 regressions of deforestation rates on economic and demographic characteristics. It was this early wave of papers that built the evidence base on how broad sets of country-level factors including GDP, population growth, agricultural expansion, agricultural productivity, commodity trade, and access to infrastructure related to tropical deforestation levels. In line with the cross-country growth literature that was blossoming at the same time (Barro, 1991), this literature also examined the role of institutional factors such as property rights, corruption, and political instability. Building up from within-country case studies to cross-country regressions allowed researchers to begin to discern the determinants of deforestation rates in the tropics and how these determinants varied across settings. This early 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.

As Barbier and Burgess (2001) discuss, 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). 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 consider how to design policies to slow the rate of deforestation, as the threat of climate change and other negative externalities from mass tropical deforestation became more apparent.

In recent years, what might be labeled a "third wave" of economic research on tropical defor-

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[2]Bala et al. (2007) estimate similar impacts even after accounting for countervailing albedo, evapotranspiration and cloud-cover effects.

estation has surged. This wave has been triggered by the widespread availability of increasingly high-resolution, high-frequency satellite data that allows researchers and policymakers to monitor land use—and hence deforestation—across the whole planet and in near-real time. It is this measurement revolution that has facilitated a new wave of papers on the economics of tropical deforestation. Concurrent with the 'credibility revolution' in empirical economics (Angrist and Pischke, 2009), this research also has 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.

Therefore, we begin our review in Section 2 with a survey of recent developments in measurement and data availability. The advent of satellites with increasingly high-frequency and high-resolution coverage of the entire globe, as well as advances in computing and machine learning to process the petabytes of data generated by these satellites, has given rise to a new generation of remotely sensed data on deforestation. Remote sensing now allows us to observe tropical deforestation across the world at a pixel-by-pixel level over time, as illustrated in Figure 2. These data have, in turn, revolutionized our understanding of what is happening to the world's tropical forests, and form the basis for new empirical work analyzing the drivers of tropical deforestation.
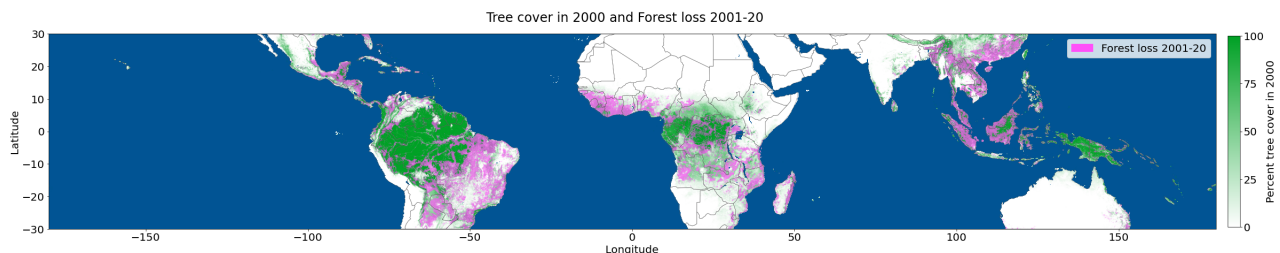

Figure 2. Tree cover in 2000 and forest loss from 2001–2020. Data on tree canopy cover in 2000 and gross forest cover loss from 2001–2020 are from (Hansen et al., 2013). Forest is defined as 50% tree cover. Loss data indicate binary occurrence of a forest loss event in a given pixel and the year in which 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 earnest 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, and the main focus is on how long to let forests grow before harvesting rather than whether to preserve the

forest at all or use the land for some alternative use.

While this basic model of optimal forest management may apply neatly to how a firm like Weyerhaeuser manages its North American forests for paper pulp 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, as in a classical model of forest management of an exhaustible resource. Rather, in many cases, the property owner's value of deforestation comes from an alternative land use. For example, growing soy beans or raising cattle (as in Brazil) or growing oil palm (as in Indonesia) requires clearing tropical forest land. 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 or in prices of agricultural commodities, from soybeans and cattle in Brazil to oil palm in Indonesia— are, indeed, an important driver of tropical deforestation. Important in this discussion is the role of increased demand for commodities from expanded international trade.

Second, we generalize the benchmark framework to include external costs 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 (whether a government or a non-governmental coalition) 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 (avoiding paying for land that would have been conserved anyway) to commitment problems. We therefore review the empirical evidence on whether, in practice, such payments for environmental 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 more ambiguous. 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 (Goetz et al., 2015). As a result, forest inventories based on such methods are sparsely available for tropical countries (Nesha et al., 2021).

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 (Giles and Burgoyne, 2008, Nesha et al., 2021). The temporal dimension 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 (Scott, 2014, Hsiao, 2021, Araujo et al., 2020). 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 Global Forest Watch in Indonesia (Bourgault, 2018) 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 use the fact that different materials on the ground absorb, reflect and emit electromagnetic energy to varying extents at different wavelengths (see, for example, Kennedy et al., 2009, Lillesand et al., 2015). Optical sensors track these differences in reflectivity, and then various statistical and machine learning techniques are used to predict tree cover, deforestation, and even vegetation type as a function of these underlying reflectivity measurements.

Sensors attached to different satellites sample light intensity across multiple discrete areas on the ground (the dimensions of which define the sensor's spatial resolution), at different intervals (which determines the frequency each geographic area is imaged), and different spectral bands (the width of which define the sensor's spectral resolution).

In recent decades, new generations of satellites have allowed for different combinations of higher resolution, higher sampling frequency, and finer spectral resolution. Images collected by Landsat satellites serve as the basis for many forest monitoring products. These provide consistent global coverage from 1999 with a 16-day repeat cycle and spatial resolution of up to 30 meters, and 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 time since 2015. Commercial Earth observation systems such as the Planet constellation, QuickBird, World-View and SPOT also provide very high resolution ($< 10m$) imagery for the monitoring of forest areas.

There are several challenges associated with using this data. First, one needs pre-processing of

the raw imagery to reduce unwanted variation due to, for instance, the solar angle, sensor viewing angle or atmospheric conditions, and to accentuate useful features (see e.g. Hansen and Loveland, 2012). While algorithms have been developed to achieve this, it remains challenging to correct, for example, for cloud and haze cover in many tropical regions, where long-term yearly average cloud cover can exceed 80% in some areas (Tarazona et al., 2021, Mitchell et al., 2017, 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, to turn a vector of radiances in different spectra into forest cover or deforestation. 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 statistically-based decision rule, such as supervised machine learning. Such 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.[3] Validation using a different secondary dataset is often used to check the accuracy of this output (see, for example, Olofsson et al. (2014)). Supervised classification over large heterogeneous areas is challenging given the need for training data to provide a complete description of classified objects. 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 (Achard et al., 2014 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, Hansen and Loveland, 2012, Jackson and Adam, 2020). This is also true for reforestation, since this 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 particular importance of understanding the data generating process underpinning remotely sensed data products and considering how this 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 systematic variation in accuracy, compounded by the nature of derived variables (for example, the bounded nature of deforestation versus no-deforestation), may result in non-classical measurement error.

