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TOPICAL REVIEW

Is working less really good for the environment? A systematic review of the empirical evidence for resource use, greenhouse gas emissions and the ecological footprint

Miklós Antal[1,2], **Barbara Plank**[2], **Judit Mokos**[3,4] **and Dominik Wiedenhofer**[2]

1 MTA-ELTE Lendület New Vision Research Group, Eötvös Loránd University, Pázmány P. s. 1/A, Budapest 1117, Hungary
2 Institute of Social Ecology, University of Natural Resources and Life Sciences, Vienna, Austria
3 MTA-ELTE Theoretical Biology and Evolutionary Ecology Research Group, Budapest, Hungary
4 Evolutionary Systems Research Group, Centre for Ecological Research, Tihany, Hungary

E-mail: antalmi@gmail.com

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Abstract

Is reducing paid working time (WT) a potential win-win climate change mitigation strategy, which may simultaneously serve environmental sustainability and human well-being? While some researchers and commentators frequently refer to such 'double-dividends', most climate and environmental discussions ignore this topic. The societal relevance of paid WT and the potential role of its reduction as a demand-side measure for mitigating the climate- and ecological crisis calls for a critical review of the evidence. Here we systematically review the empirical, quantitative literature on the relationships between paid WT and a number of environmental indicators: resource use (incl. energy), greenhouse gas emissions and the ecological footprint. We applied two comprehensive search queries in two scientific databases; screened ~2500 articles published until December 2019, and used citation snowballing to identify relevant research. However, we only found 15 fully relevant studies, as well as a number of partially relevant ones. This literature employs substantially different scopes, indicators and statistical methods, each with important caveats, which inhibits a formal quantitative evidence synthesis but usefully informs a critical discussion of the research frontier. Most studies conclude that reductions in paid WT reduce environmental pressures, primarily by decreasing incomes and consumption expenditures. However, existing research does not provide reliable guidance beyond the established link between expenditures and environmental impacts. Quantifying the effects of time use changes and macro-economic feedbacks through productivity, employment, and the complementarity or substitution between human labour and natural resources in production processes has proven to be difficult. To better understand the environmental impacts of specific types of WT reductions, new forms of data collection as well as studies at different scales and scopes are required. The critical discussion of the existing literature helps to conceptually map the pathways investigated so far and to identify crucial next steps towards more robust insights.

1. Introduction

Supply-side technological measures alone are insufficient to safely avoid catastrophic climate change, making complementary demand-side measures increasingly necessary (Anderson and Peters 2016, Creutzig *et al* 2016, 2018). These measures promise to curb societal resource use (energy, materials and land) and the resulting emissions, which are key drivers of environmental unsustainability (Krausmann *et al* 2018, UNEP-IRP 2019, Marques *et al* 2019). In the case of climate change, the global community committed itself to ambitious and rapid action to achieve net-zero greenhouse gas (GHG) emissions over the next few decades (UNFCCC 2015).

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A growing number of public intellectuals argue that working time reduction (WTR) could be an environmentally beneficial demand-side strategy for reducing resource use and GHG emissions (Schor 2010, Kopatz 2012, Raworth 2017, Suzuki 2017, Bregman 2018, Frey 2019). This, and WTR more generally, attracts substantial media attention (Hierländer 2018, Harper 2019, Taylor 2019, Semdley 2019, Kahn 2020, Spicer 2020). Some even believe that WTR is inevitable because automation and artificial intelligence will soon outperform humans in many jobs (Frey and Osborne 2017). Others see WTR as beneficial to substantially increase labour productivity (Kleinman 2019). Especially in the affluent Global North, WTR could (again) become a politically more relevant strategy, as the successes of unions fighting for it already show (Reiter *et al* 2018). Because decent work is such a central societal issue, it is also directly included in the Sustainable Development Goal Nr. 8 'economic growth and decent work' (United Nations 2019). Post-growth visions also emphasize the relation of work and well-being and explicitly mention WTR (The Guardian 2018).

Nevertheless, the topic of WTR is almost completely absent from most IPCC documents, climate strategies at the international, national and subnational levels, and high-level climate policy discussions (Antal 2018). Both this neglect, which may be due to the dearth of information on effects of WTR, and the enthusiasm of its proponents call for a deeper understanding of the climate and environmental impacts of this strategy. In other words, whether and how WTR schemes should be considered in climate discourse and policy is a controversial and open question.

Previous research, at least since Schor (1992) wrote about the 'work-spend' cycle in 'The Overworked American', has tried to highlight the necessity, display the variety, and study the role of WTR as a strategy to reduce resource use and emissions (e.g. Jackson and Victor 2011, Pullinger 2014, Antal 2014, Wiedenhofer *et al* 2018). In its simplest form, the idea is that people who already have enough could work less, therefore earn and consume less, which could increase well-being and reduce environmental impacts without harming employment. Of course, there are many complexities not captured by this first simple picture. For example, consumption patterns may shift towards more resource- and emission-intensive activities such as travel (Hanbury *et al* 2019). WTR may also trigger production-side changes over time, whose consequences for resource use and emissions depend on the substitutability or complementarity of labour, capital and resources across different sectors (e.g. Apostolakis 1990). Reflecting this complexity, interest in exploring the environmental impacts of WTR from various perspectives seems to be on the rise (Hanbury *et al* 2019, Schumacher *et al* 2019, Csala 2020, Lindsay *et al* 2020, Cieplinski *et al* 2021).

