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# Limits to Substitution between Ecosystem Services and Manufactured Goods and Implications for Social Discounting

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**Abstract:** This paper examines implications of limits to substitution for estimating substitutability between ecosystem services and manufactured goods and for social discounting. Based on a model that accounts for a subsistence requirement in the consumption of ecosystem services, we provide empirical evidence on substitution elasticities. We find an initial mean elasticity of substitution of two, which declines over time towards complementarity. We subsequently extend the theory of dual discounting by introducing a subsistence requirement. The relative price of ecosystem services is non-constant and grows without bound as the consumption of ecosystem services declines towards the subsistence level. An application suggests that the initial discount rate for ecosystem services is more than a percentage-point lower as compared to manufactured goods. This difference increases by a further half percentage-point over a 300-year time horizon. The results underscore the importance of considering limited substitutability in long-term public project appraisal.

**JEL-Classification:** Q01, Q57, H43, D61, D90

**Keywords:** Limited substitutability; Dual discounting; Ecosystem services; Subsistence; Project evaluation; Sustainability

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# 1 Introduction

This paper studies limits to substitution between ecosystem services and manufactured consumption goods. In particular, we examine implications of a subsistence requirement in the consumption of ecosystem services for the empirical estimation of the elasticity of substitution and for the theory and practice of social discounting.

Limited substitutability is a core issue in the debate on sustainable development and of central importance for environmental and resource management (Gerlagh and van der Zwaan 2002; Neumayer 2010). The issue of substitutability has recently received prominent attention in the discussion of dual discounting between ecosystem services and consumption goods and the economic appraisal of climate change. The debate following the Stern Review (2007) has triggered a rethinking of an adequate specification of the damages function from climate change (Weitzman 2010), relating in particular to the adverse impact of climate change on ecosystems and their repercussions for human well-being (Heal 2009a,b; Neumayer 2007). Sterner and Persson (2008) have shown that the evaluation of climate damages to utility crucially depends on assumptions regarding substitutability. If future comprehensive consumption is – as is the case in most integrated assessment models – not valued in relative prices that account for changing relative scarcities between ecosystem services and manufactured goods, a uniform social discount rate is inappropriate and dual discounting should be adopted (e.g. Baumgärtner et al. 2015a; Gollier 2010; Guesnerie 2004; Hoel and Sterner 2007; Traeger 2011; Weikard and Zhu 2005). This is the case when the two goods are less than perfect substitutes and have diverging growth rates. Baumgärtner et al. (2015a) estimate that the difference in good-specific discount rates, also termed ‘relative price effect’ (Hoel and Sterner 2007), amounts to almost a full percentage point.

A survey of experts on social discounting by Drupp et al. (2015) revealed that considering limited substitutability between ecosystem services and manufactured goods is one of the most mentioned issues lacking in current discounting guidance. Among the experts’ suggestions on how to improve existing criteria for intergenerational decision-making are approaches that “set limits in physical terms to the future development that

must not be exceeded for reasons of intra- and intergenerational justice [and] use a discounted utilitarian approach to optimize development only within these limits” (Drupp et al. 2015: 24). A recent proposal on considering such an absolute limit within the literature on substitutability has been to consider a subsistence requirement or survival threshold in the consumption of ecosystem services (Heal 2009a,b; Baumgärtner et al. 2015b). At the most basic level one may think of such a subsistence requirement in terms of minimum consumption levels of water, food and life-enabling ecosystem conditions. Specifically, Baumgärtner et al. (2015b) extend the usual constant-elasticity-of-substitution (CES) framework and show that the elasticity of substitution is non-constant if there is a subsistence requirement in the consumption of ecosystem services.

A key obstacle to advancing the discussion on limited substitutability – in consumption as well as production – has been the scarcity of empirical evidence (Ayres 2007; Neumayer 2010; Stern 1997). Recently, Yu and Abler (2010) and Baumgärtner et al. (2015a) have drawn on a direct relationship within the CES utility specification between the income elasticity of willingness to pay (WTP) for ecosystem services, a fundamental substitutability parameter and the elasticity of substitution between ecosystem services and manufactured goods. They provide estimates for the elasticity of substitution of 5 and 2.63 respectively, suggesting a high degree of substitutability.

This paper makes a twofold contribution to the literature: First, it provides the most comprehensive account of empirical evidence on substitution elasticities between ecosystem services and manufactured goods to date and analyses what limits to substitution imply for the estimation of substitutability. Second, it scrutinises implications of an absolute limit to substitution in form of a subsistence requirement for social discounting.

For our analysis of empirical evidence on substitutability between ecosystem services and manufactured goods we employ and extend the model framework of Baumgärtner et al. (2015b). The subsistence requirement in the consumption of ecosystem services allows for considering absolute limits to substitution. Furthermore, we draw on the direct relationship within the CES case between the income elasticity of WTP for ecosystem services, the substitutability parameter and the elasticity of substitution. While the elasticity of substitution is non-constant when considering a subsistence requirement,

the direct relationship between the income elasticity of WTP and the substitutability parameter remains. We provide empirical evidence on substitution possibilities based on income elasticities of WTP from 18 valuation studies. The resulting substitutability parameters range from -0.16 to 0.86. In the CES case, which holds in cases where there is an abundant provisioning of ecosystem services or ecosystem services are not required to meet subsistence needs, this would imply elasticities of substitution of 0.86 to 7.14. Simple aggregation over all studies yields a mean substitutability parameter of 0.57. We assume that 15 percent of the initial consumption of ecosystem services is required to satisfy subsistence needs and use an empirically-founded scenario based on Hoel and Sterner (2007) and Baumgärtner et al. (2015a) to estimate the time development of the elasticity of substitution for the empirical sample mean. We find an initial value for the elasticity of substitution of 2, which decreases to 1.56 (0.66) over 150 (300) years.

Subsequently, we explore implications for social discounting of considering absolute limits to substitution in the form of a subsistence requirement. We extend the standard dual discounting model (Hoel and Sterner 2007; Traeger 2011) through introducing a subsistence requirement in the consumption of ecosystem services and analyse the difference between the good-specific discount rates (relative price effect). We show that the relative price effect crucially depends on the consumption of ecosystem services relative to the subsistence requirement. It converges to the CES case, which is determined by the substitutability parameter and the difference in growth rates, if the consumption of ecosystem services grows over time and becomes plentiful. If, in contrast, the consumption of ecosystem services is limited, the relative price effect is non-constant and depends on the consumption of ecosystem services relative to the subsistence requirement. Specifically, if ecosystem services are in a continuous decline towards the amount required for subsistence, the relative price effect increases without bound.

We estimate the relative price effect for all empirical estimates of the substitutability parameter using the same scenario analysis and find that the initial discount rate for ecosystem services is on average 1.1 percentage-points lower compared to the rate for manufactured goods. This difference increases by more than half a percentage-point on average in the longer run due to the subsistence requirement. The case for including

ecological discount rates in project evaluation is therefore stronger on average than previously suggested (Baumgärtner et al. 2015a). A sensitivity analysis further reveals that the longer-term relative price effect is very sensitive to assumptions on the level of ecosystem services required for subsistence.

Overall, we find that it is crucial to consider limits to substitution for the estimation of the elasticity of substitution between ecosystem services and manufactured goods as well as for social discounting. A subsistence requirement in the consumption of ecosystem services lowers substitution possibilities and, if ecosystem services are declining, leads to an increasing relative price of ecosystem services over time. This has important implications for project evaluation and calls for safeguarding critical ecosystem services.

## **2 Substitutability of ecosystem services**

### **2.1 Ecosystem services and human well-being**

Human well-being fundamentally depends on the vitality of ecosystems and the ecosystem services (ES) they provide (Millennium Ecosystem Assessment (MEA) 2005). While ecosystem services are necessary constituents of human well-being (Dasgupta 2001), there is substantial uncertainty and disagreement regarding their importance compared to manufactured goods and how their provisioning will evolve over time. The MEA (2005) suggests that 15 of 24 ecosystem service categories have deteriorated, while only four have improved. Baumgärtner et al. (2015a) estimate, under the assumption that the single ecosystem service components are part of a homogenous good and using unweighted arithmetic means, that global ecosystem services had an average annual loss rate of 0.52 percent for the period 1950 to 2010. There are a few indicative clues for the relative contribution of ecosystem services to human well-being. A lower bound estimate might be the value of food production, which was about 3 percent of global GDP in 2000 (MEA 2005: 6). The economics of ecosystems and biodiversity (TEEB) initiative (ten Brink 2011) has calculated that ecosystem service related to agriculture, forestry and fisheries contribute between 15-20 percent to an adjusted GDP in Brazil,

India and Indonesia. This is close to evidence cited in Dasgupta (2010: 7), where the depreciation of forest, soil and fishery resources in Costa Rica amounted to about 10 percent of GDP. Overall this suggests that the current value-share of ecosystem services in overall global consumption might be in the range of 3-20 percent.