For example, Alix-Garcia and Millimet (2021) compare two commonly used deforestation datasets[4] and find that discrepancies between the two sources are correlated with geographic features including slope, elevation, biome and the availability of cloud-free images, and consider biases that may result from the measurement of deforestation as a binary outcome variable. They propose a solution based on an extension to the misclassification binary choice model proposed in Hausman et al. (1998). 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 if this differs

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[3]See, for example, Lillesand et al. (2015), Shi et al. (2020) and Hussain et al. (2013) for more details on classification algorithms.

[4]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).

from the scale relevant to economic decision-making – when the unit of measurement is too small, attenuation bias is introduced through unnecessary noise in independent variables, while aggregation may lead to bias when it is too large – as well as the importance of spatial lags. Ratledge et al. (2021) highlight that inference may be undermined if machine learning uses the treatment of interest for prediction and discuss 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 using validated data 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, including nonrandom misclassification, saturation effects and cloud cover.

Despite these challenges, these datasets represent an enormous step forward in our ability to monitor and analyze changes in forest use at high resolution across entire forests, nations, and the world. They provide unparalleled micro data on deforestation at a fine spatial resolution, repeated over time; microeconomic users should just be aware of some of the econometric challenges when using these derived variables on the left-hand side of regressions highlighted by this recent literature.

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 the 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), Baccini et al. (2012), Dupuis et al. (2020)). Active remote sensing products may be especially well suited to forest monitoring in tropical regions: for instance, the ability of synthetic-aperture radar to penetrate clouds could significantly improve forest monitoring in cloudy tropical landscapes, and the capacity to measure forest structure is yielding increasingly accurate measures of aboveground biomass. 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).

2.2   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 (Hansen et al., 2013) which, building on Landsat satellite images, provides annual global maps of tree cover, gain and loss since 2000 at a spatial resolution of 30 meters (for use cases in economics, see, for example, Ferraro and Simorangkir, 2020, Berazneva and Byker, 2017, Austin et al., 2019, Carlson et al., 2018, Balboni et al., 2021b, Burgess et al., 2019, Leijten et al., 2021).[5] For example, Figure 3 uses this dataset to describe 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, accounting for 32%, 16% and 9% respectively of global tropical forest loss from 2001-2020. They also reveal dramatic temporal changes in deforestation rates over time, such as the decline in deforestation in Brazil in the mid-2000s and the uptick starting in the late 2010s.

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[5]The dataset is generally reported to have high accuracy (Zhang et al., 2020, Galiatsatos et al., 2020), though recent work has considered challenges of non-classical measurement error (Alix-Garcia and Millimet, 2021) and systematic underestimation of small-scale disturbances (Burivalova et al., 2015, Milodowski et al., 2017).
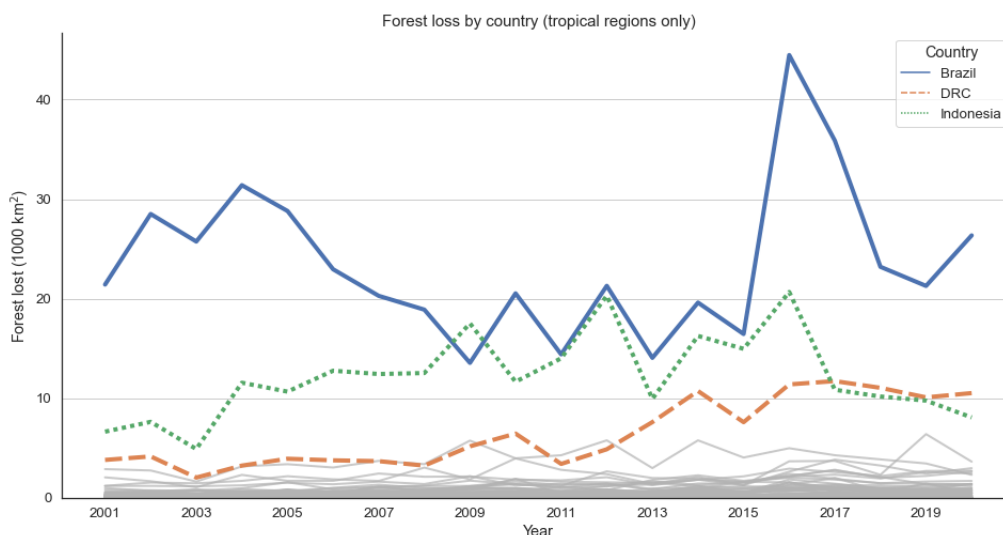
Figure 3. Forest loss by country, tropical regions only. Data on tree canopy cover in 2000 and gross forest cover loss from 2001-2020 taken from Hansen et al. (2013). Forest defined as 50% tree cover. Loss data indicates binary occurrence of a forest loss event in a given pixel and the year in which the event primarily occurred. Binary forest loss occurrence was multiplied by 2000 tree cover to indicate the extent of the loss by year, and then aggregated by intersections of country boundaries (from www.gadm.org) and the climate zone inside the region between the Tropics of Cancer and Capricorn.

Song et al. (2018) extend global coverage 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. One note is that both datasets capture 'tree cover' rather than 'forest cover' (Tropek et al., 2014), and may be used in combination with secondary data (e.g. Margono et al., 2014, Turubanova et al., 2018, Potapov et al., 2017) to consider deforestation.

Other datasets focused specifically on tropical forests have also been used in economic studies (Sze et al., 2022, Assunção et al., 2015b, Hargrave and Kis-Katos, 2013, Pfaff et al., 2014). Vancutsem et al. (2021) provides 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 and maps deforestation and degradation separately. Locally calibrated products are available in some areas, such as Brazil's official data (Programa de Monitoramento do Desmatamento da Floresta Amazônica Brasileira por Satélite) on deforestation in the Brazilian Amazon since 1988 at a resolution of 30 meters.[6]

In addition to monitoring changes in the state of forests, satellite data has also been used to detect the carbon density of tropical forests (Baccini et al., 2012) - an important input in understanding the emissions contribution of tropical deforestation - as well as forest fires, which both lead to forest degradation and are an important environmental concern in their own right. In particular, 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 onwards, used for example by Balboni et al. (2021b). The dataset routinely captures fires of at least 1000 m^2 and, under

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[6]Land use and land cover data sets, mapping a broader range of land classes, may also be used for forest monitoring. For a recent review of some of these products see, for example, Pandey et al. (2021)

optimal conditions, can detect fires as small as 50 m^2 (Giglio et al., 2015).

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 (e.g., Hotelling, 1931), and then turn in Section 3.2 to the workhorse renewable resource models (building on, e.g., Gordon, 1954; Scott, 1955; Schaefer, 1957) 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.