However, in terms of empirical evidence on the quantitative relationship between paid working time (WT) and the environment (ENV), the existing literature is scattered across different approaches and methods, making them hard to compare and synthesize. For WT, different measures, definitions and databases are used. The environmental indicators range from production-based energy and emission accounts to consumption-based carbon-, material- and ecological footprints. While all these indicators have climate-relevance because they include energy or emissions, their differences and underlying assumptions complicate comparisons. While many existing papers have short discussions and selective summaries of previous work, there is no comprehensive and critical review of the literature.

Here we conduct a systematic review of the empirical evidence on the environmental impacts of WTR, aiming for a critical understanding of the knowledge base and necessary next steps to assess WTR as a demand-side mitigation strategy. For this purpose, we employ a reproducible, systematic method to identify the relevant scientific literature on the quantitative relationship between paid WT and resource use, GHG emissions and the ecological footprint (section 2). We summarize the literature (section 3) and critically discuss its strengths and weaknesses (section 4). Then we draw together theoretical arguments and perspectives from the literature and explore promising directions for future research (section 5). Finally, we draw conclusions regarding WTR as a demand-side mitigation measure (section 6).

2. Systematic review method and criteria

To uncover all the relevant literature, a systematic search process has been conducted (figure 1).

In **step 1**, *two search queries* were developed and used to find relevant documents in Scopus and the Web of Science Core Collections, covering all entries until 2 December 2019. The search was limited to English-language titles, keywords and abstracts, yielding 2494 records.

In **step 2**, titles and abstracts of each record were screened for relevance by the first author, leaving 66 studies. Between reviewer-consistency was tested by assigning 100 records to the second author, with 100% inclusion/exclusion agreement. Studies were deemed fully relevant *if they investigated the relationship between paid working time and an environmental indicator (such as energy use, materials use, emissions and the ecological footprint), both indicators were quantitative, referred to the same system (e.g. a household, a state or a country), and the relationship was studied on the basis of measured values.* A number of studies were partially relevant, as they dealt with the same relationship but did not satisfy all conditions. These were moved to step 6.

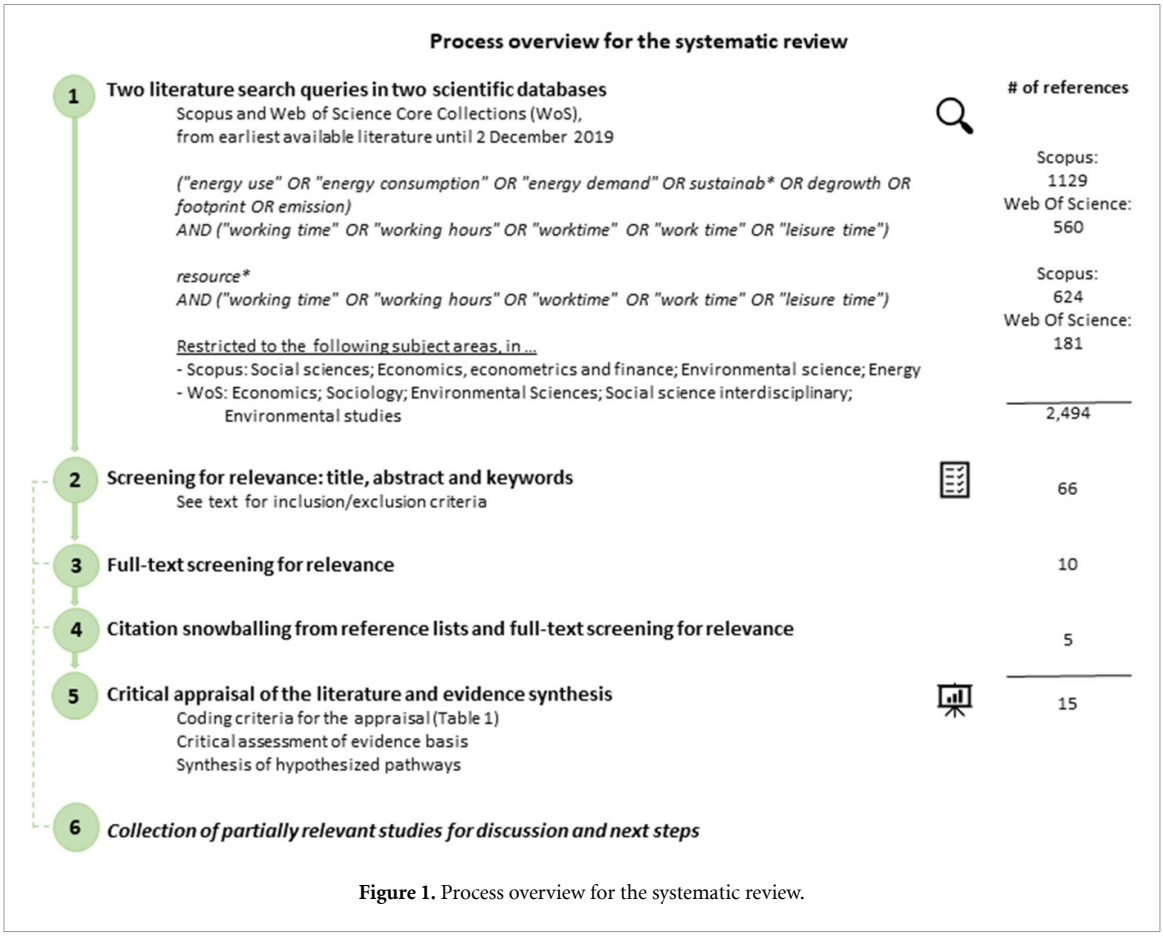

Figure 1. Process overview for the systematic review.