## 2.2 Conceptualising and modeling limits to substitution

The question of how limited substitution possibilities between ecosystem services/natural capital and manufactured goods/capital are is of fundamental importance within environmental and resource economics, particularly for the debate on strong versus weak sustainability (Gerlagh and van der Zwaan 2002; Neumayer 2010). This discussion has been predominantly set within a CES framework and thus focussed on the value of a single parameter. Yet, the notion of the elasticity of substitution is much more general and it is well-known that it need not be constant (Revankar 1971). Conceptual discussions of limits to substitution have concluded that, while there may currently still exist ample substitution possibilities at the margin, “limits to substitutability in the medium term at least are real and important” (Ayres 2007: 115); They further argue that these limits will be strictly binding in the long-run due to thermodynamic limits in the production process or subsistence requirements (Ayres 2007; Ehrlich 1989; Heal 2009a,b; Stern 1997).<sup>1</sup> Similarly, Fenichel and Zhao (2014) note that “[in] the limit, society will not be able to substitute away from clean water, clean air and a number of other natural resources”. Even if specific services were substitutable with technologies, human beings may still object to substitute them. We thus follow Traeger (2011) in assuming that limited willingness to substitute in consumption is the ultimately relevant constraint.

To allow for an absolute limit to substitution, we consider a subsistence requirement in the consumption of ecosystem services. In the related literature, Pezzey and Anderies (2003) conceptualise subsistence in terms of minimum nutrition levels, whereas Heal

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<sup>1</sup>That is: even though many parts of natural capital and ecosystem services may be replaceable by technology, Fitter (2013) argues that a number of supporting services (soil formation, water cycling etc.), selected final services (e.g. climate regulation) and goods (e.g. water supply, a safe and enjoyable environment) may be very hard if not impossible to fully substitute (cf. Ayres 2007).

(2009a: 279) notes that “[t]here is a minimum level of ecosystem services needed for survival—think of this as water, air, and basic foodstuffs”. Other discussants refer to “essential lifeservices” provided by ecosystems (Traeger 2011: 217) or remark that “a large part of what the natural environment offers us is a necessity” (Dasgupta 2001: 125). While there is no generally accepted notion of subsistence (Steger 2000), Sharif (1986: 555) argues that it must go beyond a mere consideration of basic physical needs to encompass the “needs [concerning] physical and mental survival”. Along these lines, such a subsistence requirement would not only include food, water, bodily security and more broadly life-enabling ecosystem conditions, but also cultural ecosystem services like the experience of ‘naturalness’ or the existence of sacred natural environments. Relatedly, Baumgärtner et al. (2015b: 2) define a subsistence requirement more generally to “encompass a homogeneous composite good to which an individual attaches absolute priority before considering trade-offs with other goods”. Here, we predominantly focus on a narrower definition of subsistence that includes essential services related to the consumption of water, food, air as well as life-enabling ecosystem conditions.

The modelling set-up follows Baumgärtner et al. (2015b) in extending the CES utility specification by introducing the idea of a subsistence requirement in the consumption of ecosystem services.<sup>2</sup> A representative agent’s well-being is determined by the consumption of two goods – a market-traded private consumption good  $C$  and a public ecosystem services  $E$ . The agent requires an amount  $\bar{E}$  of the ecosystem service to satisfy her subsistence needs. Her preferences are represented by a utility function

$$U(E, C) = \begin{cases} U_i(E) & \text{for } E \leq \bar{E} \\ U_h(E, C) & \text{else} \end{cases} \quad (1)$$

where  $U_h(\cdot, \cdot)$  is a twice continuously differentiable function which is strictly increasing in both arguments and strictly quasi-concave. In the following, we only consider utility in the domain where the subsistence requirement is met, i.e.  $E > \bar{E}$ . A suitable specification for  $U_h(\cdot, \cdot)$  is a generalized modification of the Stone-Geary and the CES

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<sup>2</sup>Alternatively, Dupoux and Martinet (2014) propose to examine Edgeworth-Pareto substitutability by introducing a specific ‘context-dependent substitutability function’.



functions (cf. Heal 2009a,b; Baumgärtner et al. 2015b):

$$U_h(E, C) = \left[ \alpha (E - \bar{E})^\theta + (1 - \alpha) C^\theta \right]^{1/\theta} \quad \text{with } -\infty < \theta \leq +1, \theta \neq 0; 0 < \alpha < 1, \quad (2)$$

where  $\theta$  is the substitutability parameter.<sup>3</sup>

First, we recover the standard CES case, with  $\bar{E} = 0$ , that dominates discussions on substitutability. In this case, the elasticity of substitution  $\sigma$  is solely determined by the exogenous substitutability parameter  $\theta$  as follows:

$$\sigma = \frac{1}{1 - \theta}.$$

The CES representation contains the special cases of perfect substitutes ( $\theta = 1; \sigma = \infty$ ), and perfect complements ( $\theta \rightarrow -\infty; \sigma = 0$ ), where the Cobb-Douglas case ( $\theta = 0; \sigma = 1$ ) marks the demarkation line between substitutes and complements.

For the utility function (2), the elasticity of substitution approaches the standard CES form in three special cases only (Baumgärtner et al. 2015b: 6):

$$\sigma(E, C) \rightarrow \frac{1}{1 - \theta} \quad \text{for} \quad \begin{cases} E \rightarrow \infty \\ \bar{E} \rightarrow 0 \\ E \rightarrow \bar{E} \text{ and } \theta < 0. \end{cases}$$

First, substitutability approaches the standard CES case in a ‘world of plenty’ where there is no limit to the consumption of ecosystem services. Second, it naturally collapses to the CES case when the subsistence requirement plays no role. Third, it approaches the CES case when the two goods are complements and the consumption of ecosystem services approaches the subsistence requirement. In general, however, the elasticity of substitution is non-constant and lower than under the CES case. For the relevant case where the two goods are initially considered substitutes in a ‘world of plenty’, it monotonically decreases as the consumption of ecosystem services decreases towards the subsistence requirement. Considering such a subsistence requirement provides a richer basis for discussing issues of limited substitutability. However, it complicates an empirical estimation of the elasticity of substitution.

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<sup>3</sup> The extension of  $U_h(E, C)$  for  $\theta \rightarrow 0$  is a special Stone-Geory case:  $U_h(E, C) = (E - \bar{E})^\alpha C^{(1-\alpha)}$ .

## 2.3 Estimating the substitutability of ecosystem services

The substitutability between ecosystem services and manufactured goods, respectively income, can in principle be elicited using various non-market valuation techniques, such as choice experiments, contingent valuation or revealed preference approaches.<sup>4</sup>

The only estimation route that currently provides estimates of the (constant) elasticity of substitution is a recently rediscovered way of indirectly inferring it through its relation to the income elasticity of WTP (Baumgärtner et al. 2015a, 2016; Yu and Abler 2010). Based on Kovenock and Sadka (1981), Ebert (2003) has shown that the income elasticity of WTP for the ecosystem service  $\xi$  has an inverse relationship to the elasticity of substitution between a composite manufactured consumption good and the ecosystem service  $\sigma$  for the case of the CES utility function (i.e. Eq. (2) with  $\bar{E} = 0$ ):<sup>5</sup>

$$\xi = 1 - \theta = \frac{1}{\sigma} \quad (3)$$

leading to the conclusion that

$$\text{if } \xi \begin{matrix} \leq \\ \geq \end{matrix} 1 \quad \text{then } \theta \begin{matrix} \geq \\ \leq \end{matrix} 0 \quad \text{and thus } \sigma \begin{matrix} \geq \\ \leq \end{matrix} 1$$

i.e. the income elasticity of WTP is smaller (greater) [equal to] unity if the ecosystem service and consumption good are substitutes (complements) [Cobb-Douglas].

This straightforward relationship facilitating an indirect estimation of the elasticity of substitution does not hold when ecosystem services are required to meet subsistence needs. There is still a direct relationship between  $\xi$  and  $\theta$  in this case (see Appendix A.1 for a derivation), yet this does not directly yield an estimate of the elasticity of substi-

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<sup>4</sup> While choice experiments may constitute a suitable approach for estimating the elasticity of substitution, studies have so far not been designed for such purposes. The same is the case for revealed preference studies, with the exception of Martini and Tiezzi (2014), which we address below. Further indicative evidence might be derived from the WTA/WTP disparity (Hanemann 1991).

<sup>5</sup>This implies that both goods are ‘normal’, which may not be the case for every single ecosystem service (Horowitz and McConnell 2003).

tution, which also depends on all other ingredients of utility function (2):

$$\xi = 1 - \theta \neq \frac{1}{\sigma(E, C)} = (1 - \theta) \left[ 1 - \frac{(1 - \alpha) \frac{\bar{E}}{E}}{\alpha \left[ \frac{E - \bar{E}}{C} \right]^\theta + (1 - \alpha)} \right]^{-1}. \quad (4)$$

This indirect estimation route therefore only yields correct estimates of the elasticity of substitution in special cases: In particular, if the consumption of ecosystem services is unlimited as well as if there is no subsistence requirement. As there is still a direct relationship between the income elasticity of WTP and the substitutability parameter, we gather estimates of  $\theta$  through its relation to  $\xi$  and discuss what this implies for the elasticity of substitution. The substitutability parameter can be estimated indirectly based on data from a range of valuation studies that estimate a constant income elasticity of WTP, calculated as point-elasticities evaluated at the mean values of the income variables.<sup>6</sup> These estimates have been reported in numerous contingent valuation studies. Table 1 provides empirical evidence on the substitutability parameter for estimates provided since the year 2000 that we could obtain (see Kristrøem and Riera (1996) for earlier estimates). The income elasticity of WTP for ecosystem services  $\xi$  ranges from 0.14 to 1.16, implying values of the substitutability parameter  $\theta$  of -0.16 to 0.86. Averaging over all studies,<sup>7</sup> we find a mean (median) value for the substitutability parameter  $\theta$  of 0.57 (0.69), with a standard deviation of 0.29. In a world with an plentiful ecosystem services or without a subsistence requirement, this would imply a mean elasticity of substitution  $\sigma$  of 2.31, with a range of 0.86 to 7.14.

