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WORKING PAPER

# The Social Value of the Global Food System

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# The social value of the global food system

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**Draft background paper for the Food System Economics Commission**

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## Abstract

The global food system provides nourishment to most of the world's eight billion people,[1] generates more than US$8 trillion of goods and services[2], and employs more than one billion people (Davis et al., 2023). On the other hand, the same system leaves c. 3/4 of a billion people undernourished, generates substantial health costs through unhealthy diets, and causes a range of environmental harms, including local air and water pollution, greenhouse gas emissions, and biodiversity loss.[3] Many of these negative impacts are hidden, meaning they reduce wellbeing but are either not or imperfectly accounted for by standard estimates of the economic value of the food system, most notably agricultural GDP. What then is the overall contribution of the global food system to social welfare and how might it evolve in the future along different development paths? How much greater a contribution could the global food system make to social welfare if the system followed a sustainable path? That is, how large would the net economic benefits be? This paper estimates the total economic value of the global food system in different future scenarios that integrate economic, health and environmental outcomes. It does so using the outputs of a coupled integrated assessment modelling system, which simulates the joint evolution of land use, food supply/demand, energy, climate, income and dietary health worldwide.[4] A wide range of model outputs are used to calculate social welfare using a system of nested utility functions, which is able to capture the changing relative values of income, environment and health in a structured, theory-driven way that incorporates recent developments in environmental and health economics. A novel method is used to achieve an unprecedented level of disaggregation relative to the IAM literature – outcomes are simulated and valued for representative individuals across the whole income distribution at a spatial resolution of 0.5° latitude x 0.5° longitude. Among other things, this allows the social cost of inequalities caused by the global food system to be quantified, both between and within countries. Relative to Current Trends (CT), the bundle of food policy measures contained in FSEC's Food System Transformation (FST) pathway scenario would provide a boost to social welfare equivalent to increasing global GDP by $9.6 trillion per year (on a Purchasing Power Parity basis), or about 7% of global GDP in 2020.

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[1] According to the UN Food Agriculture Organization (FAO), the percentage of the world population that is undernourished is c. 9%.

[2] Based on estimates from the World Bank of agricultural value added multiplied by the ratio post-farm to farm value in the food system (https://blogs.worldbank.org/voices/do-costs-global-food-system-outweigh-its-monetary-value).

[3] See, e.g., FOLU (2019).

[4] Led by a team at the Potsdam Institute for Climate Impact Research (PIK).

## 1. Introduction

This background paper for the Food System Economics Commission (FSEC) presents a method of calculating social welfare using the outputs of a coupled integrated assessment modelling system, which simulates the joint evolution of land use, food supply/demand, energy, climate, income and dietary health worldwide (REMIND/MAgPIE/LPJmL/MAGICC). It then applies the method to the FSEC scenarios, thereby estimating the social value of the global food system in alternative futures.

Economists often turn to the social welfare function (SWF) when asked to evaluate the consequences of different policies or courses of action. The set of policies that can be evaluated using SWFs is diverse; examples include national income tax schedules and global climate-change targets. In the context of FSEC, we can think of a policy or course of action as an application of a set of food system measures that sets the global food system on a particular scenario/trajectory.

Broadly speaking, a SWF orders social states from least preferred to most preferred by assigning each of them a real number.[5] Higher means better, though otherwise that number has no natural interpretation and so techniques have been developed for assigning it a monetary equivalent. This ordering is done exclusively based on the levels of utility or well-being attained by individual members of society (i.e., these are the domain of the function). Thus, to situate the approach in the broader ethical landscape, it is said to be consequentialist and specifically welfarist.

To order social states based on individual utility, a SWF must aggregate the utilities of different individuals. Each individual attains a level of utility in each social state. There are different bases for thinking about and determining utility[6], but in any case, it requires the application of a utility function (or set of utility functions), which itself maps the set of outcomes each individual obtains in each social state into a real number.