In **step 3**, full texts were screened, leaving ten references as fully relevant.

In **step 4**, reference lists of all relevant papers were checked ('citation snowballing'), yielding another ten potentially relevant papers. Some of these were not included in the two scientific databases we used. No further studies were identified as potentially relevant from their reference lists (2nd level snowballing). After a full-text screening, five studies were included as fully relevant.

In **step 5**, the 15 full-texts were coded based on the pre-developed criteria, critically appraised, and synthesized (see table 1). Additionally, the first author corresponded with some of the first authors of the assessed literature (Jared Fitzgerald, Qing-long Shao, Anders Fremstad, Jonas Nässén).

In **step 6**, the partially relevant studies were collected and utilized for contextualization and discussion of the evidence, as well as to inform next steps. These studies are, for example: macro-economic scenario modelling exercises; models of GHG emissions in which time use had an important role; articles in which environmental behaviour was linked to time use but not WT specifically; research on the environmental intensity of time-uses; and a number of theoretical and qualitative papers.

An important limitation of our approach is that it does not necessarily find broader descriptions of WTR schemes where ENV impacts are not specifically highlighted. Such papers may still be relevant if they discuss selected environmentally important lifestyle changes or strategies of stakeholders that matter for income, expenditures or environmental behaviour[5]. However, these papers do not provide comparable quantitative environmental indicators and would therefore require a completely different review design.

3. The empirical literature on WT, resource use and emissions

The 15 research articles identified as fully relevant are summarized in table 2 (and the supplementary information, tables S1 and S2, which is available online at stacks.iop.org/ERL/16/013002/mmedia).

Country-level studies investigate the relationship between aggregate WT and ENV indicators using econometric approaches. Besides simple regression models that try to connect the average WT of employed workers with an ENV indicator using a cross-sectional sample of countries, most papers use panel data and build on the STIRPAT approach ('stochastic impacts by regression on population, affluence and technology'). This is a multivariate non-linear model extending the classical IPAT model

[5] One example pointed out by a reviewer is the description of experiences with the four-day week at the Volkswagen Company (Promberger *et al* 1999).

Table 1. Criteria for the systematic comparative assessment for the full-text critical appraisal.

Study scope	Country/region, time period and level of analysis (country, state and households)
Sample selection	Rich countries/different country groups; all households/only above a threshold of annual hours, other selection criteria
Work time indicator	Weekly/annual working hours of workers of the country/household
Environmental indicators	Territorial (production-based)/consumption-based: energy use, materials, emissions and ecological footprint
Methodological approach	Relationship studied, statistical methods used and control variables
Acknowledgement of limitations	Robustness checks and information on explanatory power of the type of statistical analysis acknowledged by the authors
Main results and conclusions	Key quantitative WT–ENV result and headline conclusions drawn by authors

(York *et al* 2003), in which environmental impacts (I) were conceptualized as the product of population size (P), affluence (A: per capita GDP), and technology (T: environmental impacts per unit of GDP). STIRPAT can be written as $I_i = aP_i^b A_i^c T_i^d e_i$ where a is a scaling constant, while exponents b, c, d and the error term e describe the relationship for unit i (e.g. a country in a year). WT enters these models in two ways: through the *scale effect* (Schor 1992) and the *composition effect* (Nässén *et al* 2009).

The scale effect means that higher work hours yield a higher level of economic output, i.e. refers to the contribution of WT to GDP. Most studies in our sample test for the scale effect by disaggregating per capita GDP into three parts—the employed population ratio, the average WT of employees, and labour productivity (GDP per hour)—and isolate WT as the variable of interest while using the other two as control variables in the econometric model. The composition effect is the impact of WT apart from its contribution to GDP, which stems from household-level consumption decisions influenced by the availability and uses of non-work time and household incomes[6]. In econometric analyses, it is isolated net of GDP (WT is the variable of interest and GDP per capita is a control variable). Additional control variables (urbanization, manufacturing/service ratio of GDP, energy production, etc.) are also included.

Household-level studies investigate the WT–ENV relationship using econometric methods, but with WT and ENV indicators referring to individual households, not entire economies. Two main methods have been used so far. The less comprehensive option is to define WT brackets (e.g. low, medium, high) and use logit regressions to analyse the prevalence of selected environmentally relevant types of consumption in each (Devetter and Rousseau 2011). The other, more common approach is to study the relationship between total household WT and a consumption-based energy/CO_2/material footprint indicator, which is calculated from all expenditures of the household (Nässén and Larsson 2015, Buhl and Acosta 2016, Fremstad *et al* 2019). In both cases, a number of control variables can be used (e.g. household characteristics).

Figure 2 gives a summary of the main channels through which WT and ENV indicators have been connected in the literature.

4. Critical appraisal of the empirical literature

We address four groups of questions to discuss the challenges and limitations of understanding the ΔWT–ΔENV relationship:[7] (a) Which system is studied? (b) How is WT measured? (c) How is WT measured? (d) How are conclusions drawn? Figure 3 gives an overview of the relevant methodological choices and the associated concerns, which are covered in the following subsections.

4.1. Which system is studied?

The ΔWT–ΔENV relationship cannot be analysed at the global level because WT is not measured consistently. By focusing at lower levels, some impacts of ΔWT will not be captured. One potentially important effect is the sufficiency rebound (Alcott 2008), i.e. that lower demand by those who implement a sufficiency strategy—e.g. by reducing WT and consumption—may reduce prices and increase consumption by others. If consumption grows in units excluded from the analysis, then the ENV impacts of WTR may be overestimated[8]. More generally, the smaller the studied system, the more impacts will occur beyond its boundaries. This is an important limitation for all household-level studies.