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, given that period's price, 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 manifestation of this sharpened attention to natural resource scarcity, was a special *Review of Economic Studies* symposium issue on the economics of exhaustible resources was assembled 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. Weinstein and Zeckhauser (1975), writing to countervail contemporary concerns about resource scarcity, show that under certain simple conditions, the path of consumption of a purely exhaustible natural resource under competitive market conditions in fact coincides precisely with the socially efficient path of depletion.

Many other studies have attempted to reconcile the Hotelling rule result with data on the realized trajectory of natural resource prices. [7] In the specific context of forestry, Livernois et al.

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[7]Hassler et al. (2016) provide a modern treatment of the Hotelling result and a review of the empirical evidence, focusing primarily on the prices of metals and fossil fuels. They note key differences in the observed price trajectories of these resources from the classic Hotelling result—they have primarily remained constant or even decreased over the long run—and propose several explanations for this divergence, including gradual discovery of new resource deposits, uncertainty over resource stocks, technological change, and a rolling planning horizon on the part of extracting agents (Spiro, 2014). Similarly, Anderson et al. (2018) study the responsiveness of oil production to oil prices in Texas, finding that the data are consistent with a modified version of the Hotelling model in which the drilling of new wells—rather than overall production levels—responds to price shocks in a manner predicted by the Hotelling

([2006](#)) conduct a direct test of the Hotelling rule, 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 federal forest land. The authors develop a modified version of Hotelling's theory and, in most of their specifications, fail to reject it.

Several studies, by the end of the 1970s, had concluded that the world was not "running out" of natural resources in an economic sense—after accounting for technological change and after appropriately discounting the future—and that the future of economic growth did not look as dire as forewarned in the Club of Rome study, as Weinstein and Zeckhauser ([1975](#)) and Peterson and Fisher ([1977](#)) highlight. Input-saving technological change, which would respond to the endogenously rising prices as predicted by Hotelling, has emerged as a particularly important avenue of research, as many studies have framed it as a potentially important mechanism through which the adverse effects of resource scarcity may be mitigated (see, for example, Dasgupta and Heal, [1974](#); Hassler et al., [2021](#)).

Much of this early work on resource scarcity was motivated by non-renewable fuel resources such as oil and minerals, but it in many ways reflects the economic role that forest resources have played throughout history. For example, Rosenberg ([1973](#)) highlights the "national crisis" of timber scarcity that confronted pre-industrial England and necessitated severe restrictions on furnace operations in overly depleted areas.

Although timber is no longer viewed as a primary source of fuel and raw material in most economic contexts, 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. Nonetheless, most research following Hotelling 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 present here a simple model of *renewable* resource extraction under sole ownership. This model will allow 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 the exposition here follows that of Peterson and Fisher ([1977](#)) and Fisher ([1981](#)), who in turn build on the foundation laid by Gordon ([1954](#)), Scott ([1955](#)), and Smith ([1968](#)).[8]

We assume that forest property owners are price-takers in the market for forest products.[9] 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

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framework.

[8]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 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.

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

the owner of extracting y_t given current stock X_t.[10] 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.[11] We assume for simplicity that the interest rate r is constant and that extraction costs $c(\cdot, X_t)$ are convex in y_t.[12]

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.

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[10]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.

[11]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.

[12]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 $\int_0^\infty y_t\, dt \leq X_0$.

Figure 4 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.
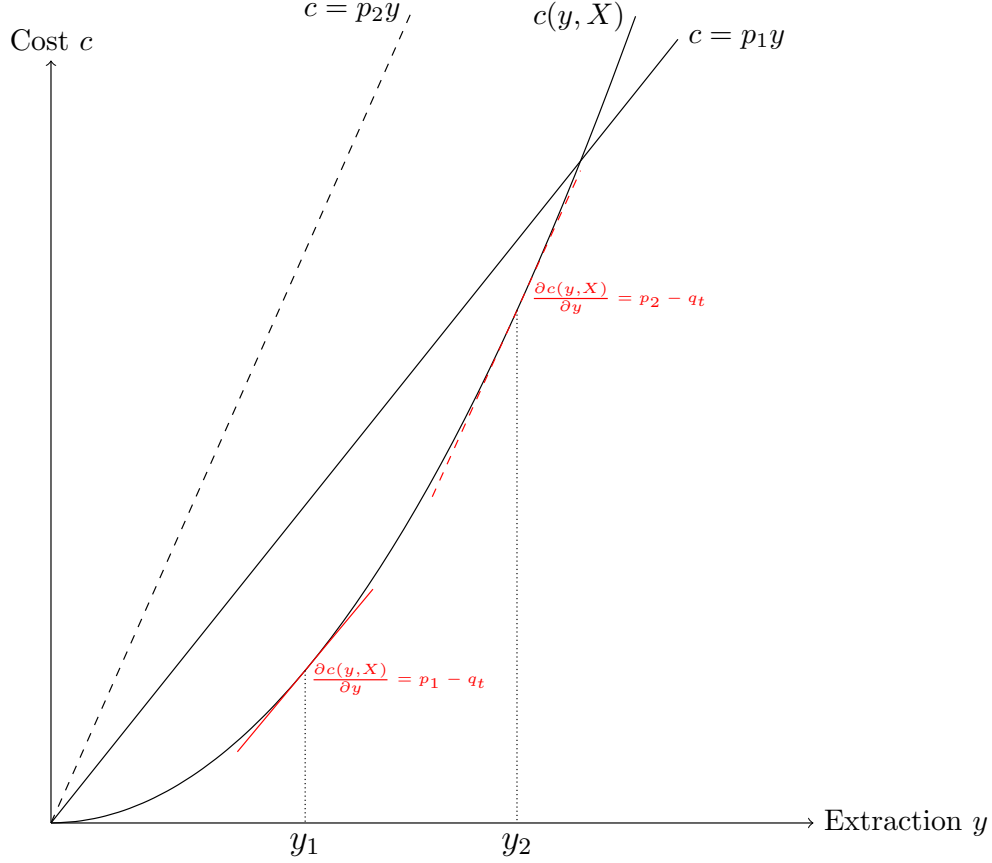

Figure 4. Sole owner's single-period extraction decision

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:

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

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

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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.[13] 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).[14]

Figure 5 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 4. 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 $g(X^*_{SO})$.

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[13]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.