In many economic applications, there is just one outcome that determines utility and that is an individual's aggregate consumption of goods and services (equivalent to an individual's income, less savings). However, there can also be multiple determinants of utility. In the context of FSEC, these include incomes, diets and environmental outcomes. Unbundling the determinants of utility poses modelling challenges but has the advantage of enabling explicit and more flexible assumptions about the substitutability of different goods and services, and how their relative value changes as they become more or less scarce (Hoel and Sterner, 2007; Sterner and Persson, 2008; Traeger, 2011; Baumgärtner *et al.*, 2017; Dietz and Venmans, 2019; Drupp and Haensel, 2021). To give an example, the approach can directly address the issue of what happens when individual incomes increase, yet this happens to be accompanied by diets becoming less healthy and/or environmental damages increasing (which is not to say that such trade-offs are necessary). Intuitively, marginal improvements in diet and environmental quality become more valuable relative to marginal income along such a trajectory. By explicitly considering multiple determinants of utility, the method also

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[5] Specifically, this discussion focuses on the so-called Bergson-Samuelson SWF, after Abram Bergson and Paul Samuelson.

[6] That is, preference satisfaction, mental states such as happiness, or objective lists of outcomes of value.

has some affinity with multi-attribute utility theory (Keeney and Raiffa, 1976) and some applications of multi-criteria decision analysis.

The remainder of this paper is structured as follows. Section 2 outlines how the SWF approach is applied, starting with the SWF itself, followed by a nested structure of utility functions that handles how incomes, diets and environmental outcomes affect utility, and finally a set of damage/response functions, which convert the raw environmental variables produced by the integrated modelling system into a set of environmental goods that increase individual utility. Section 3 outlines how the parameters of this welfare model are calibrated, and Section 4 explains how the non-trivial step of calculating a monetary equivalent of the welfare change from taking food system measures is made. Section 5 reports the results.

## 2. Model

### *Social welfare and utility functions*

At the heart of the analysis is the SWF. In this paper, I use an average utilitarian SWF:

$$W = \sum_{t=0}^{T} \bar{U}_t (1+\delta)^{-t}\,, \tag{1}$$

where $W$ is a real-valued measure of social welfare, $\bar{U}$ is average utility at time $t$ and $\delta$ is the utility discount rate. The initial year $t=0$ is 2020 and the final year for which I have data on all dimensions from the integrated assessment model $t=T$ is 2050.

The use of average utilitarianism is open to debate. In cases where population does not vary across scenarios, classical/total utilitarianism may be preferred (i.e., substituting average utility in (1) with total utility over the population). However, population often varies between FSEC scenarios (e.g., between different SSP socio-economic scenarios), thus welfare analysis using classical/total utilitarianism may in principle lead to a scenario being preferred (assigned a higher value of *W*) just because it has a higher population. The aim is to avoid this outcome, given FSEC is focusing on food system interventions that would have at most indirect effects on total population.

Individual utility depends on measures of (i) income, (ii) environmental quality and (iii) health. Average utility is calculated over a set of individuals *i* using the following function:

$$U_{i,t} = \frac{1}{1-\eta}\left[a_C C_{i,t}^{\rho_C} + (1-a_C)\left[a_E E_{i,t}^{\rho_E} + (1-a_E) H_{i,t}^{\rho_E}\right]^{\frac{\rho_C}{\rho_E}}\right]^{\frac{1-\eta}{\rho_C}}, \tag{2}$$

where $C$ stands for consumption/income[7], $E$ for environmental quality and $H$ for health. The structure of the utility function assumes $E$ and $H$ are combined in a nest to form non-material consumption and this is in turn combined with material consumption to generate overall utility. This structure is supported by evidence on the substitutability of $C$ and $E$ being similar to that between $C$ and $H$ (Drupp and Haensel, 2021). The parameter $\rho_C \in (-\infty, 1]$ governs the substitutability of material and non-material consumption, while $\rho_E \in (-\infty, 1]$ governs the substitutability of environment and health. Further, $\rho_d = 1 - 1/\sigma_d,\ d \in \{C, E\}$, where $\sigma_d$ is the elasticity of