On the other hand, the larger the studied system, the more impacts not related to ΔWT can be expected to influence the ENV indicator. A key question is how important WT (or ΔWT) is compared to drivers of ENV (or ΔENV) that are excluded from the analysis.

[6] This is a consumption-side perspective put forward in the reviewed papers. For a discussion of production-side impacts and how they interact with consumption, see section 5.

[7] Δ refers to changes of variables. Many studies talk about both a static WT–ENV and a dynamic ΔWT–ΔENV relationship.

[8] The price interactions will be stronger if the entities whose ENV indicator changes are more strongly connected through markets, so it might be a more serious limitation for household level studies.

Table 2. Summary of the empirical literature on WT versus resource use and emissions. For a more detailed table see the supplementary information. If not stated otherwise, total CO_2 emissions include emissions from fossil fuel use and industrial processes.

Authors	Level of analysis	Countries investigated	Working time indicator	Environmental Indicator		Key conclusions
Fitzgerald et al. 2018	Country/state level	USA	Average weekly working hours	Production-based/territorial perspective	CO_2 emissions from fossil fuels	US states with higher WT tend to have higher CO_2 emissions (2007-2013)
Shao & Shen 2017		EU-15	Average annual working hours		Total CO_2 emissions and primary energy consumption	Different effects at different WT levels on CO_2
Shao & Rodríguez-Labajos 2016		Mix of countries	Average annual working hours		Total CO_2 emissions per capita	Higher WT was associated with higher emissions in rich countries (1980-2000), but relationship turned negative (2000-2010)
Shao 2015		EU-15	Average annual working hours		Total CO_2 emissions and primary energy consumption	Higher WT mostly associated with higher energy and CO_2, but large regional differences (1970-2010)
Fitzgerald et al. 2015		Mix of countries	Average annual working hours		Primary energy consumption	WT effect on energy use was negative in the past (1990), positive later (2008), strengthened over time
Knight et al. 2013		OECD	Average annual working hours		Total CO_2 emissions	Positive relationship between WT and CO_2 (1970-2007)
Rosnick & Weisbrot 2007		EU-15 & USA	Total national working hours		Primary energy consumption	Positive relationship between WT and energy use, especially in high-income countries (2003)
Knight et al. 2013		OECD	Average annual working hours	Consumption-based/footprint perspective	Ecological footprint	Positive relationship between WT and ecological footprints (1970-2007)
Hayden et al. 2009		Mix of countries	Average annual working hours		Ecological footprint	Positive relationship between WT and ecological footprints (2000)
Schor 2005		OECD	Annual working hours per employee and per working-age person		Ecological footprint	Significant positive correlation between WT and ecological footprints
Fremstad et al. 2019	Household level	USA	Annual household working hours		Total CO_2 footprint	Positive (but not very strong) relationship between WT and CO_2
Buhl & Acosta 2016		Germany	Weekly working hours		Material footprint	Positive income and negative time use effect on material use
Nässén & Larsson 2015		Sweden	Paid working time in minutes		GHG footprint (excl. land use change), primary energy footprint	WT increases energy use and GHG, income effect larger than time use
Devetter & Rousseau 2011		France	Work hours per week per household		Environmentally harmful types of consumption	WT increases likelihood of environmentally intensive consumption
Nässén et al. 2009		Sweden	Work and non-work time use in minutes		GHG footprint (excl. land use change), primary energy footprint	WT increases energy use and GHG, positive income and slightly negative time use effect

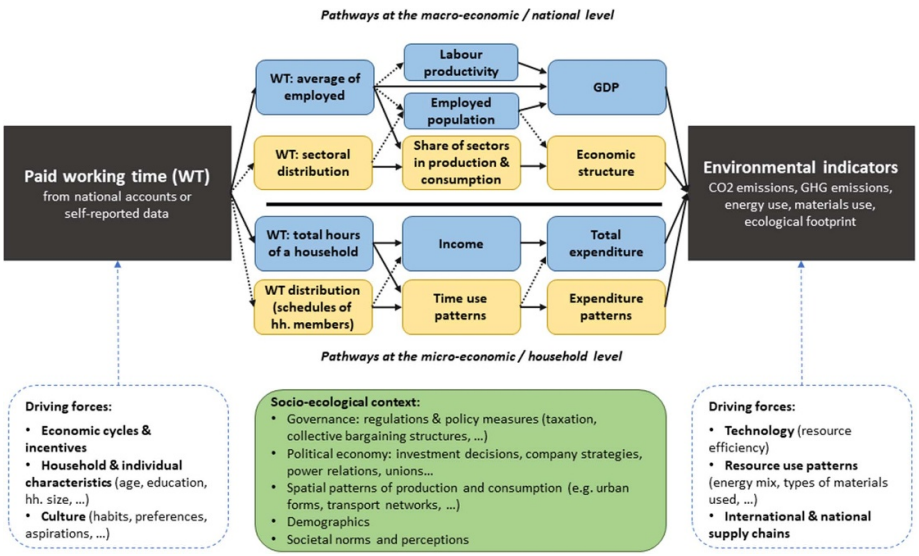

Figure 2. WT–ENV pathways at the macro-economic and micro-economic levels. Dashed arrows denote weaker relationships. Blue boxes matter for scale, yellow boxes for composition.