Although we will treat the values of the substitutability parameter contained in Table 1 as different estimates of the substitutability of a homogenous ecosystem service,

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<sup>6</sup>Note that income elasticities are generally not constant but may vary across individuals and also across aggregate measures, as e.g. found in Barbier et al. (2015) and Ready et al. (2002). Broberg (2010) finds that a model with a constant income elasticity does not produce a worse overall fit than those where the income elasticity of WTP is a (non-)linear function.

<sup>7</sup>We thereby follow procedure adopted elsewhere to assume that the single ecosystem service components are part of a homogenous ecosystem service good (Baumgärtner et al. 2015a) and to use an unweighted arithmetic mean (see, additionally, Høekby and Søderqvist (2003)).

Table 1: Estimates of the substitutability parameter  $\theta$ 

Study	$\theta$	$\theta$ error range	Ecosystem service	Location
Barbier et al. (2015)	0.74		Water quality improvement	Baltic Sea
Barton (2002)	0.86		Water quality improvement	Costa Rica
Broberg (2010)	0.63	0.53 – 0.73	Existence of predator species	Sweden
Carlsson/J.-Stenman (2000)	0.68		Air quality improvement	Sweden
Chiabai et al. (2011)	0.69		Aggregate forest services	Global
Hammitt et al. (2001)	0.78		Wetland preservation	Taiwan
Høekby/Sønderqvist (2003)	0.32		Aggregate env. services	Sweden
Jacobsen/Hanley (2009)	0.62	0.48 – 0.76	Aggregate biodiversity	Global
Lindhjem/Tuan (2012)	0.27		Aggregate biodiversity	Asia&Oceania
Liu/Stern (2008)	0.58		Aggregate marine services	Global
Martini/Tiezzi (2014)	-0.16	-0.41 – 0.08	Air quality improvement	Italy
Ready et al. (2002)	0.41		Water quality improvement	Latvia
Schläpfer/Hanley (2003)	0<		Landscape amenities	Switzerland
Sønderqvist/Scharin (2000)	0.73		Water quality improvement	Sweden
Wang/Whittington (2000)	0.73	0.63 – 0.83	Air quality improvement	Bulgaria
Wang et al. (2013)	0.79	0.74 – 0.84	Water quality improvement	China
Whitehead et al. (2000)	0.76	0.58 – 0.94	Recreation improvements	USA
Yu/Abler (2010)	0.80	0.69 – 0.91	Air quality improvement	China
<i>Mean Empirical</i>	0.57	0.28 – 0.86	Aggregate	Aggregate

For Schläpfer/Hanley (2003),  $\theta$  is assumed to be -0.01 in the following. For Chiabai et al. (2011) [Barton (2002)],  $\theta$  is based on the unweighted mean of  $\xi$  for recreational and passive use [for four regions]. For Lindhjem/Tuan (2012) [Barbier et al. (2015)] the estimate is based on the mean of three [four] different model runs. Error ranges for individual  $\theta$ 's derive from the standard errors in estimating the income elasticity of WTP.

we will examine three subgroups more closely: Water and air quality improvement as well as meta-studies. The five studies examining WTP for water quality improvement provide  $\theta$  estimates of 0.41 to 0.86. This high degree of substitutability may indicate that many benefits that come with improved water quality could be substituted by marketed-traded goods. Turning to the four studies on air quality improvement, we observe substantial heterogeneity, with estimates of the substitutability parameter ranging from 0.8 in China (Yu and Abler 2010) to -0.16 in Italy (Martini and Tiezzi 2014). In particular, there is a substantial difference between estimates derived from revealed preference ( $\theta = -0.16$ ) and contingent valuation studies ( $0.68 \leq \theta \leq 0.8$ ). Finally, we consider the five meta-studies that have estimated WTP for global forest ecosystem services (Chiabai

et al. 2011), global biodiversity conservation (Jacobsen and Hanley 2009), biodiversity conservation in Asia and Oceania (Lindhjem and Tuan 2012), global marine ecosystem services (Liu and Stern 2008) as well as an aggregate environmental service in Sweden (Høekby and Søderqvist 2003). These studies estimate income elasticities  $\xi$  in the range of 0.31 to 0.73, implying values of the substitutability parameter  $\theta$  of 0.27 to 0.69. The estimate that comes closest to approximating substitutability of a composite ecosystem service at the global scale is the meta-study by Jacobsen and Hanley (2009), who gather 145 different WTP estimates from 46 contingent valuation studies across six continents (Baumgärtner et al. 2015a). They find, averaging over the very different biodiversity-related ecosystem services (i.e. assuming that they are part of a homogeneous good), that the income elasticity of the WTP for ecosystem services is 0.38 [0.24 to 0.52], implying a substitutability parameter of 0.62 [0.48 to 0.76]. With an abundant provisioning of ecosystem services or without a subsistence requirement, this would imply a mean elasticity of substitution of 2.63 [1.92 to 4.17].

We now turn to estimating the elasticity of substitution in a world where ecosystem services are limited. We simulate the time development of the elasticity of substitution for an empirically founded, yet hypothetical case study. For this, we draw on estimates from the existing literature on limited substitutability and dual discounting (Hoel and Sterner 2007; Baumgärtner et al. 2015a). In line with the 300-year simulation exercise of Hoel and Sterner (2007), we normalise initial values of the consumption of the ecosystem service  $E_0$  and of the manufactured good  $C_0$  to unity, and adopt a value of 0.1 for the preference parameter concerning the importance of ecosystem services in consumption  $\alpha$ . For the growth rates of ecosystem services and manufactured goods, we use estimates for both aggregate goods for the past 60 years from Baumgärtner et al. (2015a), with  $g_C(t) = 1.88$  percent and  $g_E(t) = -0.52$  percent. We therefore consider the case where these two growth rates remain constant.<sup>8</sup> Additionally to these baseline specifications, we add a subsistence requirement  $\bar{E}$  of 0.15, i.e. we assume that 15 percent of the initial

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<sup>8</sup>While these growth rates do not stem from an optimising behaviour of the representative agent and thus only facilitate valuation along a non-optimal trajectory, this approach represents standard practice in the literature (cf. Hoel and Sterner 2007; Traeger 2011).

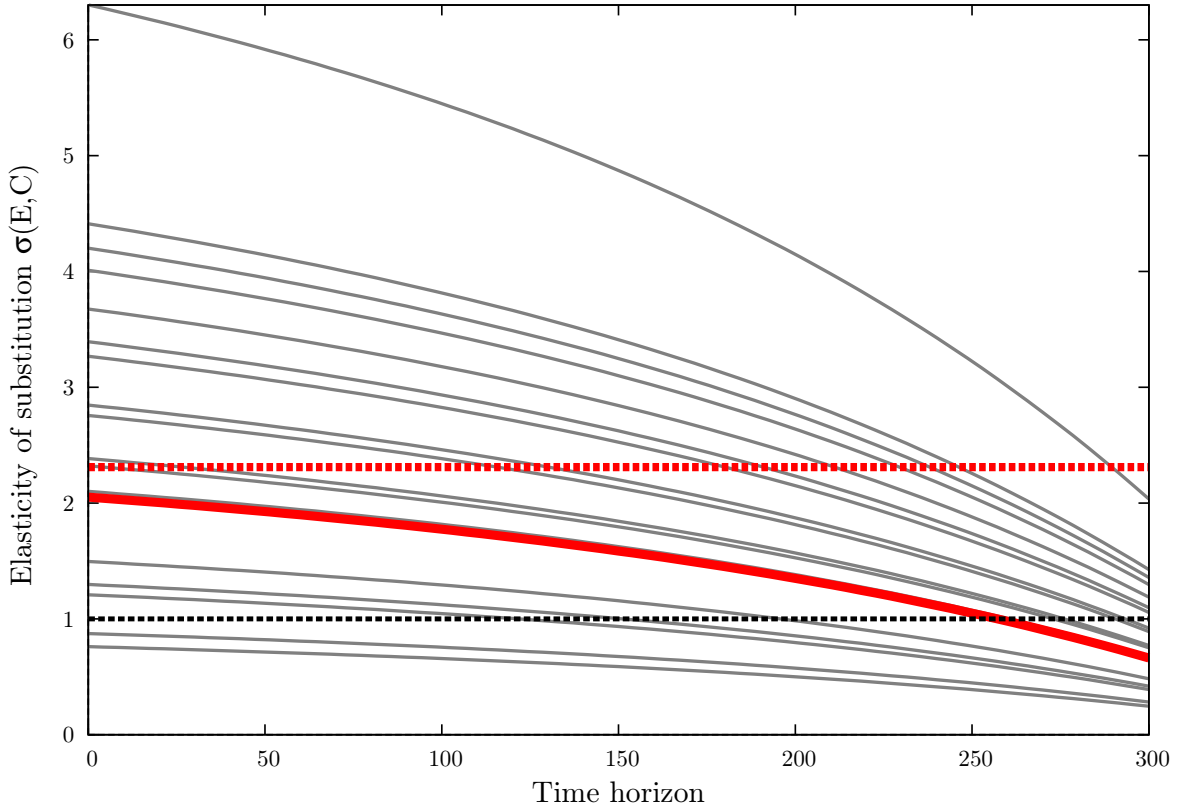


Figure 1: Elasticity of substitution  $\sigma(E, C)$  as a function of time for all empirical estimates of the substitutability parameter  $\theta$  depicted as thin grey lines (with the sample mean estimate of  $\theta = 0.57$  depicted as a thicker red line); with  $\alpha = 0.1$ ,  $E_0 = C_0 = 1$ ,  $\bar{E} = 0.15$ ,  $g_C(t) = 1.88$  percent, and  $g_E(t) = -0.52$  percent. The horizontal, dotted black (red) line marks the dividing line between substitutes and complements (the constant elasticity of substitution corresponding to the sample mean  $\theta = 0.57$ ).

consumption of ecosystem services are required to meet subsistence needs.