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[7] These concepts will be treated as interchangeable, even though in reality (dis)saving drives a wedge between consumption and income. The integrated assessment modelling system provides income as an output, not consumption.

substitution between the two elements of utility in question. Thus, the function assumes a constant elasticity of substitution (CES). The parameter $a_C \in [0,1]$ is the share of material consumption in utility relative to non-material consumption, similarly $a_E \in [0,1]$ is the share of environment in non-material consumption relative to health.

The parameter $\eta$>0 is the elasticity of marginal utility. This is assumed to be positive, so there is diminishing marginal utility with respect to consumption, environment and health. This in turn has the effect of introducing aversion to inequality in utility, both over time and between individuals at time *t*. In the latter respect, the analysis incorporates inclusion concerns, albeit in a consequentialist/utilitarian fashion. That is, the analysis puts more weight on individuals with lower incomes, who experience lower environmental quality, and attain lower health outcomes. As $\eta$ is a constant, the function also assumes a constant elasticity of substitution of overall consumption between time periods and between individuals.

***Environmental quality, and health***

Health is a function of dietary health specifically and is measured in terms of years of life lost per capita (YLL). These are converted into the health index *H* (a good) using the following health 'damage function':

$$H_{i,t} = 1/\left(1 + \gamma_H \mathrm{YLL}_{i,t}{}^2\right), \tag{3}$$

where $\gamma_H$ is the health damage coefficient. Only the diet-related FSEC scenarios contain variation in deaths avoided.

Environmental quality *E* is a function of (i) climate services, (ii) local ecosystem services, and (iii) local nutrient surplus. The determinants of *E* reflect what is available from the integrated assessment modelling system. The three elements of environmental quality are combined using a nested CES function,

$$E_{i,t} = \left(a_G G_{i,t}^{\rho_G} + a_B B_{i,t}^{\rho_G} + a_N N_{i,t}^{\rho_G}\right)^{1/\rho_G}, \tag{4}$$

where *G* stands for global climate services, *B* for local ecosystem functioning and *N* for the absence of local nitrogen pollution, $\rho_G \in (-\infty, 1]$ governs the substitutability of each of these, and the share parameters $a_G + a_B + a_N = 1$ and are individually non-negative.

Each measure of environmental quality represents a transformation of the raw outputs from the integrated assessment modelling system.

Global climate services: global mean surface temperature *T* is used to calculate global climate services using the following function,

$$G_{i,t} = 1 - \gamma_G T_t^2. \tag{5}$$

Thus, global climate service flows are a quadratic decreasing function of temperature, with the steepness of the slope governed by the coefficient $\gamma_G$. Note that all individuals' climate service provision is the same, therefore conceptually this part of the model tracks a global public good rather than local effects, which would require estimates of local temperature.

Local ecosystem services: the value of the Biodiversity Intactness Index (BII) is used to calculate local ecosystem services using the following relationship,

$$B_{i,t} = \left(1 - \left(1\text{-BII}_{i,t}\right)\right)^{\gamma_B} = \text{BII}_{i,t}{}^{\gamma_B}. \tag{6}$$

Thus, local ecosystem services are an increasing function of BII. The BII is the estimated percentage of the original number of species that remain and their abundance in any given area. Isbell *et al.* (2015) argue that theoretical and empirical results from ecology support a coefficient $0 < \gamma_B < 1$, so local ecosystem service flows are a decreasing function of BII. That means the loss of ecosystem functioning tends to be small for initial losses in biodiversity but increases more steeply as further biodiversity is lost.