[14]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.
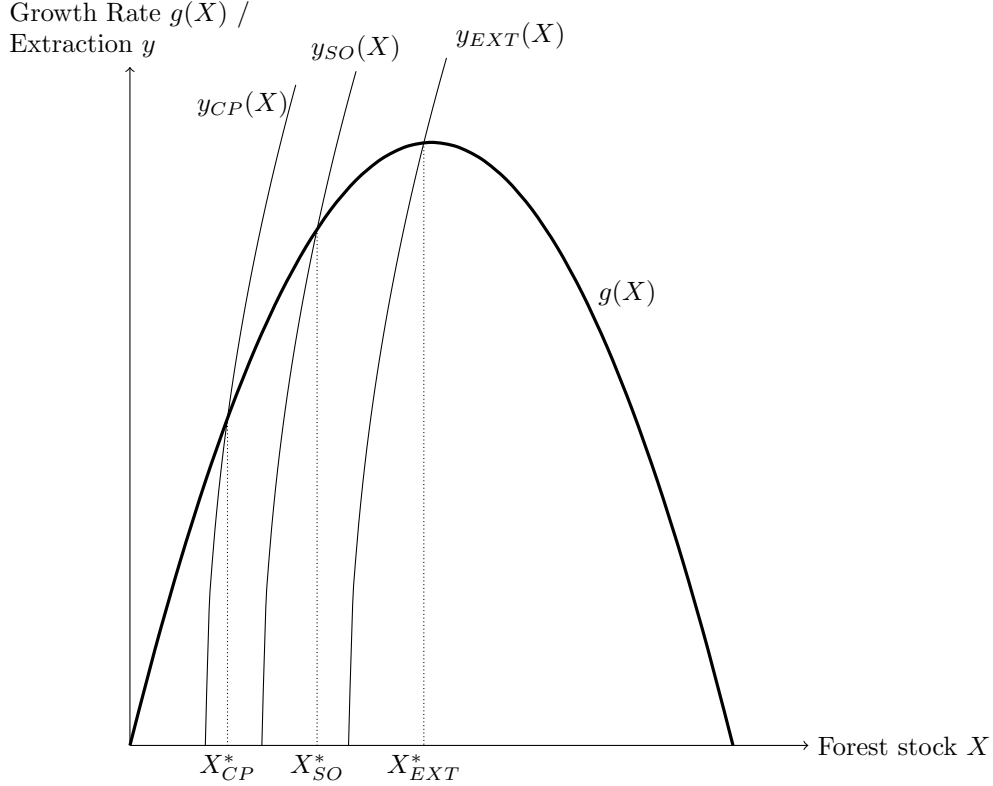
Figure 5. Steady-state extraction and forest stocks

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 5, 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.

4 Beyond the Classical Model: Extensions and Empirical Evidence

The benchmark model developed above is useful for deriving general principles of renewable resource management, but it misses several key features that characterize forestry in general and tropical

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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 such failure 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 in practice.

4.1 Modeling land use choice

One feature that is not explicitly captured in the above model, and which 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. 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 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

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 X_{max} 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.

Alongside models of optimal renewable resource depletion, other work has developed 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 forested land to clear for agriculture. A notable early example in this line of inquiry is Pfaff (1999), who was among the first to exploit satellite-based measures of forest loss in his analysis of factors driving deforestation in the Brazilian Amazon. Angelsen and Kaimowitz (1999) provide an overview of over 140 early economic models of the drivers of deforestation, including both theoretical and empirical models of alternative land uses and their predictions for deforestation levels. 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 proportional 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 across academic disciplines. For example, several studies have examined the impact of road infrastructure on deforestation in various settings (e.g., Chomitz and Gray, 1996; Pfaff et al., 2007; Vilela et al., 2020), finding that higher road density is associated with higher deforestation rates in most cases. Endogeneity of road placement presents a persistent challenge to causal interpretation, but Asher et al. (2020) employ a variety of microeconometric approaches using panel data to recover the causal effects of India's road network expansion over time. At least in their setting, the construction of new highways 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.

Openness to international trade is another form of market access that has played a particularly salient role in academic and policy discussions of deforestation. Trade openness may affect deforestation through a number of direct and indirect channels, such that the overall effect is of ambiguous sign. Early theoretical discussions focused primarily on trade's effects on timber markets, and highlighted that trade could increase or decrease deforestation depending on baseline deforestation levels and the strength of property rights. We return to a discussion of particular these issues in Section 4.3. 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 a direct effect through changes in agricultural prices, which change 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 may 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 landowner's decision as a static one. Given prevailing prices at a particular time, landowners choose the highest-return use for their land in that period. However, as emphasized by Scott (2014), 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 and taking into account the fact that if they do not change their land's use today they reserve the option to do so in the future.

Recent methodological advances in the empirical industrial organization literature have provided tools to specify and estimate discrete-choice models of agricultural land use, including those that incorporate dynamics. Scott (2014) provides an important contribution, developing an Euler equation-based empirical framework that can incorporate unobservable heterogeneity across plots. Using this 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 defor-estation 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 that price would affect demand in a particular location. For example, Assunção et al. (2015a) examine the impacts of crop and cattle prices on deforestation in Brazil. They use prices obtained from elsewhere in Brazil (in the state of Paraná, located outside the Amazon) and interact this with a fixed, cross-sectional measure of these outputs in a given municipality measured at baseline. Using this approach they find that deforestation increases when output prices increase.

Similarly, several analyses have documented that increases in the global price of palm oil led to substantial increases in oil palm cultivation in Indonesia, using for identification the global palm oil price interacted with some type of cross-sectional measure of suitability for oil palm cultivation. Edwards (2019) applies a reduced-form version of this strategy, similar to Assunção et al. (2015a), although his main outcome of interest is poverty rather than forest cover per se. Hsiao (2021) takes a more structural approach, modeling palm mill owners' decisionmaking with an inter-temporal 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 such as firewood, furniture, and paper. They further argue that, over the time period they study, India was essentially closed to global trade in timber products, and therefore 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. Souza-Rodrigues (2019) takes an alternative approach based on market access. He uses cross-sectional variation in access to the Brazilian transportation network (for which he instruments, in turn, with straight-line connections between state capital cities and nearest ports) 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 transporta-tion costs. He shows that greater road access leads to greater deforestation and more farming, which he interprets through his model as a responsiveness to net agricultrual prices received. Ob-served deforestation, as recorded in the Brazilian Agricultural Census, thus allows him to back out farmers' value of agricultural (i.e., deforested) land. Having estimated the structural param-eters of landowners' decisionmaking 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 regression analysis and finds some evidence that lower trade barriers may have increased deforestation (as measured over the period 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) entering into force. They argue that whether an RTA is signed and when all parties ratify it are both highly uncertain outcomes, leaving little scope for confounding anticipatory factors. They verify this hypothesis by running event study regressions, using a panel of 189 countries spanning 2001–2012, which allow them to test for (and rule out) pre-ratification trends in deforestation. Their event study estimates suggest that deforestation rates cumulatively increased by between 19% and 26% over the three years following ratification of an RTA and, moreover, that the effect is driven almost entirely by developing countries in the tropics. 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 post-RTA deforestation increases.