Figure 1 depicts the elasticity of substitution  $\sigma(E, C)$  as a function of time for the whole range of empirical estimates of the substitutability parameter  $\theta$  (cf. Table 1). The dotted, horizontal red line illustrates the standard CES of 2.31 for the sample mean estimate of the substitutability parameter  $\theta$  of 0.57. The thick red curve shows the time development of the elasticity of substitution in the presence of the subsistence requirement for the sample mean estimate of the substitutability parameter. The elasticity of substitution has a value of 2 initially, yet declines as ecosystem services become scarcer and their consumption level approaches the amount required to satisfy subsis-

tence needs. After 255 years, the elasticity of substitution falls below the threshold of unity, thus shifting the substitutability relation to complementary (depicted as the horizontal black line in Figure 1).<sup>9</sup> Figure 1 thus highlights the importance of considering absolute limits to substitution for estimating the elasticity of substitution. While the estimation error for incorrectly considering a CES specification is ‘only’ 0.31 initially, and thus an overestimate of the elasticity of substitution of 16 percent, this overestimate increases to 32 (73) [251] percent in 100 (200) [300] years.

## 2.4 Discussion of empirical substitutability estimates

Our survey and analysis of available estimates on substitutability between ecosystem services and manufactured goods has shown that the only approach that currently yields empirical estimates is an indirect route via the income elasticity of WTP. In a world with an abundant provisioning of ecosystem services or without a subsistence requirement (i.e. where the CES approximation is valid), almost all of these indirect estimates suggest that the elasticity of substitution is substantially larger than one.<sup>10</sup> This stands in stark contrast to parameter values used in applied modelling. While (almost) all empirical studies suggest a substitutability parameter of greater zero, (almost) all applied modelling studies have chosen values of the substitutability parameter smaller than zero for their analysis and thus assumed a complementary relationship between ecosystem services and manufactured goods (Gollier 2010; Hoel and Sterner 2007; Kopp et al. 2012; Sterner and Persson 2008). An explanation for this finding could be that none of these modelling studies could base their substitution elasticity on empirical evidence. While these modelling studies would have perhaps more carefully considered the empirical estimates of the substitutability parameter presented in Table 1 to compute their (constant

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<sup>9</sup>A sensitivity analysis with respect to the subsistence requirement reveals that elasticity of substitution would fall below the threshold of unity after 201 (333) years for a value of  $\bar{E}$  of 0.2 (0.1).

<sup>10</sup>The clear result of income elasticities smaller than unity obtained throughout the contingent valuation literature has been challenged by Schlöpfer (2006, 2008, 2009). Schlöpfer argues that the small income elasticities may be an artefact of the current design of contingent valuation studies, which may lower the income effect. He compares contingent valuation with voting-based studies (Schlöpfer and Hanley 2003, 2006) and finds support for an income elasticity of WTP equal to or greater than unity.

approximation of the) elasticity of substitution, we hypothesise that they might not and should not have simply settled with these estimates due to limiting biases of the assumption of a normal, homogenous ecosystem service. Indeed, both a consideration of heterogenous ecosystem services as well as their potential subsistence good nature become relevant for putting the empirical estimates gathered into perspective.

First, our sample on the substitutability of ecosystem services is non-representative of the true global distribution of ecosystem services. Valuation studies have been carried out predominantly in developed countries and do not capture income effects on the whole range of ecosystem services. In particular, valuation studies of climate- and disease regulation as well as aesthetic and spiritual ecosystem services, which might have a low degree of substitutability by manufactured goods, are often not accounted for.<sup>11</sup>

Second, even if information on the income elasticity of WTP were available for the whole distribution of heterogenous ecosystem services, the non-trivial question of how to aggregate these into a single ecosystem service good remains. In line with the previous literature, we assume that ecosystem services are a homogenous good and use the unweighted arithmetic mean to aggregate substitutability of different ecosystem services with respect to manufactured goods. This implies that ecosystem services considered are assumed to have the same weight and the elasticity of substitution with respect to the manufactured good, or that there is perfect substitutability within the aggregate ecosystem service. For example, we have given equal weight to the substitutability parameter derived from the single valuation study on recreation improvements (Whitehead et al. 2000) as to the meta-studies on biodiversity conservation by Jacobsen and Hanley (2009) or Lidhjem and Tuan (2012). Firstly, one might argue that meta-studies might be weighted in relation to the single studies of our sample according to the number of

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<sup>11</sup>While the omission of more difficult to measure WTPs for ecosystem services might lead to an overestimate of the overall degree of substitutability, the effect of an over-proportionate availability of studies from developed countries might bias the estimate in the opposite direction. The available evidence so far suggests that the income elasticity of WTP might be higher in high-income populations (Barbier et al. 2015; Ready et al. 2002), suggesting that the current sample mean would underestimate the average degree of substitutability. We cannot [can] confirm this finding when comparing developed and developing countries for estimates derived from all [contingent] valuation studies in Table 1.



primary studies they contain. Secondly, one might argue, for example, that biodiversity is more important for the aggregate state of ecosystem services than recreational benefits and that it should be given a greater weight. Furthermore, ecosystem services will in general not have the same elasticity of substitution with respect to manufactured goods, or be perfectly substitutable among each other.<sup>12</sup> Regarding the implications of heterogeneity of ecosystem services for substitutability of an aggregate ecosystem service, it is worth quoting Sterner and Persson’s (2008: 71) consideration of choosing a low degree of substitutability for their analysis at length:

*“[If] there is a range of [ES] with different elasticities of substitution, then the relevant aggregate number is very likely not going to be the average of those elasticities. It is the goods [...] with low elasticities that will dominate the calculation, since these will be the ones with increasing shares in utility. This goes for clean water, pollination services, and many other subtle aspects of the ecosystem that we take for granted as long as they are plentiful.”*

Finally, we have considered a suggestion by Heal (2009) and further developed in Baumgärtner et al. (2015b) that the elasticity of substitution between ecosystem services and manufactured goods is non-constant due to the existence of a subsistence requirement in the consumption of ecosystem services. Specifically, the elasticity of substitution varies with the availability of ecosystem services in relation to the services necessary to meet basic subsistence needs. Humans will not be willing to substitute ecosystem services for manufactured goods if these subsistence requirements are not sufficiently met and will be willing to substitute when ecosystem services are “plentiful”. This implies that even if the aggregate ecosystem service is initially considered as a substitute, it will – given a declining availability of ecosystem services – eventually become a complement (see Figure 1). Even though the scenario analysis shows that the elasticity of substitution does not reach values of limited substitutability adopted in the applied modelling literature (Sterner and Persson 2008) even for a time horizon of 300 years, it suggests

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<sup>12</sup>Similar to measuring biodiversity (Bertram and Quaas 2016) one might consider an overall index of ecosystem service abundance with imperfect substitutability among the single ecosystem services.

that the general notion of the concern addressed by the quote provided above may to a considerable degree be resolved by considering absolute limits to substitution of ecosystem services in the form of a subsistence requirement. More generally, it reconciles the idea that mankind ultimately cannot, or is not willing to, live without the provisioning of fundamental ecosystem services and the available empirical evidence suggesting that the current willingness to substitute ecosystem services for manufactured goods still seems to be high. This motivates exploring the implications of including an absolute limit to substitution of ecosystem services in the form of a subsistence requirement for intertemporal decision-making and social discounting.

### 3 Dual discounting under a subsistence requirement

#### 3.1 Model and definitions

This section develops a dual discounting model that considers a subsistence requirement in the consumption of ecosystem services. Our modelling set-up extends the approaches of Hoel and Sterner (2007) and Traeger (2011): A social planner has perfect knowledge about the future and maximizes an intertemporal, constant intertemporal elasticity of substitution (CIES) social welfare function based on the CES instantaneous utility function  $U_h$  (from Equation 2). Welfare is given by

$$W = \int_0^T \underbrace{\frac{1}{1-\eta} [\alpha(E_t - \bar{E})^\theta + (1-\alpha)C_t^\theta]^{\frac{1-\eta}{\theta}}}_{u(E,C)} e^{-\delta t} dt, \quad (5)$$

$\underbrace{\hspace{10em}}_{U(E,C,t)}$

where  $\delta$  is the utility discount rate and  $\eta$  is the inverse of the CIES with respect to the within-period aggregate consumption bundle  $\tilde{C} = [\alpha(E_t - \bar{E})^\theta + (1-\alpha)C_t^\theta]^{\frac{1}{\theta}}$ . We assume a finite time horizon  $T$ , due to the possibility of  $E_t < \bar{E}$ , implying that intertemporal welfare would be undefined.