Local nitrogen pollution: local nutrient surpluses cause a wide range of environmental effects via the nitrogen cascade, including local air and water pollution. Agriculture is a major source of reactive nitrogen in the environment. A variable *N* is defined – the good – which is inversely proportional to the local nutrient surplus as estimated by the integrated assessment modelling system:

$$N_t = 1/\left(1 + \gamma_N \text{nsurplus}_{i,t}^2\right), \tag{7}$$

where $\gamma_N$ is the slope coefficient and nsurplus is the local nutrient surplus in units of kg N/ha/yr.

***Notion of an individual (level of disaggregation)***

Data on BII and nutrient surplus are available on an 0.5° latitude x 0.5° longitude grid. Income and health data are available at the country level. However, further disaggregation of income is possible, because for each country estimates of GDP per capita and the Gini coefficient are provided. Assuming income is lognormally distributed over the population of each country, GDP per capita and the Gini coefficient can be used to estimate the mean and standard deviation of the income distribution using the following pair of formulae,

$$\mu_t = \ln(\text{GDPpcap}_t) - \sigma_t^2/2, \tag{8}$$

$$\sigma_t = 2\text{erf}^{-1}(\text{GINI}_t). \tag{9}$$

In turn, the mean and standard deviation of the income distribution can be used to estimate individual incomes at different percentiles of the distribution.

Putting these data sets together, I approximate a distribution of individuals *i* within each 0.5° x 0.5° grid cell. Thus, in principle every individual worldwide can experience a unique combination of income, environmental quality and health. The income of each individual depends on their position on the national income distribution. Dietary health is uniformly distributed across individuals within a country. The local ecosystem services and nitrogen pollution experienced by each individual vary by grid cell, but within a grid cell they are uniformly distributed. Global mean temperature is the same for all individuals worldwide by definition. This relatively high level of disaggregation enables inequality/inclusion concerns to be incorporated to a much fuller extent than is usual in integrated economy-environment modelling.

## 3. Calibration

The above model of social welfare contains a set of parameters to be calibrated. Some of these parameters have been estimated by previous literature and those estimates can be imputed directly. For example, there is an extensive literature on the utility discount rate $\delta$ and the elasticity of marginal utility of consumption $\eta$. While these parameters remain the subject of vigorous debate, it is relatively straightforward to obtain a measure of central tendency from the range of estimates in the literature (e.g., from Drupp *et al.*, 2018), plus the range itself can be used in sensitivity analysis. Estimates are also available for $\rho_C$, the substitutability of material and non-material consumption (Drupp and Haensel, 2021), and some of the damage function parameters.

For the remaining parameters – including the share parameters, some of the substitution parameters and some of the damage function parameters – there is a lack of previous estimates based on empirical evidence. This is a problem facing all research that seeks to directly specify utility functions depending on non-market goods, including the papers cited in the introduction. Given this challenge, calibration of these remaining parameters relies to a large extent on expert judgement, including judgements made by other scholars about corresponding parameters in previous studies.

However, it is still possible to partially constrain these unknown parameter values using empirical evidence. The model can be used to compute implicit shadow prices of the environmental and health variables, then these can be checked against corresponding empirical estimates and the unknown parameters tuned until they match. Implicit shadow prices of the environmental and health variables are given by the marginal rate of substitution of consumption for the variable in question. For (i) dietary health, this is $\partial U_{i,t}/\partial YLL_{i,t} / \partial U_{i,t}/\partial C_{i,t}$. This marginal rate of substitution is the monetary value of a statistical life year and can be compared with the extensive literature on the same quantity.[8] The same procedure can be followed for (ii) GHG emissions and (iii) local nitrogen pollution, two quantities for which there are empirical literatures estimating shadow prices.[9] Thus, a set of three implicit shadow prices is obtained, which the calibration procedure seeks to match with empirical counterparts by varying the unknown parameters. There are more unknown parameters than shadow prices, so this approach cannot uniquely identify all the unknown parameters. But equally, many combinations of unknown parameter values cannot be reconciled with the set of empirical shadow prices.