Productivity. While increases in output prices should lead to increases in deforestation, it is less clear *ex ante* how changes in agricultural productivity will affect deforestation. If demand for the agricultural good in question is inelastic, productivity increases could lead to a similar (or higher) level of output being produced on less land. This is the so-called "Borlaug hypothesis," articulated by Nobel Peace Prize winner Norman Borlaug: agricultural technological improvement may 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 regional or local level, or at the level of 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, using the empirical variation in hydro-power based electrification from Lipscomb et al. (2013). Szerman et al. show that electrification increased productivity in agriculture, for example, by enabling temperature-and-humidity controlled storage facilities, and by facilitating the use of electrical pumps for irrigation. By contrast, they show little impact on livestock productivity. They then show empirically that this led farmers to switch from livestock to crops. Because livestock is much more land-intensive than crops, they find that, in some of their measures, deforestation ultimately declined. Similarly, Abman et al. (2020) find that the introduction of high-yield variety (HYV) seeds and agricultural training by an NGO in Uganda reduced deforestation relative to nearby villages which did not receive the program, 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.

Credit constraints and cash transfers. In the presence of credit and liquidity constraints, which are common in many developing country settings, switching costs between different land uses may be particularly salient: even if an alternative land use is more profitable than the current use, 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 new restrictions were binding, so rural credit concessions decreased substantially; as a result, the policy change reduced the cumulative deforested land by up to 60% over a nine-year period. Note that this study bundled credit with explicit environmental compliance rules; it would be useful to study the impact of a pure credit constraint choice that was not explicitly linked to environmental compliance.

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 productivity (and hence income) was otherwise low. On the other hand, cash transfer programs may induce deforestation effects through the channel of 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 (especially in localities that were relatively less market-integrated). 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, e.g., for farms or pasture. 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. Moreover, landowners may derive nonpecuniary benefits from different land use choices, and there may be unobservable heterogeneity in the returns to different land uses and in switching costs. Agricultural output prices alone, therefore, may not accurately capture landowners' values of alternative land uses relative to conservation. 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. The landowner's choice problem includes considering the value of land use for crops or pasture and the potential value of retaining the forest. 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 problem formulated in Section 3; 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. Discrete-choice frameworks are also readily estimable using the type of rich micro-data on deforestation discussed in Section 2 combined with cross-sectional differences across pixels.

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). As the authors discuss, 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. Similarly, agents' value of conserving a given parcel of land may depend on the conservation status of neighboring parcels, whether due to strategic interactions between landholders or to effects on the probability of species survival (Albers et al., 2008; Lewis et al., 2011). In the language of the discrete-choice framework, land use choices made in one parcel affect the profit functions and switching costs in other parcels. As noted by Scott (2014), such cross-parcel interactions have not yet been tackled in the agricultural discrete-choice literature and remain a potentially fruitful area for future methodological innovation.

In summary, models of optimal renewable resource extraction and discrete-choice-based models of alternative land uses capture different aspects of deforestation. While the former presents a theoretically tractable optimal control framework that can easily accommodate the biological features of forest growth, the latter presents an empirically tractable framework that is suitable for recovering structural parameters of landowners' decision making—and hence for evaluating counterfactual policies, as in Souza-Rodrigues (2019), Araujo et al. (2020), and Hsiao (2021).

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 decrease the amount of deforestation in equilibrium. Even if the extraction of forest resources were shown not to be a hindrance to future growth, one particular reason that a social planner might still decrease deforestation is that it entails meaningful social costs that are external to the agents directly engaged in forestry. We next consider such deforestation externalities in the context of our stylized model of renewable resource extraction.

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, although Samuelson (1976) anticipates their possibility. He 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; that is, socially optimal forestry might yield a larger standing forest in the steady state. Another key distinction between tropical forests and other natural resources is that forests provide amenity services in addition to extractive benefits.[15] 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

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[15]Samuelson (1976) speculates that the consumption value of forest may vastly outweigh its existence value at low levels of income. This argument can be seen as an early precursor of the Environmental Kuznets Curve hypothesis.

discovery and which a single owner would not internalize when solving for their optimal harvesting plan.[16]

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

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).[17]

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.[18] 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.

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[16]A distinct body of research has developed that attempts 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).

[17]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.

[18]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.

The influence of externalities on the socially optimal deforestation level is depicted graphically in Figure 6. 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 4) to $c_2(y, X)$, drawn in blue. The optimal level of extraction within each period decreases accordingly, from y_1 to y_2.
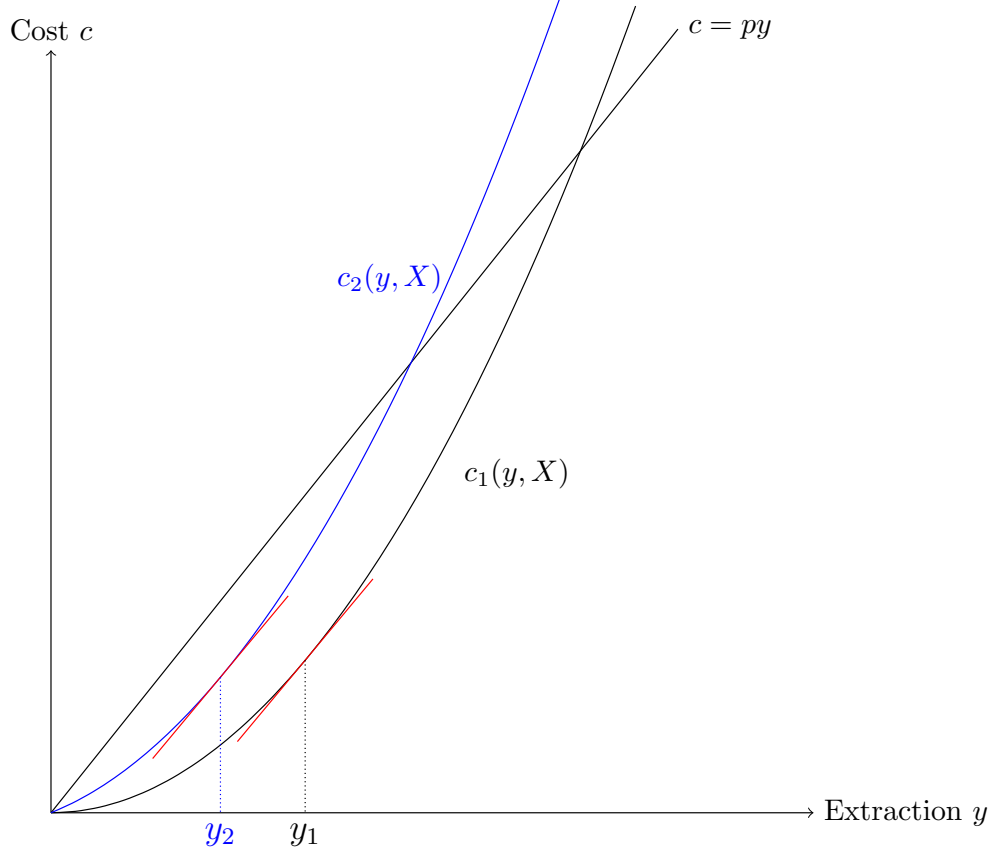

Figure 6. Optimal extraction with externalities

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 make doing so challenging.