Social *discount factors* and *-rates* are defined as follows (cf. Traeger 2011: 216): The good-specific *discount factors* for good  $x_i$ , with  $x_i, x_j \in \{E_t, C_t\}$  and  $i \neq j$ , relating the additional value of good  $x_i$  between the points in time  $t$  and  $t_0$  for a given, but not

necessarily optimal, consumption path of  $E$  and  $C$  are

$$P_i(t, t_0) = \frac{\frac{\partial U(E_t, C_t, t)}{\partial x_i}}{\frac{\partial U(E_t, C_t, t_0)}{\partial x_i}}. \quad (6)$$

The corresponding good-specific *discount rates* for good  $i$  are given by:

$$\rho_i(t) = -\frac{\frac{\partial^2 U}{\partial t \partial x_i}(t) + \frac{\partial^2 U}{\partial^2 x_i}(t)\dot{x}_i(t) + \frac{\partial^2 U}{\partial x_j \partial x_i}(t)\dot{x}_j(t)}{\frac{\partial U}{\partial x_i}(t)}. \quad (7)$$

Using  $U(E, C, t)$  from Equation (5), this can be expressed in simpler terms as:

$$\rho_E(t) = \delta + \psi_{EE}g_E(t) + \psi_{EC}g_C(t) \quad (8)$$

and

$$\rho_C(t) = \delta + \psi_{CC}g_C(t) + \psi_{CE}g_E(t), \quad (9)$$

where  $\psi_{EE}$  (resp.  $\psi_{CC}$ ) is the elasticity of marginal utility of consumption of the aggregate ecosystem service (resp. manufactured good) with respect to the ecosystem service (manufactured good), and  $\psi_{EC}$  ( $\psi_{CE}$ ) is the elasticity of marginal utility of consumption of the ecosystem service (manufactured good) with respect to the manufactured good (ecosystem service); The growth rates  $g_i$  are defined as  $g_i(t) = \frac{\dot{x}_i(t)}{x_i(t)}$ .

## 3.2 Results

We now derive and analyse the good-specific discount rates and in particular their difference – also termed ‘relative price effect’ (Hoel and Sterner 2007) – in the presence of a subsistence requirement in the consumption of ecosystem services (Appendix A.2).

The discount rate for ecosystem services is given by:

$$\rho_E(t) = \delta + \frac{E_t}{E_t - \bar{E}} \frac{\alpha\eta(E_t - \bar{E})^\theta + (1 - \alpha)(1 - \theta)C_t^\theta}{\alpha(E_t - \bar{E})^\theta + (1 - \alpha)C_t^\theta} g_E(t) + \frac{(1 - \alpha)C_t^\theta(\eta - (1 - \theta))}{\alpha(E_t - \bar{E})^\theta + (1 - \alpha)C_t^\theta} g_C(t) \quad (10)$$

while the manufactured good discount rates is given by

$$\rho_C(t) = \delta + \frac{\alpha(1 - \theta)(E_t - \bar{E})^\theta + (1 - \alpha)\eta C_t^\theta}{\alpha(E_t - \bar{E})^\theta + (1 - \alpha)C_t^\theta} g_C(t) + \frac{\alpha E_t (E_t - \bar{E})^{\theta-1} (\eta - (1 - \theta))}{\alpha(E_t - \bar{E})^\theta + (1 - \alpha)C_t^\theta} g_E(t). \quad (11)$$

Deriving the difference in good-specific discount rates yields the ‘relative price effect’:

$$\Delta\rho(t) = \rho_C(t) - \rho_E(t) = (1 - \theta) \left[ g_C(t) - \frac{E_t}{E_t - \bar{E}} g_E(t) \right]. \quad (12)$$

In the following, we analyse and illustrate the properties of the relative price effect only.

For the special case of  $\bar{E} = 0$ , the relative price effect  $\Delta\rho$  collapses to the formula of the standard CES case  $\Delta\rho^{CES} = (1 - \theta) \times [g_C(t) - g_E(t)]$ . If a positive amount of ecosystem services is necessary for subsistence ( $\bar{E} > 0$ ), the relative price effect not only depends on the substitutability parameter and the difference in the two good-specific growth rates, but also on the consumption of ecosystem services over and above the subsistence requirement. This amplifies the effect of the growth rate of ecosystem services on the relative price effect. For the relevant case of imperfect substitutes ( $\theta < 1$ ) and a subsistence requirement ( $\bar{E} > 0$ ), we derive the following results from Equation (12):

**Proposition 1**

For  $E_t > \bar{E} > 0$ , bounded growth rates, and  $\theta < 1$ , the ‘relative price effect’  $\Delta\rho$  has the following properties:

$$\frac{\partial\Delta\rho(t)}{\partial\bar{E}} \begin{matrix} \geq \\ \leq \end{matrix} 0 \quad \text{for} \quad g_E(t) \begin{matrix} \leq \\ \geq \end{matrix} 0 \quad (13)$$

$$\frac{\partial\Delta\rho(t)}{\partial\theta} \begin{matrix} \geq \\ \leq \end{matrix} 0 \quad \text{for} \quad g_E(t) \frac{E_t}{E_t - \bar{E}} \begin{matrix} \geq \\ \leq \end{matrix} g_C(t) \quad (14)$$

$$\dot{\Delta\rho}(t) > 0 \quad \text{for} \quad \dot{g}_C(t) = \dot{g}_E(t) = 0 \quad \text{and} \quad g_E(t) \neq 0 \quad (15)$$

$$\Delta\rho(t) \rightarrow \infty \quad \text{for} \quad g_E(t) < 0 \quad (16)$$

$$\Delta\rho(t) \rightarrow \Delta\rho^{CES} \quad \text{for} \quad \begin{cases} g_E(t) > 0 \\ \bar{E} \rightarrow 0 \end{cases} \quad (17)$$

$$\Delta\rho \begin{matrix} \geq \\ \leq \end{matrix} \Delta\rho^{CES} \quad \text{for} \quad g_E(t) \begin{matrix} \leq \\ \geq \end{matrix} 0. \quad (18)$$

*Proof.* See Appendix A.3. □

Equation (13) states that the relative price effect increases (decreases) with the stringency of the subsistence requirement if the growth rate of ecosystem services is

smaller (larger) than zero for all times. For the empirically relevant case of a decline in ecosystem services, the relative price effect would therefore increase if a larger share of ecosystem services would be necessary to meet subsistence needs.

Equation (14) shows that the effect of an increase of the substitutability parameter depends on the relative magnitude of the growth rate for manufactured goods and the growth rate for ecosystem services, weighted by their consumption relative to the amount consumed over and above the subsistence requirement. As  $\frac{E_t}{E_t - \bar{E}} > 1$  for  $E_t > \bar{E} > 0$ , the presence of the subsistence requirement reduces the parameter value space for which an increase of the substitutability parameter will lead to a lower relative price effect.

Equation (15) shows that if the growth rates are constant (see, e.g., Baumgärtner et al. 2015a; Hoel and Sterner 2007), and the growth rate of ecosystem services is different from zero, then the relative price effect will increase over time.

If ecosystem services are limited, the relative price effect is non-constant and depends on the consumption of ecosystem services relative to the subsistence requirement. Specifically, if ecosystem services are in a continuous decline, the relative price effect increases and approaches infinity in the limit (Equation 16). Equation (17) shows, in contrast, that if the provisioning of ecosystem services grows over time and becomes ‘plentiful’, the relative price effect becomes equivalent to the standard CES case, as it does for a vanishing subsistence requirement.

Equation (18) states that the relative price effect is larger (smaller) than its CES version if the growth rate of ecosystem services is negative (positive). Naturally, for the case of a constant consumption of ecosystem services, the two versions coincide and only depend on the substitutability parameter and the growth rate of manufactured goods.

### 3.3 Empirical Analysis

We now illustrate the time development of the relative price effect of ecosystem services under different assumptions on substitutability and limits to substitution. We consider the same scenario that we adopted in the analysis of the elasticity of substitution (Section 2.3). The baseline parameters are taken from the 300-year simulation exercise of Hoel and Sterner (2007), now including the social rate of pure time preference  $\delta$  of 1

percent and a measure of intertemporal inequality aversion in consumption  $\eta$  of 1.5.

First, we estimate the relative price effects of ecosystem services  $\Delta\rho$  corresponding to each of the empirical estimates of the substitutability parameter  $\theta$ . As the relative price effect is non-constant due to the subsistence requirement and a decreasing availability of ecosystem services, we compute it for three different points in time:  $t = 0, 150$  and 300 years (cf. Table 2, which also reports the relative price effect for prominent assumptions in the dual discounting literature). Where possible, we also compute the initial error range for the relative price effect.<sup>13</sup> For the sample mean of empirical estimates of the substitutability parameter we observe that the initial relative price effect is 1.08 percentage-points and rises up to 1.15 (1.61) percentage-points in 150 (300) years. For the smallest empirical estimate of the substitutability parameter (derived from Barton 2002), the relative price effect is only 0.35 percentage-points initially and rises to 0.52 percentage-points in 300 years. However, for the value derived from the revealed preference study on air quality improvement in Italy (Martini and Tiezzi 2014), the relative price effect is 2.89 percentage-points initially and rises up to 3.08 (4.31) percentage-points in 150 (300) years.