The elements of individual utility are measured on different scales because the units differ. Consumption is measured in dollars, while the environmental and health variables, through their respective damage-function transformations (3) and (5)-(7), end up being measured on an index from zero to one.[10] Therefore, the share parameters must be estimated with care. Setting $a_C$, the share of material consumption in utility, to a value of 0.7 does not imply that material consumption has a 70% share of utility given that consumption is measured on a different scale to the health/environment composite. Therefore, calibration of the share parameters is further informed

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[8] To do this, the individual/spatial unit *i* and time period *t* for which the comparison is made need to be specified. I use global average values in 2020.

[9] The procedure naturally also yields a shadow price of the Biodiversity Intactness Index, but this is less useful for calibration as empirical counterparts do not exist.

[10] *G* can be exactly zero whereas zero is an asymptote for *B* and *N*.

by explicitly targeting particular shares of each element in overall utility using data on average consumption, health and environmental outcomes in the first period, 2020. For example, to obtain a share of material consumption in utility of 70%, I calculate $a_C$ = 0.19.

***Parameter values***

Table 1 lists the parameters of the model, their values and the sources used for calibration.

*Table 1. List of parameters, values and notes on calibration.*

| Parameter | Description | Value | Source |
|---|---|---|---|
| $\boldsymbol{\delta}$ | Utility discount rate/pure rate of time preference | 0.5% | Drupp *et al.* (2018) expert survey |
| $\boldsymbol{\eta}$ | Elasticity of marginal utility of consumption | 1.01 | Drupp *et al.* (2018) expert survey |
| $\boldsymbol{a_C}$ | Share of material consumption in utility | 0.19 | Calibration (target share of 0.7) |
| $\boldsymbol{\rho_C}$ | Substitutability of material and non-material consumption | 0.23 | Drupp and Haensel (2021) meta-analysis |
| $\boldsymbol{a_E}$ | Share of environment in non-material consumption | 0.7 | Calibration (target share of 0.5) |
| $\boldsymbol{\rho_E}$ | Substitutability of environment and health | 0.01 | Assumption (approximates Cobb-Douglas) |
| $\boldsymbol{a_G}$ | Share of global climate services in environmental quality | 0.5 | Calibration |
| $\boldsymbol{a_B}$ | Share of local ecosystem services in environmental quality | 0.25 | Calibration |
| $\boldsymbol{a_N}$ | Share of local nutrient surplus in environmental quality | 0.25 | Calibration |
| $\boldsymbol{\rho_G}$ | Substitutability of climate services, local ecosystem services and local nutrient surplus | 0.01 | Assumption (approximates Cobb-Douglas) |
| $\boldsymbol{\gamma_H}$ | Health damage coefficient | 328 | Calibration |
| $\boldsymbol{\gamma_G}$ | Temperature damage coefficient | 0.016 | (Drupp and Haensel, 2021) |
| $\boldsymbol{\gamma_B}$ | Biodiversity damage coefficient | 0.3 | (Isbell *et al.*, 2015) |
| $\boldsymbol{\gamma_N}$ | Nitrogen damage coefficient | 3E-4 | Calibration |

## 4. Calculating the change in welfare

Welfare *W* lacks an intuitive measure and, in any case, utility is only unique up to a positive, affine transformation (changes in parameters change the range of *W*), so it is standard to express changes in welfare using a money metric.

A simple way to do this is to convert the difference in *W* between any pair of scenarios into an equivalent amount of money using the marginal utility of (material) consumption in 2020. However, this method faces complications. First, the marginal utility of consumption depends on the levels of consumption, environmental quality, and health. For this simple conversion of the overall difference in *W* into money units, a single combination of consumption, environmental quality, and health must be chosen, for example it could be the 2020 average. But this will only approximate the weighted average marginal utility of consumption calculated at the values actually enjoyed by each individual and it could be a poor approximation. Second, this method relies on a marginal (first-order) approximation of what could be a large, non-marginal difference in welfare between scenarios.