ΔWT-ΔENV				
concerns	⇩ countries	⇦ system studied ⇨	households ⇩	concerns
sample not large	only affluent countries	sample	all households in a survey	heterogeneity
indicators not consistent & various types of changes	various countries		certain types of households: survey subsample	
small, open economies	states of a country		separate surveys for income and time use effect	data from different households
country specific effects cannot be controlled for	cross-sectional	temporality	cross-sectional	representativity for WTR not guaranteed
changes in WT-ENV, esp. structural breaks	longitudinal		multiple cross-sectional	
			longitudinal	limited availability
WT				
database reliability low for some countries: differences & revisions	average WT of employed population	indicator	paid working time — total hh WT	number of individuals in hh, indirect ΔWT
			paid working time — total hh WT: bracket	can be too crude
potentially different trends for excluded groups	average WT of a part of the employed population (e.g. nonfarm private employees)		income	changes in net savings: lagged effects important
			expenditure	uncertain changes as a result of WTR
changes over time comparable, annual values not	from national accounts (calculated using various sources & complex, country-specific adjustments)	measurement	self-reported (survey)	database, edition, known biases
data not representative	business surveys		time use diary	database, time period
ENV				
fuel mix as a driver	total primary energy supply	indicator	energy use	monetary (not physical) indicator, no government consumption included
LULUCF missing, database differences	CO_2 emissions		CO_2 emissions	
database reliability	GHG emissions		materials use impacts	
uncertain calculation, difficult interpretation	ecological footprint		probability of environmentally relevant types of consumption	important ENV impacts missed
may be driven by trade	territorial	measurement	expenditure survey + IO based footprint calculation	problematic expenditure categories, IO/ MRIO
calculation reliability	consumption-based		budget survey data	
methods and conclusions				
method insufficient	simple regression	statistical method & interpretation	simple WT-ENV regression	cross-sectional data not representative of WTR
multicollinearity	based on the STIRPAT model			
unobserved variables			income regression + time use model	income effect and time use effects not independent
statistical issues			logit regression	
different types of ΔWT simultaneously affect WT-ENV relationship effects beyond boundaries (e.g. sufficiency rebound) lagged effects of ΔWT on ENV ΔWT-ΔENV not necessarily linear ΔWT-ΔENV may change over time				

Figure 3. Methodological choices and associated concerns and limitations of WT–ENV studies and implications for WTR. Grey refers to a study that did not attempt to comprehensively estimate ENV impacts.

In the reviewed country-level studies, the answer is not encouraging: the importance of unobserved variables seems to be enormous. Changes in resource use and emissions are strongly driven by technical and structural changes in sectors like electricity, transport, buildings, and business/industry (Jackson *et al* 2019). We tested how changes in the electricity mix affect the results of Fitzgerald *et al* (2018) and found a very substantial influence[9]. All other reviewed studies may be similarly affected as none of them control for changes in the efficiency of power plants or the changing fuel mix. Similar issues will arise for transport, buildings and industrial production. Because WT usually changes by less than 2% per year (and often <1%), changes due to unobserved variables are likely to have substantially larger effects than ΔWT. Therefore, there is a great risk of finding spurious relationships.

Regarding the temporal dimensions of the system under study, how much of ΔENV appears immediately matters. If there are substantial lagged effects, e.g. because everyday practices slowly adapt to lower WT and incomes, or because of macro-economic feedbacks, then studies assuming a contemporaneous relationship give an incomplete or inaccurate picture. This problem is generally ignored in the empirical literature (table 2).

4.2. How is WT measured?

Country-level studies usually use average annual hours per employed worker obtained from datasets based on national accounts (table 2). Calculation methods of these WT values are not internationally standardized, so they are only suitable for comparisons of trends over time and not between countries for a given year (de Vries and Erumban 2017, OECD 2020). WT data from labour force surveys differ from these values by up to 10%–20%, with large variations between countries (Bick *et al* 2019b). Unfortunately, none of the reviewed studies have used adjusted, consistent WT values, which have only become available recently for a subset of countries (Wingender 2018). In some of the reviewed papers, data availability was also a reason to use WT indicators that excluded certain groups of workers[10].

The reliability of WT values is a further question. To test this, we compared average annual WT from the databases of The Conference Board (TCB) and the OECD, both of which are widely used in country-level studies (supplementary information, table S1). The results[11] show that annual changes in the differences between the datasets are sometimes larger than changes of the WT values themselves, potentially undermining statistical analyses (supplementary information, figure S1). Similarly, trends in the differences between databases are concerning as the strength of any effect of ΔWT on ΔENV will appear to be different depending on the database. In addition, Bick *et al* (2019a) report substantial revisions of WT data in the OECD and TCB databases: e.g. the difference between US and EU values has changed by 40% between the 2003 and 2016 releases of the same database.

Obtaining appropriate WT data at the household level is also difficult. Using WT data of individuals is inadequate because changes of individuals' WT in a household are often strongly coupled, e.g. through the unpaid work of other household members (Jacobs and Gerson 2001, Lewis *et al* 2008, Wielers *et al* 2014, Spiegelaere and Piasna 2017). Total household WT depends on the number of household members who work as well as the working hours of each. As ENV may depend on how total WT is shared, the ideal dataset contains information on the number of individuals as well as their WT values, as in Fremstad *et al* (2019). Without such data, the other reviewed studies analyse the effects of hypothetical WTR schemes that are assumed to affect WT, incomes or expenditures in specific ways (e.g. proportionally reduce all three).