A much more common policy 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 that Pigouvian taxes face. On the other hand, PES programs are, in practice, unable to penalize landholders for defaulting and deforesting in spite of the contract, which makes them potentially quite costly to administer at

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scale. Moreover, PES programs entail the unique challenge of identifying marginal landowners—those who would not conserve the forest in the absence of payments. We review empirical evidence surrounding this design challenge (also termed "additionality") below.

Jayachandran (2013) studies another theoretical difficulty—the role of credit constraints—in modulating the effectiveness of PES due to the voluntary nature of participation in such schemes. She argues 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 (e.g., emergency medical bills), 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 maintaining forested land.

The fact that tropical forests generate both domestic and international external benefits—and the fact that these benefits are *nonexcludable*—makes them an archetypal example of a "conservation good." Harstad (2016) develops a dynamic game theoretic model of the market for conservation goods which helps explain the prevalence and structure of PES-like contracts, including those initiated across countries. In Harstad's model, the owner ("seller") of forest property, which could be a country with a large tropical forest stand, values consumption of the forest less than an outside party ("buyer") values its conservation, and decides whether to cut the forest. Conservation is the socially efficient outcome of this game, but Harstad shows that within a broad class of equilibrium concepts, there exists no pure strategy equilibrium in which conservation is achieved. The only equilibria are in mixed strategies, which are inefficient because cutting occurs with positive probability. Harstad draws an important distinction between domestic conservation, and the case in which a coalition of developed countries, seeking to correct an international externality, attempts to prevent deforestation abroad. 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 (e.g., for foreign conservation), which reflects the actual structure of many REDD+ contracts that have emerged between developed and developing countries.

Trade policy. A recent literature has considered trade policy as an alternative tool to PES for correcting international deforestation externalities when domestic regulation is inadequate. In early theoretical work, Barbier and Rauscher (1994) develop a model of deforestation in which timber is extracted for the purpose of domestic consumption or for export. Importing countries can impose tariffs or import bans in order to increase the equilibrium stock of intact tropical forest, but Barbier and Rauscher's analysis indicates such policies may, under certain circumstances, have the opposite effect and increase deforestation. By contrast, they find that international transfers, such as PES arrangements, 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 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 for the prevention of tropical deforestation and 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 empirical evidence on the effectiveness of PES has been mixed—see, e.g., Pattanayak et al., 2010 for a review—many recent evaluations have shown 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 the treatment condition, enrollees would 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, the authors find that 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. The authors extrapolate their findings under various assumptions about program scale-up, discounting, and the social cost of carbon, and find that the social benefits of the carbon emissions averted by such a program would outweigh the fiscal costs of the transfer under most sets of assumptions.[19]

On the other hand, Edwards et al. (2022) provide less sanguine evidence from a randomized evaluation of a village-level PES program aimed at reducing forest fires for land clearing in Indonesia. Under this program, villages were paid 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 nonetheless 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. They provide evidence that the reduction was driven by recipients' increased reporting of illegal deforestation activity by others, rather than a reduction in their own deforestation activity.

Even if successful, PES programs may be vastly expensive to implement because of additionality concerns; as such, researchers have paid increasing attention to the design and targeting of PES programs. At least part of the additionality challenge arises from the fact that landowners have heterogeneous costs of compliance with the PES program's conservation requirement; 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.[20] 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).

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. More-

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[19]The only scenario in which the program does not pass a cost-benefit analysis is one in which the effects of the program dissipate immediately after the 1.5-year endline evaluation.

[20]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.

over, because landowners' compliance costs fluctuate over time in response to changing agricultural output prices, Assunção et al. (2015b) highlight that PES contracts should (but currently typically do not) respond to agricultural prices in order to flexibly accommodate landowners' changing outside option. Additional rigorous evidence on whether PES programs work, and how to design them to be even more cost-effective, is a useful dimension for future research.

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.

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. In particular, he notes that common law institutions in many developing countries favor polluters' rights (rather than "pollutees' 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.[21]

Balboni et al. (2021b), examining the strategic use of fire for land clearing in Indonesia's forest estate, provide a modern example of a setting in which a Coasean solution to deforestation externalities has not been reached. Landowners frequently use fire to clear deforested land despite the fact that doing so is illegal. Using MODIS satellite data to identify fires, the authors document that strategic fire-setting is less likely to occur on days with high wind speed—when fires are more likely to spread out of control—in pixels that are surrounded by the landowner's own land, compared to windy days when pixels are surrounded by land owned by others. In other words, landowners who clear land with fire do not internalize the risk that fire-setting poses to neighboring land the way they do their own land, suggesting a failure of Coasean bargaining to internalize externalities.

Trade policy. Hsiao (2021) quantitatively explores the precise conditions under which import tariffs can function as a Pigouvian tax, and hence can effectively 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 extensive-margin decision to construct oil palm mills and their intensive-margin decision over how much land to deforest. Incorporating dynamics into the model is important for Hsiao's analysis, as well, because imposing tariffs on palm oil 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. Estimating his model using a combination of discrete- and continuous-choice Euler methods, Hsiao finds that coordinated and committed tariffs can nearly replicate the deforestation reductions achieved by a first-best domestic tax, adding that coordination can be achieved through transfers among importers and commitment through strong institutions.

Dominguez-Iino (2021) also evaluates the effectiveness of import tariffs as a Pigouvian mechanism. He 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

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[21]Moreover, enforcement of property rights tends to be weak, a point we return to in the following section.

to Hsiao's finding on the need for coordination, he concludes that tariffs in the South American context 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, whether or not centralized government policy instruments exist simultaneously. 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" that emerged in Brazil between 2007 and 2015. As part of these agreements, meatpacking companies agreed 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 to evaluate their effects on deforestation rates in two Brazilian states in the Amazon. Despite widespread adoption of such agreements, the authors find that they had no aggregate effects on deforestation in their sample, and that this null result may have been the 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 (FSC) certification in Mexico and find no significant effects, though they note that FSC certification may still have impacts on key unobserved outcomes such as forest degradation.

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 in the Amazon and did not lead to leakage of deforestation activity, exploiting a triple-differences framework that compares trends in forest cover in areas suitable for soy cultivation relative to soy-unsuitable areas in the Amazon. In light of these mixed results, identifying precise conditions under which such non-state deforestation commitments may effectively curb deforestation activity (without causing leakage to unmonitored areas) 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: in order for the path of extraction derived in that section to be optimal, the agent must expect to receive the future stream of returns to which they are entitled (Scott, 1955). 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 in practice.