An alternative method is to calculate equivalent variations in consumption for each individual, and then discount and average these variations across all individuals. Take two food system scenarios, Current Trends (CT) and a Food System Transformation (FST) pathway that provides better environmental and health outcomes. For each *i* and *t*, one can calculate the level of consumption that, when combined with CT environmental and health outcomes, delivers the same utility as the FST consumption, environmental and health outcomes:

$$\hat{C}_{i,t}^{BAU} = \left[\frac{1}{\alpha_C}\left((1-\eta)U_{i,t}^{FSDP}\right)^{\frac{\rho_C}{1-\eta}} - \frac{1-\alpha_C}{\alpha_C}\left(X_{i,t}^{BAU}\right)^{\rho_C}\right]^{\frac{1}{\rho_C}}.$$

(10)

where *X* denotes non-material consumption.

The difference between this level of consumption and CT consumption is the equivalent variation:

$$EV_{i,t} = \hat{C}_{i,t}^{BAU} - C_{i,t}^{BAU},$$

(11)

Each *i*'s stream of *EV* over time is then discounted back to 2020 using individual-specific consumption discount factors priced on the CT trajectories, before the average is taken over all individuals. This can be summed across all individuals to give an aggregate amount.

Once a monetary equivalent of the difference in *W* is obtained, there remains one last question – how to express it in an intelligible way. Recall the difference in *W* is measured as the discounted sum of utility flows over thirty years (2020-2050). Therefore, taking the monetary equivalent of this difference and expressing it relative to annual income today (2020, say) would yield an extremely large proportion that is liable to be misinterpreted. Arguably a more intuitive measure of the relative monetary value of the welfare gain is obtained by converting it into an annuity that pays out over the analysis period, i.e., 2020-2050. That is, this measure tells us what constant flow of income from 2020 to 2050 would be equivalent to the monetary value of *W*, and we can express that as a share of current income.

## 5. Results

Table 2 presents the results of the analysis. The top row of results is the headline: it shows that the bundle of food policy measures contained in the FST scenario would increase social welfare globally, relative to a CT scenario anchored on SSP2. The monetary equivalent value of the increase in social welfare is US$9.6 trillion per year (in 2020 Purchasing Power Parity prices) or 7.2% of global GDP in 2020.

The next three rows of results decompose the overall welfare increase into the contributions from changes to income, environmental quality and health. The FST increases incomes for the majority of people (Bodirsky *et al.*, 2023), and slightly reduces income inequality across countries, both of which are socially valuable in the above framework. The welfare value of these income changes is equivalent to boosting global GDP by US$3.5 trillion per year, or 2.6% in 2020. The FST also improves environmental quality on all dimensions, which is worth the equivalent of US$3.7trn per year, or 2.7% of global GDP in 2020. Dietary health improvements due to the FST are worth the equivalent of US$2.3trn per year, or 1.7% of global GDP in 2020.

In the next set of rows, the impact of specific policy bundles can be seen – on diets, livelihoods, the biosphere, agriculture, and adding external transformations. These all increase the social value of the global food system but by differing amounts. Besides external transformations, a scenario which adds favourable socio-economic trends from the SSP1 scenario, the largest increase in social welfare comes from dietary measures, followed by agriculture, livelihoods and biosphere.

Table 2 also contains a sensitivity analysis, which tests the sensitivity/robustness of the results to variations in the key parameters of the social welfare, utility and damage functions. The results are robust to many parametric variations. The only sensitive dependence is to the share of material consumption in utility, $\alpha_C$. The social value of the FST is higher, the lower is this material consumption share, because more weight is put on improvements in environmental quality and health, and FST delivers larger relative improvements in these outcomes than in incomes. However, it is important to note that low-end/high-end values for the share of material consumption are hard to reconcile with empirical data on the shadow prices of health, carbon emissions and nitrogen pollution, via the calibration procedure explained above. Thus, although the results depend sensitively on this parameter, its value is significantly constrained by data.