At the household level, the source of WT data matters for reliability. When WT values are self-reported, which is usually the case when no diary-based time use dataset is applied, then its known biases should be kept in mind. For instance, long hours are usually overreported while short hours are underreported (Frazis and Stewart 2014), increasing the width of measured WT distributions. Besides, WT estimates are less reliable if many workers are self-employed or have irregular schedules (Niemi 1993, Robinson and Bostrom 1994, Bonke 2005, Walthery and Gershuny 2019), which makes statistical estimates more noisy.

4.3. How is ENV measured?

Most country-level studies use territorial/production-based ENV indicators (table 2). Total primary energy

[9] They study territorial CO_2 emissions from fossil fuels for US states. The data they used (EPA 2017) is broken down by main sectors, showing that in 31 out of the 50 states CO_2 emissions were mainly driven by changes in the electricity sector. Taking total power generation values from the EIA (2020) reveals that in 23 of these states, the main reason was the change of the fuel mix, not the change of total power generation. (An additional remark is that North Dakota, one of their outliers, had 10%, not 20% increase in overall emissions.)

[10] For example, Fitzgerald *et al* (2018) exclude public and farm employees, i.e. 15%–20% of the total labour force from their analysis of US states. Whether this makes a significant difference is unknown, but the study period is 2007–2013 when the economic turmoil may have had different effects in different sectors. Furthermore, some of their outlier states had radical changes in the private sector (a fracking boom) which may not have affected the excluded sectors.

[11] The values for many European countries are exactly the same. However, for other countries, such as the USA, Mexico, Japan or Korea (randomly chosen non-EU countries), which are all included in several WT–ENV studies (Rosnick and Weisbrot 2007, Hayden and Shandra 2009, Fitzgerald *et al* 2015), there are significant differences.

supply (TPES) denotes the total amount of primary energy that a country has at their disposal[12]. It includes domestic energy production and energy imports and excludes energy exports and fuels used for international shipping and aviation[13]. Territorial indicators for CO_2 emissions represent either only the domestic combustion of fossil fuels, or in some reviewed studies also industrial processes such as cement and steel production. None of the studies use complete emissions accounts from agriculture, forestry and other land uses, as well as land use changes (AFOLU emissions) (Smith *et al* 2014).

From a WT–ENV perspective, these production-based indicators have important weaknesses. Excluding emissions from international flights and shipping can cause an error of 10% or more, especially for small and affluent countries (e.g. the Netherlands), with substantial trends over time (van Goeverden *et al* 2016, Eurostat 2020). If industrial emissions from cement production are excluded (as in Fitzgerald *et al* 2018), then changes in the building sector may be lost. Excluding emissions from land use change and forestry underestimates impacts of food and other biomass products (Houghton 2020), which are crucial for tropical countries[14] and for certain impacts through lifestyle changes (e.g. diet shifts and the use of biofuels). Not accounting for other GHGs means that potentially relevant emissions (e.g. methane from food production) are not considered. Even more critical is that none of these production-based ENV indicators reflect the resource/energy/carbon impacts of imports and exports, which make up 10%–30% of emissions attributable to consumption (Wiebe *et al* 2012), or even more (∼50%) for small and open economies like US states (Erickson *et al* 2012) or European countries (Tukker *et al* 2016). Trends of territorial ENV indicators have been strongly influenced by changes in energy/emissions embodied in trade, which grew very significantly in the early 2000s and flattened out around 2010 after the global financial crisis (Pan *et al* 2017, Wood *et al* 2019a).

In addition, even for the ENV indicators for which official reporting is most accurate, data reliability remains a concern. To show this, we compared territorial CO_2 emissions from fossil fuels per capita from the databases of the Global Carbon Project and the World Bank (supplementary information, figure S2)[15]. The analysis shows that differences in reported emissions can be substantial, up to 20% in the chosen case. These differences can have structural breaks (1989–1990 in figure S2). Moreover, changes of the difference between the databases is very often comparable, and often larger, than the change of the values themselves (supplementary information, table S3). In addition, sometimes there are substantial revisions of ENV values, especially for countries with weaker institutions or lower commitment to transparency (e.g. Liu *et al* 2015).

More comprehensive ENV indicators are even less reliable, e.g. because of the uncertainties of AFOLU emissions (Petrescu *et al* 2020) or non-CO_2 GHG emissions (IPCC 2014). Further complexities and uncertainties characterize consumption-based footprint indicators that attribute all resources/energy/emissions occurring along international supply chains to final consumption. The ecological footprint is conceptually the most comprehensive consumption-based ENV indicator, which provides a measure of global environmental pressures due to food consumption, housing, transportation, consumer goods, and services[16]. This indicator, used in several reviewed papers (table 2), has been heavily criticized, not least because of the arbitrary weighting of the various environmental problems (e.g. van den Bergh and Verbruggen 1999, Ayres 2000, Wiedmann and Barrett 2010, van den Bergh and Grazi 2014). Besides the complicated interpretation, inaccuracies in the data used to calculate the ecological footprint and its carbon component may also make its statistical use misleading (Jóhannesson *et al* 2020).

At the household level, ENV indicators are not measured directly. Two main types of indicators have been used so far. Devetter and Rousseau (2011) considered selected types of consumption as binary ENV indicators, e.g. 'electricity bills belonging to the highest quartile' or 'having a large house (more than 6 rooms)'. The justification for this approach is that certain types of consumption have substantially higher resource/emission intensities than others (Jalas 2002), and are easier to measure. However, picking consumption categories may bias evaluations towards directly accessible information and risks ignoring other effects.