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 foresting firms is such that the marginal yield to effort on each forest parcel is equalized. Under a common property regime, however, this allocation cannot be an equilibrium, because any individual firm could do better by instead moving to a parcel with higher *average* yield. Moreover, as long as average yield on the least fertile parcel exceeds marginal costs of extraction effort, the prospect of positive rents induces additional firms to enter. The key force in Gordon's model is free entry of agents who cannot expect to appropriate profits in any future period: rents from forest ownership are thus dissipated and profits in each time period are competed down to zero.

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 Section 3. 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 7, as the common-property equilibrium occurs where the aggregate cost curve intersects the line $c = py$.
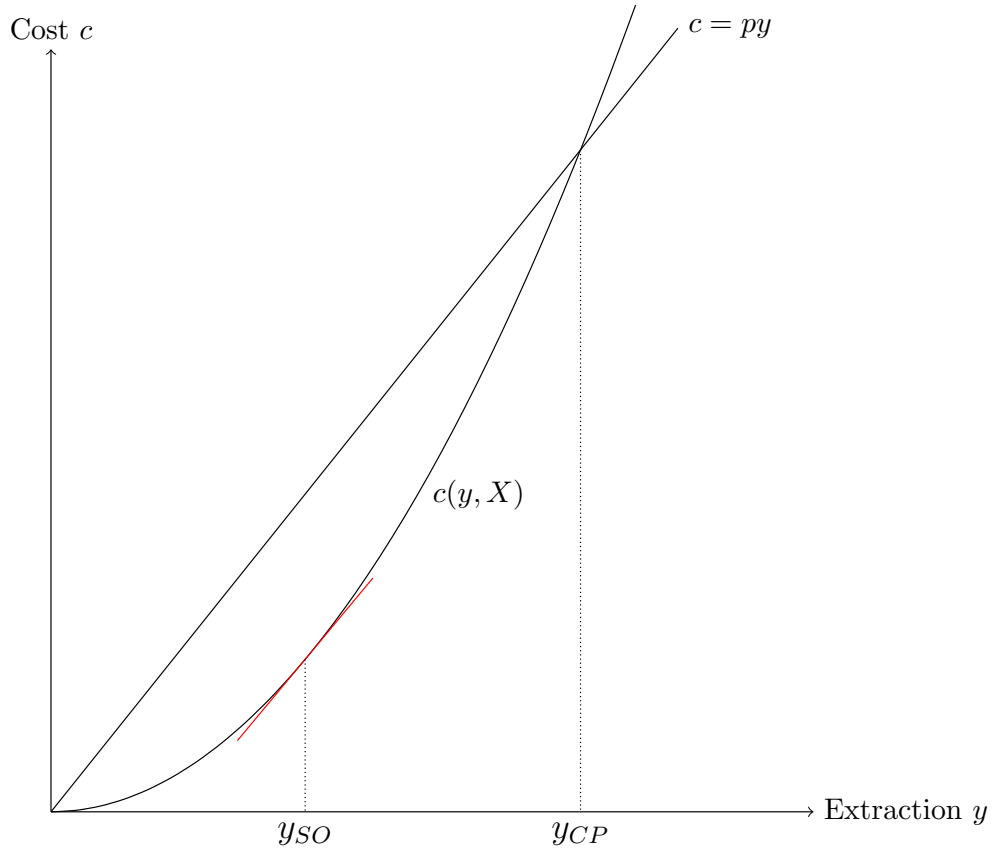

Figure 7. Common-property levels of aggregate extraction

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).

Weakly enforced property rights raise a distinct but related set of theoretical issues with nuanced implications for the design of policy.[22] 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 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, while for capital-intensive extractive resources such as petroleum, insecure property rights will lead to under-exploitation of the resource.

Policy instruments. The theory of common property resources suggests several possible solutions to overcome the tragedy of the commons and recover socially optimal forest usage. Many authors, dating back at least to Hardin (1968), have considered tax instruments. In the context of fisheries, Peterson and Fisher (1977) and Dasgupta and Heal (1979) consider how optimality could be restored under a two-part tax, one reflecting the cost of reducing the resource stock (which in turn affects its future growth) and the other reflecting the cost of crowding diseconomies in the present. Of course, such a tax would be informationally intensive and difficult to implement in practice, and indeed few examples exist of successful applications of taxes on the usage of common-property resources.

Other work has emphasized how optimal deforestation may be restored by assigning ownership to, or government appropriation of, previously open-access resources (e.g., through land titling programs in forest areas), such that the newly propertied agents internalize the marginal user cost of

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[22]Although many of the papers discussed here take the strength of property rights as given, property rights regimes are, of course, not set exogenously. A large body of literature dating back to the work of Demsetz (1967) studies the determinants and evolution of property regimes. More recently, Copeland and Taylor (2009) provide a modern treatment of endogenous property rights and discuss implications for deforestation and the extraction of other natural resources under different assumptions on primitive country characteristics.

forest exploitation.[23] Such approaches are discussed in the early work of Gordon (1954) and Scott (1955), and synthesized by Ostrom (1990) in her Nobel Prize-winning work on the topic. Assigning property rights may not be a panacea, however. Although reductions in deforestation through secure property rights are a typical feature of renewable resource models featuring common-property access, 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 as a result (Farzin, 1984; Liscow, 2013). Moreover, other authors have derived theoretical results highlighting potential adverse political economy and redistributive implications of conversion from common to private property. For instance, Weitzman et al. (1974) shows that a variable factor (e.g., labor) without property rights will be weakly worse off under private ownership than under free access to the commons, 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; 1997; 1998). 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, as in the renewable resource depletion model of Schaefer (1957). Brander and Taylor (1997a) analyze the case of a small open economy with a comparative advantage in production of the natural resource. In a two-country version of the model, Brander and Taylor (1997b) 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.[24]

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

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[23]Hardin (1968) argued succinctly, "the tragedy of the commons as a food basket is averted by private property, or something formally like it."

[24]One assumption implicit throughout most of Brander and Taylor's analysis is that both countries can produce and trade the natural resource. Framed differently, the motive for deforestation in this model is to satisfy foreign demand for timber *per se*. As discussed in Section 4.1, a major driver of tropical deforestation is land conversion for agricultural production, which is not captured in these models.

theory relating to property rights allocation and formalization. The findings in this empirical literature are mixed. 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). The authors provide suggestive evidence that land titling may have 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 as measured in 2001, consistent with the presence of "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. 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 mainly in regulating firewood and fodder extraction ("lopping"), but that Van Panchayat-managed forests were otherwise observably similar to other, nearby forests. 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% based on satellite data from Song et al. (2018).

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. Moreover, in a recent experimental evaluation of a village-level community monitoring program in Uganda Eisenbarth et al. (2021) find evidence that such community monitoring 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 seems like an important area for future research.