Second, we contrast the sample mean of empirical evidence on the substitutability parameter with prominent assumptions from the literature (Gollier 2010; Sterner and Persson 2010). Figure 2 depicts the relative price effect  $\Delta\rho$  for these three cases of the substitutability parameter  $\theta$  and contrasts this with the constant relative price effect  $\Delta\rho^{CES}$  prevalent in the previous literature. The three cases represent initial high ( $\theta = 0.57$ ), Cobb-Douglas ( $\theta = 0$ ) and limited substitutability ( $\theta = -1$ ). The relative price effect in the presence of the subsistence requirement is always higher than under the standard CES non-subsistence case for our parameter specifications, i.e.  $g_E(t) < 0 < g_C(t)$ , and increases without bound as we approach the subsistence requirement in terms of ecosystem services.<sup>14</sup> Over time, the relative price effect with subsistence consumption rises considerably above its CES counterpart. For the limited

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<sup>13</sup>The error range for the sample mean is computed as the standard deviation of the single initial relative price effect estimates.

<sup>14</sup>With the given specifications, we approach the subsistence requirement after 363 years.

Table 2: Estimates of the relative price effect of ecosystem services  $\Delta\rho$  (in %-points)

Study	$\Delta\rho_{t=0}$	$\Delta\rho_{t=0}$ error range	$\Delta\rho_{t=150}$	$\Delta\rho_{t=300}$
Barbier et al. (2015)	0.65		0.69	0.97
Barton (2002)	0.35		0.37	0.52
Broberg (2010)	0.92	0.67 – 1.17	0.98	1.37
Carlsson/J.-Stenman (2000)	0.80		0.85	1.19
Chiabai et al. (2011)	0.77		0.82	1.15
Hammitt et al. (2001)	0.55		0.58	0.82
Høekby/Søederqvist (2003)	1.69		1.80	2.53
Jacobsen/Hanley (2009)	0.95	0.60 – 1.30	1.01	1.41
Lindjhem/Tuan (2012)	1.82		1.94	2.71
Liu and Stern (2008)	1.05		1.11	1.56
Martini/Tiezzi (2014)	2.89	2.29 – 3.51	3.08	4.31
Ready et al. (2002)	1.47		1.57	2.19
Schläpfer/Hanley (2003)	2.52	$\geq 2.52$	2.68	3.75
Søederqvist/Scharin (2000)	0.67		0.72	1.00
Wang/Whittington (2000)	0.67	0.42 – 0.92	0.72	1.00
Wang et al. (2013)	0.52	0.40 – 0.65	0.56	0.89
Whitehead et al. (2000)	0.60	0.15 – 1.05	0.64	0.89
Yu/Abler (2010)	0.50	0.22 – 0.77	0.53	0.74
<i>Mean Empirical</i>	1.08	0.36 – 1.80	1.15	1.61
Gollier (2010)	2.49	1.66 – 4.98	2.65	3.72
Hoel/Serner (2007)	4.98	2.49 – 4.98	5.31	7.43
Kopp et al. (2012)	3.32	2.49 – 4.98	3.54	4.95
Serner/Persson (2008)	4.98	2.49 – 4.98	5.31	7.43

substitutability assumption of Serner and Persson (2008), the CES specification would entail a relative price effect of 4.80 percentage-points. Yet, with a subsistence requirement, this relative price effect would be 4.98 percentage-points initially and rise to 5.31 (7.43) percentage-points in 150 (300) years. While the absolute percentage-point difference depends on the substitutability parameter, the relative difference between the relative price effect with and without the subsistence requirement is independent of it. Given our specifications, the standard CES approach would underestimate the relative price effect by 4 (11) [55] percent after 0 (150) [300] years. Figure 2 thus shows that the relative price effect and its difference to the standard CES case can be substantial.

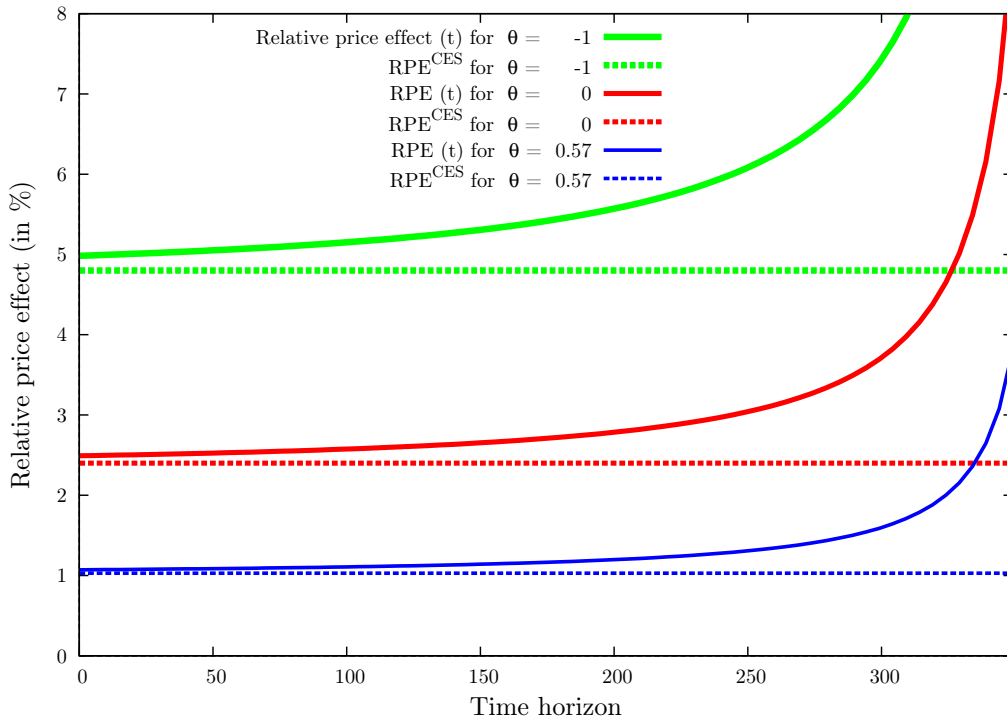


Figure 2: Relative price effect (RPE) of ecosystem services in percentage-points for different values of the substitutability parameter. We compare the RPE for the sample mean of the empirical evidence on substitutability [ $\theta = 0.57$ ; blue] with the assumptions of Gollier (2010) [ $\theta = 0$ ; red] and Sterner and Persson (2008) [ $\theta = -1$ ; green]. The dotted horizontal lines depict the respective standard CES relative price effects,  $\Delta\rho^{CES}$ , while the relative price effects,  $\Delta\rho(t)$ , in the presence of a subsistence requirement (solid lines) increase over time. Common parameter values are  $\alpha = 0.1$ ,  $E_0 = C_0 = 1$ ,  $\bar{E} = 0.15$ ,  $g_C(t) = 1.88$  percent, and  $g_E(t) = -0.52$  percent.

Finally, we consider the effect of the absolute limit to substitution in form of the subsistence requirement in terms of ecosystem services. Proposition 1 shows that the relative price effect increases with a more stringent subsistence requirement if the growth rate of ecosystem services is negative. As the assumption to set  $\bar{E}$  equal to 0.15 could not be based on any available empirical evidence, we will scrutinise this assumption in the following. Figure 3 illustrates the relative price effect of ecosystem services as a function of the subsistence requirement,  $\bar{E}$ , for the sample mean of empirical estimates on the substitutability parameter for different time horizons ( $t = 0$  grey dotted line,  $t = 150$  red solid line,  $t = 300$  black dashed-dotted line). This reveals that while the



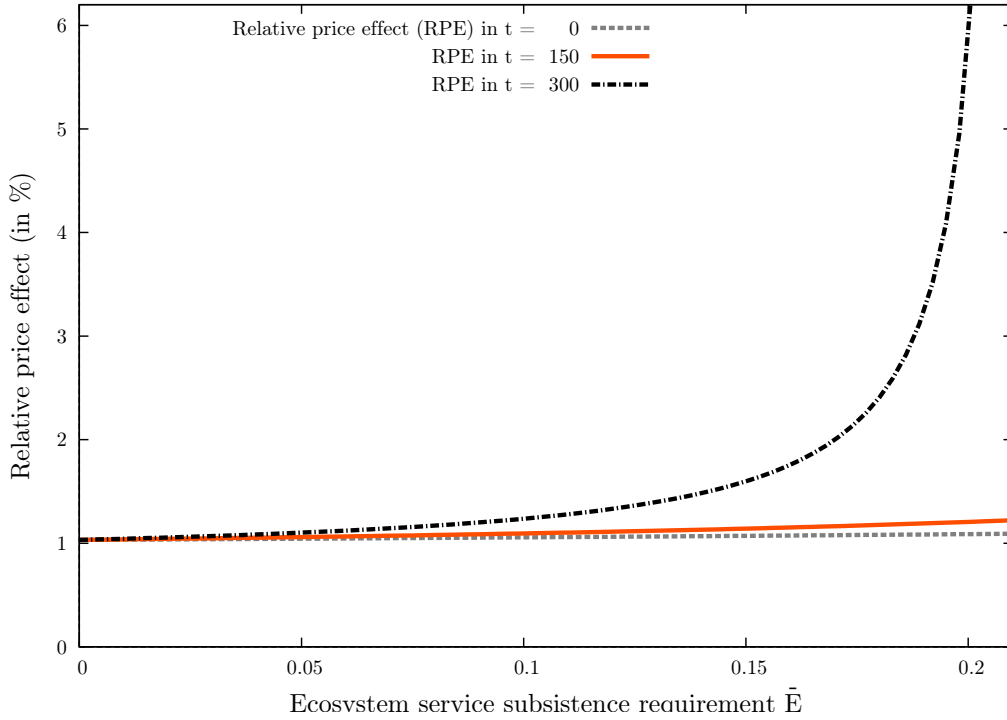


Figure 3: Relative price effect  $\Delta\rho$  of ecosystem services as a function of the subsistence requirement,  $\bar{E}$ , for the sample mean of empirical estimates on the substitutability parameter for different time horizons ( $t = 0$  grey dotted line,  $t = 150$  red solid line,  $t = 300$  black dashed-dotted line).