The more comprehensive, but also more difficult, approach is to account for the environmental impacts of all types of consumption corresponding to the lifestyle(s) of household members using footprint estimation methods (Nässén and Larsson 2015,

[12] It measures energy contents before transformation to other end-use fuels, e.g. the energy content of coal or gas used in a power plant, not the electricity.

[13] Changes of fuel stocks are also included, but these are usually small.

[14] E.g. Brazil or Colombia, which are included in Fitzgerald *et al* (2015) and Shao and Rodríguez-Labajos (2016).

[15] We chose four European countries (France, Denmark, Germany and the Netherlands), for which Shao and Shen (2017) suggest a negative WT–ENV relationship, i.e. increasing ENV as a result of decreasing WT.

[16] Environmental pressures from these are converted into a common metric: global hectares, representing the amount of productive land and water areas at average world productivity required to continuously produce the resources and assimilate the wastes associated with consumption (Wackernagel and Beyers 2016). A large fraction of it is due to the theoretical land area required to hypothetically sequester annual emissions. International trade flows are simplified into 'national yield factors'. The National Footprint Accounts are the most widely used EF dataset, providing data for most countries and the world for 1961–2014, based primarily on publicly available UN datasets (Lin *et al* 2018).

Buhl and Acosta 2016, Fremstad *et al* 2019). Footprint indicators are usually derived from expenditure survey data using input-output (IO) analysis, which maps the structure of economies by quantifying supply chain interactions[17].

A number of limitations follow. First, household expenditure surveys do not capture all consumption. For instance, some employers provide direct assistance for transport or housing. More importantly, government consumption and investments are not attributed to households, so—among many other effects—emissions linked to large construction projects are not visible in household-level footprints (Chen *et al* 2018).

Second, IO modelling assumes homogenous prices for each product group (Miller and Blair 2009). This may cause problems, e.g. for energy in countries with liberalized energy retail markets (all case studies reviewed here). In Germany, price differences are up to 20%–30% (Gugler *et al* 2018) and household energy use is responsible for 25%–30% of CO_2 footprints (Gill and Möller 2018), resulting in 5%–10% uncertainty of household level emissions. Sufficient sectoral/product-detail is thus crucial: IO analysts usually suggest differentiating at least 30–50 expenditure categories. Studies with substantially fewer categories, such as the reviewed paper by Buhl and Acosta (2016), are therefore limited.

Third, it attributes resource use/emissions proportional to monetary flows (Weisz and Duchin 2006). This likely overestimates ENV impacts at high levels of expenditure in consumption categories like housing (or cars) where costs are driven by location (or brand names) without proportionally changing the resource use/emissions associated with their production. Conversely, at low levels of expenditure ENV impacts may be underestimated (Girod and De Haan 2010).

Fourth, results are sensitive to the allocation of resources and emissions to economic sectors and to errors in trade data (Owen *et al* 2017)[18]. Studies using national IO models, such as Nässén and Larsson (2015) and Fremstad *et al* (2019) may also be problematic if imports are substantially cleaner/dirtier than domestic products (e.g. for small, open economies with clean electricity like Sweden) (Lenzen *et al* 2004). This is especially important if the composition of household consumption is systematically related to WT.

[17] Officially reported data from national accounts, trade statistics, energy, materials and emissions reporting and official surveys are combined into a consistent modelling framework (Miller and Blair 2009).

[18] For comparisons across studies, another complicating factor is that different IO-models exist and the underlying datasets for emissions and resource use (Owen *et al* 2014, Wood *et al* 2019b) as well as expenditure surveys, national accounts and multi-regional IO tables (Min and Rao 2018) can all differ slightly, affecting footprint estimates.

4.4. How are conclusions drawn?

Before discussing how conclusions are drawn at different levels, a general comment is relevant for all reviewed studies. Changes of WT can take place in various ways (Pullinger 2014, De Spiegelaere and Piasna 2017) with potentially divergent effects on WT and ENV indicators, as well as their relationship. Table 3 lists important differences between ΔWTs and refers to their implications regarding the studied relationship.

If different types of ΔWTs simultaneously affect the indicators, then their effects will be confounded in the analysis, preventing conclusions regarding specific types of ΔWTs. This limitation is crucial for country/state-level studies using aggregate WT indicators, and likely relevant for household level studies based on nationally representative surveys.

4.4.1. Country-level studies

WTR is usually discussed as an option for high-income countries to translate productivity growth into lower WT instead of higher consumption (Jackson and Victor 2011, Knight *et al* 2013, Antal and van den Bergh 2016). Including low- and middle-income countries in cross-country comparisons as several reviewed studies do (table 2) is questionable because not only the drivers of WT and ENV, but also their definitions can be completely different (e.g. in countries with prevalent subsistence agriculture and informal work relations)[19].

Several studies use country groups to reduce heterogeneity. However, it is unclear whether grouping is enough to avoid all problematic types of heterogeneity. For instance, if WT increases in some countries and decreases in others within a group[20], then a potentially asymmetrical ΔWT–ΔENV relationship may confuse results (Sorrell *et al* 2020). Furthermore, in some studies, country groups change over time. If countries jump from one group to another, then specific trends affecting WT or ENV in a country can distort the results or some groups may have too few countries for statistical analysis (as in Shao and Shen 2017).