4.4 Political economy determinants of tropical deforestation

Our theoretical analysis to this point has primarily 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 differ from what a social planner would implement when the externalities of deforestation are properly internalized. However, political economy considerations—such as the influence of pro-exploitation lobbies—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 technological and financial 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 government administrations presiding over an exhaustible resource.[25] 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 extract a unit of timber (or to expand agricultural production) 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 returns may lead to large changes in deforestation levels. Harstad also conducts a normative analysis of cost-effective conservation policy, deriving conditions under which it is more cost-effective for a pro-conservation donor to pay the pro-exploitation interest *not* to lobby, as well as when it is more efficient to provide payments to the general public rather than directly to the presidential administration.

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

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[25]Harstad's main analysis focuses on the case of a purely depletable resource, but his main results hold for renewable resources so long as the resource does not fully recover.

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 aggregate 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 there less lucrative, thus reducing enforcement costs for neighboring districts.

The effect of administrative centralization in this model is thus theoretically ambiguous: district governments either extract less or more timber than a centralized social planner would implement, depending on the strength of institutions governing local timber markets. By the same mechanism, this model also has implications for the optimal design of conservation contracts between a donor and a country whose forest they seek to conserve. Harstad and Mideksa's results suggest that a conservationist donor should contract with a central government when most logging occurs legally and directly with individual district governments when local enforcement costs to prevent illegal logging are large.

4.4.2 Empirical approaches

National conservation policies. Burgess et al. (2019) provide evidence supporting the importance of national forest management policy in the Brazilian Amazon, constructing a spatial

regression discontinuity design that exploits the discrete shift in policy regime that occurs at the national border between Brazil and its Amazonian neighbors. Using a panel of Landsat 7 image data from Hansen et al. (2013), 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 deforestation rates increased again in 2014 during a time of weakened environmental regulation. The authors' main conclusion is that the Brazilian national policy regime with respect to environmental protection had a meaningful impact on both conservation and exploitation of the Amazon from 2000–2018, including in remote areas along the border where weak property rights and imperfect enforcement may have cast doubt *ex ante* on the effectiveness of national conservation policies.

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 deforestation 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, given a government's objective function (e.g., minimizing carbon emissions or area deforested) and an constraint on the number of districts or total area that can be monitored. They find that the priority list that was actually implemented led to roughly 12 percent higher carbon emissions than the optimal priority list under their framework.

Mangonnet et al. (2022) focus in particular on the national-level processes that influence which land is actually conserved under conservationist 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.[26] 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. They provide suggestive evidence that this effect is not driven by an attempt to garner support from voters in opposition areas (i.e., by creating unfavorable economic conditions that voters may attribute to local politicians), but rather 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, uses satellite imagery to detect changes in tropical forest cover and alert environmental law enforcement authorities

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[26]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.

accordingly. In order to evaluate the effectiveness of this technology, 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 by the environmental authority in that municipality. 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 spanning 2006–2016. The authors note that they are not able to distinguish whether their estimated effects are driven by offenders updating their beliefs about the probability of detection or by a reduction in logging firms' deforestation-specific capital as a result of being fined.

Interactions with elections. As discussed above, illegal forest extraction may follow an electoral cycle as local officials become relatively more permissive of illegal activity in years leading up to a local election. Pailler (2018) provides a thorough discussion of forces that may give rise to a political deforestation cycle in the context of the Brazilian Amazon. She argues that tropical forests are particularly well suited to cyclical exploitation: deforestation leads to short-run economic benefits within a geographically concentrated area and longer-run, geographically diffuse costs, for which voters are less likely to punish local officials. Facilitating greater logging activity in election years may therefore improve the electoral performance of incumbents.

Studies testing for electoral deforestation cycles typically employ locality-level panel data and difference-in-differences specifications to compare within-locality trends in deforestation and their coincidence with election timing. Pailler adopts this strategy to estimate the effect on deforestation rates when a municipality's mayor runs for re-election relative to municipalities where the incumbent does 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 also notes that deforestation may even be a mechanism for funding electoral campaigns directly: self-funded campaign contributions for incumbents are positively correlated during her sample period.

Of course, an empirical difficulty is the potential endogeneity of incumbents deciding to seek re-election. Subsequent studies have surmounted this challenge by exploiting idiosyncratic variation in the timing of local elections to illustrate the coincidence of elections and deforestation activity. 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. Cisneros et al. (2021) exploit this variation to test for the presence of a land-clearing cycle for palm oil plantations from 2001–2016. They find that, on average, deforestation rates increased by about 4% in the year leading up to a district head election. Moreover, at least in this setting, 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, as proxied by soil suitability for growing oil palm. Balboni et al. (2021a), using the same MODIS satellite data on Indonesian forest fires as Balboni et al. (2021b), find that the incidence and physical scope of forest fires, which are often illegally used as a means of land clearing, declined significantly in district head election years from 2005–2014. This result—a decrease in environmental degradation during election campaign years—contrasts with Pailler's (2018) and Cisneros et al.'s (2021) findings of cycles in which deforestation rates *increase* during election years. The divergence potentially stems from the conspicuous nature of fire-setting: forest fires create air pollution and other salient, short-run negative externalities, while illegal logging imposes external costs that are longer term and more diffuse.

Cross-jurisdiction interactions. Burgess et al. (2012) provide direct empirical evidence of local officials' economic incentives to permit deforestation activity and of pecuniary externalities across districts. To do so, they exploit the occurrence of administrative splits in local district governments, which occurred at different times in different Indonesian provinces, that led to an increase of approximately 65% in the number of districts in Indonesia's main forest islands between 2000 and 2008. The authors find that the creation of an additional district within a market (operationalized by province boundaries) led to an 8.2% increase in the deforestation rate within that province, as measured by MODIS satellite imagery data, and that this increase occurred both in areas where logging is legally sanctioned and in areas in which it is illegal. This increase in deforestation activity was 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. The authors provide additional evidence for this prediction, 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, as the authors emphasize, decentralization of forest management to local communities may not necessarily lead to lower deforestation rates in settings where local elites can collect rents from their control over 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 electorally in response to perceived corruption and the degradation of public goods. As a result, they show that district splits that resulted in greater within-district ethnic homogeneity led to a decrease in deforestation within those districts and, subsequently, a higher probability of re-election of incumbent district heads. However, the magnitude of this effect is ultimately outweighed by the positive deforestation effects of increased Cournot competition between districts 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 been mainly 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 understanding the complex and fast-moving land use changes revealed by the data. 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 on the economics of tropical deforestation.

There are many fronts on which further progress in this research area can be made. To conclude our review, we point to some key areas where we believe the need for further 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, 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. Landowners will deforest when the value they derive from keeping forest intact is below the value they gain from converting it to an alternative use. 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 environmental services considers how to calibrate, finance, and structure these types 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. There is thus 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. For example, governments could bring 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. Understanding which reforms may make them more accountable to domestic and international citizens who favor conservation constitutes an important area for future work.

Finally, a 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—without relying on the actions of local policy

makers or bureaucrats—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. Such policies may play an important role in better aligning domestic conservation policies with global social costs.

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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