relative price effect is only 1.06 (1.10) [1.24] percentage-points in  $t=0$  ( $t=150$ ) [ $t=300$ ] years for a subsistence requirement level of  $\bar{E} = 0.1$ , it rises to up to 1.09 (1.21) [5.88] percentage-points for  $\bar{E} = 0.2$ . Already for a subsistence requirement of  $\bar{E} = 0.22$ , subsistence needs could not be satisfied for a 300 year time horizon due to the decline in ecosystem services. Assumptions on the amount of ecosystem services required for subsistence therefore crucially impact the relative price effect. Examining the effect of the subsistence requirement on the relative price of ecosystem services also reveals a noteworthy correspondence between the empirical estimate of the substitutability parameter  $\theta$  in a limits-to-substitution framework and a limited substitutability assumption in the standard CES setting: Assuming that approximately 43.3 [19.8] percent of the initial consumption of ecosystem services is necessary to meet subsistence needs would be equivalent in terms of the relative price effect in 150 [300] years to assuming the limited substitutability value of  $\theta = -1$  in the CES specification (cf. Sterner and Persson 2008).

## 4 Conclusions

This paper has examined implications of an absolute limit to substitution of ecosystem services on estimating substitutability between ecosystem services and manufactured consumption goods as well as on the theory and empirics of social discounting of ecosystem services. Specifically, we have considered a case where substitution possibilities are restricted by a subsistence requirement in the consumption of ecosystem services, which may consist of water, food and life-enabling ecosystem conditions.

Based on a framework that allows for absolute limits to substitution, we provide empirical estimates of the substitutability of ecosystem services. In the standard constant elasticity of substitution (CES) case discussed in the literature (Baumgärtner et al. 2015a; Yu and Abler 2010), there is a direct relationship between the income elasticity of WTP for ecosystem services, a fundamental substitutability parameter and the CES. If there is a subsistence requirement, this direct relationship between the elasticity of substitution and the other two breaks down. We provide a comprehensive account of empirical estimates on the substitutability parameter and, by means of a long-term simulation, analyse what these may imply for estimating the elasticity of substitution.

We find that empirical evidence on the substitutability parameter suggests that most ecosystem services are considered highly substitutable by manufactured goods. Assuming that individual services are part of a homogenous ecosystem service good, we aggregate over all studies and find a mean substitutability parameter of 0.57. The scenario analysis reveals that the elasticity of substitution for the sample mean has a value of 2 initially and decreases to 1.56 (0.66) over 150 (300) years. Compared to this variable elasticity of substitution, the CES approach would result in an estimation error of 32 (73) percent after 100 (200) years. Substitution possibilities are thus considerably lower as previously suggested (Yu and Abler 2010; Baumgärtner et al. 2015a).

Subsequently, we explore implications of considering an absolute limit to substitution for social discounting, by extending dual discounting models to allow for a subsistence requirement in the consumption of ecosystem services. We find that the relative price effect of ecosystem services not only depends on the substitutability parameter and the

difference in the two good-specific growth rates but also on the consumption of ecosystem services over and above the subsistence requirement. This extension produces results similar to the standard non-subsistence model only if ecosystem services are plentiful. If, however, the provision of ecosystem services is limited and in decline – as suggested by empirical evidence – the model produces markedly different results compared to the non-subsistence case. In particular, we find that in such a case the ‘relative price effect’ is not constant but grows without bound as the consumption of ecosystem services declines towards the amount required for subsistence.

We illustrate the results using the same scenario as for the estimation of substitutability. We find that the initial discount rate for ecosystem services should be on average 1.1 percentage-points lower compared to the rate for manufactured goods. This difference increases by more than half a percentage-point on average over a 300-year time horizon due to the subsistence requirement. A sensitivity analysis further reveals that the longer-term relative price effect is very sensitive to assumptions on the level of ecosystem services required for subsistence.

Our findings are relevant in several respects: First, we have provided a conceptual framework and empirical evidence to tackle one of the main gaps in the literature on sustainability: the estimation of substitutability between ecosystem services and manufactured goods. This provides a number of lessons. Firstly, estimates on substitutability from sources other than contingent valuation studies are scarce and further research should be channelled into estimating substitution possibilities using other methodological approaches, such as choice experiments and revealed preference studies. Secondly, the possibility to adequately estimating the elasticity of substitution is complicated by the existence of a subsistence requirement. This becomes directly relevant as it sheds light on the substantial magnitude by which previous CES approaches may have overestimated the elasticity of substitution. Furthermore, our analysis shows that the elasticity of substitution does not reach values of limited substitutability adopted in the applied modelling literature (Sterner and Persson 2008) even for a time horizon of 300 years. Furthermore, our analysis suggests that attempts at adequately capturing substitutability between ecosystem services and manufactured goods should not only focus

on methodological approaches other than contingent valuation studies, but in particular that one would need to perform long-term studies that scrutinise how estimates of the elasticity of substitution develop over time and which factors influence it. Lastly, the substantial heterogeneity of indirect estimates on substitutability of ecosystem services that we obtained – across as well as within different ecosystem services categories – questions the often adopted assumptions of a homogenous ecosystem service good. Future research should therefore explore implications of this heterogeneity.

Second, our results provide stronger support for the case of including ecological discount rates in project evaluation (Baumgärtner et al. 2015a). We find that ecosystem services should be discounted at a rate that is 1.1 percentage points lower than the rate for manufactured consumption goods initially and 1.6 percentage points for a 300-year time horizon. Indeed, none of our estimates suggests that ecosystem services should be discounted at the same rate as manufactured goods.

Finally, this paper relates to the intensely debated notions of ‘planetary boundaries’ (Rockström et al. 2009) and ‘catastrophic’ climate change (Millner 2013; Weitzman 2009). By emphasising the importance of basic subsistence needs, our approach re-connects the study of intertemporal allocation to the original notion of sustainability (WCED 1987). Furthermore, in our setting, ‘catastrophic’ climate change might be conceptualised as the loss of ecosystem services required for subsistence, such as an adequate food supply, fresh water, and life-enabling ecosystem conditions. This does not imply that a focus on fat-tailed probability distributions of climate damages is superfluous – indeed these ‘dismal’ results would be even more ‘dismal’ if catastrophe occurs already at a strictly positive consumption level. But it follows that more effort should be channelled into discussing the substance of the notion of ‘catastrophe’ and that climate policies should aim first and foremost at securing the provisioning of subsistence services. Our analysis has shown that the relative price effect of ecosystem services is very sensitive to the amount of ecosystem services required for subsistence. More research, as well as societal and political discussions, should therefore be channelled into determining the magnitude and composition of such a subsistence requirement.

# Appendix

## A.1 Relationship between the income elasticity, the CES substitutability parameter and the elasticity of substitution

This appendix clarifies the relationship between the income elasticity of WTP, the CES substitutability parameter and the elasticity of substitution.

The agent's income is exogenously given and denoted by  $Y$ . The consumption good is traded on a market at given price  $p > 0$ , while the consumption of the ecosystem service is fixed at an exogenously given level  $E > 0$ . The agent maximizes its utility subject to the budget constraint and fixed level of the ecosystem service:

$$\max_{E,C} U_h(E, C) \quad \text{s.t.} \quad pC = Y \text{ and } E \text{ fixed} . \quad (\text{A.1})$$

Following Ebert (2003) the income-equivalent total WTP for the ecosystem service at level  $E$  is defined as the WTP  $w$  per unit times the total number  $E$  of units:

$$\text{WTP} = w E . \quad (\text{A.2})$$

The marginal willingness to pay  $w$  is implicitly defined as the virtual price that yields the ecosystem service level  $E$  as the ordinary (unconditional) Marshallian demand in the hypothetical choice problem where the ecosystem service is considered a private market good. It can be derived from the agent's indirect utility function  $V(p, E, Y)$  by an extension of Roy's identity (Ebert 2003: 440).

$$w = \frac{\partial V(p, E, Y) / \partial E}{\partial V(p, E, Y) / \partial Y} . \quad (\text{A.3})$$

With utility function (Equation 2) the indirect utility function is

$$V(p, E, C) = \left[ \alpha (E - \bar{E})^\theta + (1 - \alpha) \left( \frac{Y}{p} \right)^\theta \right]^{1/\theta} \quad (\text{A.4})$$

and the partial derivatives:

$$\frac{\partial V(p, E, Y)}{\partial E} = \alpha (E - \bar{E})^{\theta-1} \left[ \alpha (E - \bar{E})^\theta + (1 - \alpha) \left( \frac{Y}{p} \right)^\theta \right]^{1/\theta-1} \quad (\text{A.5})$$

$$\frac{\partial V(p, E, Y)}{\partial Y} = (1 - \alpha) p^{-\theta} Y^{\theta-1} \left[ \alpha (E - \bar{E})^\theta + (1 - \alpha) \left( \frac{Y}{p} \right)^\theta \right]^{1/\theta-1} \quad (\text{A.6})$$