The last part of the table compares the FST with CT using a different SSP scenario as the reference. The FST scenario increases social welfare regardless of the SSP, but the increase is highest for SSP4 and lowest for SSP1. The combination of the FST with SSP1 increases welfare by a large amount relative to CT with SSP2 socio-economic trends.

*Table 2. Welfare changes resulting from comparing different food system scenarios, including one-factor-at-a-time sensitivity analysis.*

| Scenario | Reference scenario | Parametric variation | Interpretation | Change in welfare (% global GDP) | Change in welfare (2020 $US trn) |
|---|---|---|---|---|---|
| **SSP2FST** | SSP2CT | - | - | 7.15% | 9.64 |
| **Decomposition** | | | | | |
| **SSP2FST, income only** | SSP2CT | - | Income from SSP2FST, CT environment and health | 2.60% | 3.50 |
| **SSP2FST, environment only** | SSP2CT | - | Environment from SSP2FST, CT income and health | 2.74% | 3.69 |
| **SSP2FST, health only** | SSP2CT | - | Health from SSP2FST, CT income and environment | 1.68% | 2.27 |
| **Food system measures** | | | | | |
| **Diets** | SSP2CT | - | - | 5.08% | 6.85 |
| **Livelihoods** | SSP2CT | - | - | 2.24% | 3.02 |
| **Biosphere** | SSP2CT | - | - | 1.13% | 1.53 |
| **Agriculture** | SSP2CT | - | - | 2.62% | 3.54 |
| **External transformations** | SSP2CT | - | - | 18.83% | 25.37 |
| **Sensitivity analysis** | | | | | |
| ***Welfare parameters*** | | | | | |
| **SSP2FST** | SSP2CT | $\delta = 0.1\%$ | Low pure time preference rate | 7.30% | 9.83 |
| **SSP2FST** | SSP2CT | $\delta = 2.5\%$ | High pure time preference rate | 6.48% | 8.74 |
| **SSP2FST** | SSP2CT | $\eta = 0.5$ | Low elasticity of marginal utility of consumption | 7.47% | 10.06 |
| **SSP2FST** | SSP2CT | $\eta = 2.4$ | High elasticity of marginal utility of consumption | 6.43% | 8.66 |
| ***Goods shares and elasticities of substitution*** | | | | | |
| **SSP2FST** | SSP2CT | $\alpha_C = 0.48$ | High consumption share of 90% | 3.65% | 4.92 |
| **SSP2FST** | SSP2CT | $\alpha_C = 0.09$ | Low consumption share of 50% | 14.14% | 19.04 |
| **SSP2FST** | SSP2CT | $\alpha_E = 0.5$ | Low environment share of 25% in environment/health nest | 7.51% | 10.11 |
| **SSP2FST** | SSP2CT | $\alpha_E = 0.9$ | High environment share of 75% in environment/health nest | 6.79% | 9.14 |