Employing (A.3), the marginal WTP is given by

$$w = \frac{\alpha}{1 - \alpha} p^\theta (E - \bar{E})^{\theta-1} Y^{1-\theta} \quad (\text{A.7})$$

Plugging this into Equation (A.2) yields

$$\text{WTP}(Y) = wE = \frac{\alpha}{1 - \alpha} p^\theta E (E - \bar{E})^{\theta-1} Y^{1-\theta}. \quad (\text{A.8})$$

With utility function 2, the agents total WTP for the ecosystem service at level  $E$  then depends on income  $Y$  and the other model parameters as follows:

$$\text{WTP}(Y) = \nu Y^{1-\theta} \quad \text{with} \quad \nu = \frac{\alpha}{1 - \alpha} p^\theta E (E - \bar{E})^{\theta-1}. \quad (\text{A.9})$$

The (constant) income elasticity of WTP, denoted  $\xi$ , is thus given by  $1 - \theta$ . We have therefore established that, as in the CES case, if  $\xi \leq 1$  then  $\theta \geq 0$ . Yet, as the inverse of the elasticity of substitution not only depends on  $\theta$  but on all other parameters and variables of the model as follows (cf. Baumgärtner et al. 2015b: 6)

$$\frac{1}{\sigma(E, C)} = 1 - \theta \left[ 1 - \frac{(1 - \alpha) \frac{\bar{E}}{E}}{\alpha \left[ \frac{E - \bar{E}}{C} \right]^\theta + (1 - \alpha)} \right]^{-1} \quad \text{for } E > \bar{E}, \quad (\text{A.10})$$

there is no straightforward relationship between the income elasticity of WTP and the elasticity of substitution. We can only unambiguously state that  $\sigma(E, C) < 1$  iff  $\theta \leq 0$ .

## A.2 Derivation of the good-specific discount rates

To derive the good-specific discount rates  $\rho_E(t)$  and  $\rho_C(t)$ , we gather the necessary inputs:

The FOCs of  $u(E, C, t)$

$$u_E = \alpha (E_t - \bar{E})^{\theta-1} \left[ \alpha (E_t - \bar{E})^\theta + (1 - \alpha) C_t^\theta \right]^{\frac{1-\eta-\theta}{\theta}} \quad (\text{A.11})$$

$$u_C = (1 - \alpha)C_t^{\theta-1} [\alpha(E_t - \bar{E})^\theta + (1 - \alpha)C_t^\theta]^{\frac{1-\eta-\theta}{\theta}} \quad (\text{A.12})$$

and SOCs

$$u_{EE} = -\alpha(E_t - \bar{E})^{\theta-2} (\alpha\eta(E_t - \bar{E})^\theta + (1 - \theta)(1 - \alpha)C_t^\theta) [\alpha(E_t - \bar{E})^\theta + (1 - \alpha)C_t^\theta]^{\frac{1-\eta-2\theta}{\theta}} \quad (\text{A.13})$$

$$u_{CC} = -(1 - \alpha)C_t^{\theta-2} (\alpha(1 - \theta)(E_t - \bar{E})^\theta + \eta(1 - \alpha)C_t^\theta) [\alpha(E_t - \bar{E})^\theta + (1 - \alpha)C_t^\theta]^{\frac{1-\eta-2\theta}{\theta}} \quad (\text{A.14})$$

$$u_{EC} = u_{CE} = (1 - \alpha)\alpha C_t^{\theta-1} (E_t - \bar{E})^{\theta-1} (1 - \eta - \theta) [\alpha(E_t - \bar{E})^\theta + (1 - \alpha)C_t^\theta]^{\frac{1-\eta-2\theta}{\theta}} \quad (\text{A.15})$$

are used to derive the respective elasticities of marginal utility

$$\psi_{EE} := -\frac{u_{EE}(\cdot)E_t}{u_E(\cdot)} = \frac{E_t}{E_t - \bar{E}} \left[ \frac{\alpha\eta(E - \bar{E})^\theta + (1 - \alpha)(1 - \theta)C_t^\theta}{\alpha(E_t - \bar{E})^\theta + (1 - \alpha)C_t^\theta} \right] \quad (\text{A.16})$$

$$\psi_{CC} := -\frac{u_{CC}(\cdot)C_t}{u_C(\cdot)} = \frac{\alpha(1 - \theta)(E - \bar{E})^\theta + (1 - \alpha)\eta C_t^\theta}{\alpha(E_t - \bar{E})^\theta + (1 - \alpha)C_t^\theta} \quad (\text{A.17})$$

$$\psi_{EC} := -\frac{u_{EC}(\cdot)C_t}{u_E(\cdot)} = \frac{(1 - \alpha)C_t^\theta(\eta + \theta - 1)}{\alpha(E_t - \bar{E})^\theta + (1 - \alpha)C_t^\theta} \quad (\text{A.18})$$

$$\psi_{CE} := -\frac{u_{CE}(\cdot)E_t}{u_C(\cdot)} = \frac{\alpha E(E - \bar{E})^{\theta-1}(\eta + \theta - 1)}{\alpha(E_t - \bar{E})^\theta + (1 - \alpha)C_t^\theta}. \quad (\text{A.19})$$

Using these, the good-specific discount rates are given by (cf. Equations (8) and (9)):

$$\rho_E(t) = \delta + \frac{E_t}{E_t - \bar{E}} \frac{\alpha\eta(E - \bar{E})^\theta + (1 - \alpha)(1 - \theta)C_t^\theta}{\alpha(E_t - \bar{E})^\theta + (1 - \alpha)C_t^\theta} g_E(t) + \frac{(1 - \alpha)C_t^\theta(\eta - (1 - \theta))}{\alpha(E_t - \bar{E})^\theta + (1 - \alpha)C_t^\theta} g_C(t) \quad (\text{A.20})$$

and

$$\rho_C(t) = \delta + \frac{\alpha(1-\theta)(E-\bar{E})^\theta + (1-\alpha)\eta C_t^\theta}{\alpha(E_t-\bar{E})^\theta + (1-\alpha)C_t^\theta} g_C(t) + \frac{\alpha E(E-\bar{E})^{\theta-1}(\eta - (1-\theta))}{\alpha(E_t-\bar{E})^\theta + (1-\alpha)C_t^\theta} g_E(t). \quad (\text{A.21})$$

### A.3 Proof of Proposition 1

We now derive the properties of the relative price effect,  $\Delta\rho(t) = (1-\theta) \left[ g_C(t) - \frac{E_t}{E_t-\bar{E}} g_E(t) \right]$  for  $E_t > \bar{E} > 0$  and  $\theta < 1$ , as presented in Proposition 1:

Equation (13):

$$\frac{\partial \Delta\rho(t)}{\partial \bar{E}} = -(1-\theta) \frac{g_E(t)E_t}{(E_t-\bar{E})^2} \begin{matrix} \geq \\ \leq \end{matrix} 0 \text{ for } g_E(t) \begin{matrix} \leq \\ \geq \end{matrix} 0 \quad (\text{A.22})$$

Equation (14):

$$\frac{\partial \Delta\rho(t)}{\partial \theta} = g_E(t) \frac{E_t}{E_t-\bar{E}} - g_C(t) \begin{matrix} \geq \\ \leq \end{matrix} 0 \text{ for } g_E(t) \frac{E_t}{E_t-\bar{E}} \begin{matrix} \geq \\ \leq \end{matrix} g_C(t) \quad (\text{A.23})$$

Equations (17) and (16):

$$\begin{aligned} \Delta\dot{\rho}(t) &= (1-\theta) \left[ g_C\dot{(t)} - g_E\dot{(t)} \frac{E_t}{E_t-\bar{E}} + \frac{g_E(t)\dot{E}_t\bar{E}}{(E_t-\bar{E})^2} \right] \\ &= (1-\theta) \left[ g_C\dot{(t)} - g_E\dot{(t)} \frac{E_t}{E_t-\bar{E}} + \frac{g_E(t)^2 E_t \dot{\bar{E}}}{(E_t-\bar{E})^2} \right]. \end{aligned} \quad (\text{A.24})$$

For the special case of  $g_C\dot{(t)} = g_E\dot{(t)} = 0$  and  $g_E(t) \neq 0$ , we obtain for Equation (15):

$$\Delta\dot{\rho}(t) = \frac{g_E(t)\dot{E}_t\bar{E}}{(E_t-\bar{E})^2} = \frac{g_E(t)^2 E_t \dot{\bar{E}}}{(E_t-\bar{E})^2} > 0. \quad (\text{A.25})$$

For  $g_E(t) > 0$ ,  $E_t \rightarrow \infty$  as  $t \rightarrow \infty$ . Therefore, we obtain for Equation (17):

$$\lim_{E_t \rightarrow \infty} \Delta\dot{\rho}(t) = (1-\theta) \left[ g_C\dot{(t)} - g_E\dot{(t)} \right] = \Delta\dot{\rho}^{CES}(t). \quad (\text{A.26})$$

For  $g_E(t) < 0$ , and bounded (changes in) growth rates,  $E_t \rightarrow \bar{E}$  in finite  $t$ . Therefore, we obtain for Equation (16):

$$\lim_{E_t \rightarrow \bar{E}} \Delta\dot{\rho}(t) = \infty. \quad (\text{A.27})$$



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