| | | | | | |
|---|---|---|---|---|---|
| **SSP2FST** | SSP2CT | $\rho_C$ = -1 | Low substitutability of material and non-material consumption | 6.43% | 8.67 |
| **SSP2FST** | SSP2CT | $\rho_C$ = 1 | High substitutability of material and non-material consumption | 7.29% | 9.81 |
| **SSP2FST** | SSP2CT | $\rho_G$ = -1 | Low substitutability of environmental goods | 7.89% | 10.63 |
| **SSP2FST** | SSP2CT | $\rho_G$ = 1 | High substitutability of environmental goods | 6.82% | 9.19 |
| **SSP2FST** | SSP2CT | $\rho_E$ = -1 | Low substitutability of environment/health | 7.34% | 9.88 |
| **SSP2FST** | SSP2CT | $\rho_E$ = 1 | High substitutability of environment/health | 6.99% | 9.42 |
| ***Damage function parameters*** | | | | | |
| **SSP2FST** | SSP2CT | $\gamma_H$ = 164 | Low health damages | 6.43% | 8.67 |
| **SSP2FST** | SSP2CT | $\gamma_H$ = 492 | High health damages | 7.71% | 10.39 |
| **SSP2FST** | SSP2CT | $\gamma_G$ = 0.008 | Low temperature damages | 7.12% | 9.59 |
| **SSP2FST** | SSP2CT | $\gamma_G$ = 0.024 | High temperature damages | 7.19% | 9.68 |
| **SSP2FST** | SSP2CT | $\gamma_B$ = 0.1 | Less linear biodiversity damages | 7.13% | 9.61 |
| **SSP2FST** | SSP2CT | $\gamma_B$ = 0.5 | More linear biodiversity damages | 7.17% | 9.66 |
| **SSP2FST** | SSP2CT | $\gamma_N$ = 1.5E-4 | Low nitrogen damages | 6.50% | 8.76 |
| **SSP2FST** | SSP2CT | $\gamma_N$ = 4.5E-4 | High nitrogen damages | 7.52% | 10.12 |
| **FST versus CT on different SSP scenarios** | | | | | |
| **SSP1FST** | SSP1CT | - | - | 5.89% | 7.93 |
| **SSP1FST** | SSP2CT | - | - | 23.31% | 31.41 |
| **SSP3FST** | SSP3CT | - | - | 6.44% | 8.68 |
| **SSP4FST** | SSP4CT | - | - | 7.38% | 9.95 |
| **SSP5FST** | SSP5CT | - | - | 6.37% | 8.58 |

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## References

Baumgärtner, S. *et al.* (2017) 'Income inequality and willingness to pay for environmental public goods', *Journal of Environmental Economics and Management*, 85, pp. 35–61.

Bodirsky, B. L. *et al.* (2023). A food system transformation can enhance health, environmental conditions and social inclusion. https://doi.org/10.21203/rs.3.rs-2928708/v1

Davis, B. *et al.* (2023) 'Estimating global and country-level employment in agrifood systems', *FAO Statistics Working Paper Series, No. 23-34*. Rome.

Dietz, S. and Venmans, F. (2019) 'The endowment effect, discounting and the environment', *Journal of Environmental Economics and Management*, 97. doi: 10.1016/j.jeem.2019.01.010.

Drupp, M. *et al.* (2018) 'Discounting disentangled', *American Economic Journal: Economic Policy*, 10(4), pp. 109–134.

Drupp, M. and Haensel, M. (2021) 'Relative Prices and Climate Policy: How the Scarcity of Nonmarket Goods Drives Policy Evaluation', *American Economic Journal: Economic Policy*, 13(1), pp. 168–201.

FOLU (2019) *Growing Better: Ten Critical Transitions to Transform Food and Land Use*.

Hoel, M. and Sterner, T. (2007) 'Discounting and relative prices', *Climatic Change*. Springer, 84(3–4), pp. 265–280.

Isbell, F. *et al.* (2015) 'The biodiversity-dependent ecosystem service debt', *Ecology Letters*, 18, pp. 119–134.

Keeney, R. L. and Raiffa, H. (1976) *Decisions with Multiple Objectives: Performances and Value Trade-Offs*. New York: Wiley.

Sterner, T. and Persson, U. M. (2008) 'An even Sterner review: introducing relative prices into the discounting debate', *Review of Environmental Economics and Policy*. Oxford University Press, 2(1), pp. 61–76.

Traeger, C. P. (2011) 'Sustainability, limited substitutability, and non-constant social discount rates', *Journal of Environmental Economics and Management*. Elsevier, 62(2), pp. 215–228.