With a relatively limited set of countries, the next question is how long the time period should be. There are cross-sectional studies that refer to a single year (Schor 2005, Rosnick and Weisbrot 2007, Hayden and Shandra 2009). This is inadequate because neither WT values nor ENV values are comparable[21]. For longitudinal studies, the goal is to identify a time period with enough data points and a stable ΔWT–ΔENV relationship. The relationship may change due

[19] From this perspective, a state-level analysis of the US (Fitzgerald *et al* 2018) is better because of the uniform definitions and methodology.

[20] E.g. in the USA vs. continental Europe since the 1980s.

[21] At least no feasible method has been shown so far to control for all country-specific drivers of ENV.

Table 3. Types of ΔWT and their implications for ENV impacts.

Factor	Types of ΔWT	Selected implications
Participation	• Collective in a country • Collective at an organization • Individual opt-in scheme	Influences the need for wage compensation (necessary at lower wages in collective schemes), and the likelihood of ΔWT of non-participants (workers in individual WTR schemes may be replaced, spouses of WTR participants may increase their WT if their share of unpaid work decreases)
Timing	Changes in • Daily WT • Weekly WT • Annual WT • Lifetime WT	Influences lifestyles and associated ENV impacts: 10%–20% difference between ENV indicators possible due to different timing (King and van den Bergh 2017), e.g. less commuting with more days off than with shorter workdays (Schumacher *et al* 2019, Percoco 2018), changes in heating and cooling, required building floor space, leisure travel, etc.
Financial impacts	How total income is affected & how monetary costs (if any) are shared by employees, employers and governments	Matters for competitiveness, employment and spending power, thereby influencing WT and production beyond the direct impacts (e.g. Hunt 1999, Marimon and Zilibotti 2000, Chemin and Wasmer 2009)
Length	Temporary (e.g. for a year) or permanent ΔWT	Matters for individual aspirations and associated financial and lifestyle strategies (e.g. savings, investments, moving homes, etc.)
Speed	Abrupt or step by step ΔWT	Influences perceptions of impacts and associated WT preferences
Starting point	Changes from any given initial level of WT	Influences effects on the intensity and productivity of work (reductions from 55 h per week likely increase productivity more than reductions from 15 h per week) as well as changes in patterns of consumption (forced reductions of part-time work differ from voluntary reductions of overtime)

to cultural transitions, demographic changes influencing the number of people in different life-stages, or technological innovations that affect both the ENV intensity of given time uses and the lifestyle choices reflected in time use patterns (Jalas and Juntunen 2015)[22]. Therefore, it seems necessary to avoid structural breaks when WT or ENV change abruptly for external reasons; e.g. 1990 for ex-socialist countries[23], the great recession around 2009 for most countries, and the fracking boom around 2010 in the USA[24].

While a full assessment of the statistical methods of each study is beyond the scope of this paper, we make a few observations about questionable statistical methods and results that are very difficult to reconcile with theory or common sense. First, as changes of WT can affect both productivity (Golden 2012) and the employed population ratio (Zwickl *et al* 2016, De Spiegelaere and Piasna 2017), it is surprising that not all country-level studies check multicollinearity between the three factors that make up GDP per capita. Second, some studies use methods that look inadequate, e.g. Fitzgerald *et al* (2015) apply the Prais–Winsten method even though the number of countries exceeds the number of time periods and the time period is short[25]. Third, many papers get very strange results, which should serve as (further) cautionary signs. Shao and Shen (2017) find significantly negative effects of population size and GDP per capita on carbon emissions[26]. For population, Shao (2015) and Knight *et al* (2013) get suspiciously low and high values, respectively (less than 0.3 vs.

[22] However, splitting up the time period in non-arbitrary ways is very difficult because of the various factors that may influence the strength of the ΔWT–ΔENV relationship and the very noisy data that makes statistical methods used for this separation questionable.

[23] 1990 is in the middle of the first study period for Shao and Rodríguez-Labajos (2016) who include Bulgaria, the Czech Republic, Estonia, Germany, Hungary, Latvia, Lithuania, Poland, Romania, Russia, Slovakia, and Slovenia, as well as the study period of Knight *et al* (2013) who include the Czech Republic, Estonia, Germany, Hungary, Slovakia, and Slovenia.

[24] Without the fracking boom, there would be no apparent relationship on the ΔWT-ΔENV figure in Fitzgerald *et al* (2018). Model runs without these states are said to be 'substantively similar' to the findings including them, but it would have been useful to numerically report these findings to illustrate the (in)significance of this methodological decision. The results of this study are also very questionable because the study period 2007–2013 includes the financial crisis and the great recession, which drastically changed the US economy, affecting CO_2 emissions in various ways in the different states.

[25] This was pointed out by one of the reviewers.

[26] In both cases they refer to unobserved variables to suggest potential explanations. However, the same unobserved variables may have affected their findings regarding the WT–ENV relationship, potentially invalidating their headline findings.

more than 2). Both studies get very different results for closely related ENV indicators (TPES & territorial CO_2, and carbon footprint & territorial CO_2). Fitzgerald *et al* (2015) find negative effects of the employed population ratio on TPES, whereas Fitzgerald *et al* (2018) find no significant effect of GDP per hour and GDP per capita on territorial CO_2 emissions. Despite all uncertainties, Shao and Rodríguez-Labajos (2016) draw counterintuitive conclusions regarding the ΔWT–ΔENV relationship on the basis of weak statistical results, without sufficient sensitivity analysis.

A summary of the main concerns for each country-level study is given below (table 4).

4.4.2. Household-level studies

Not all household level studies try to draw conclusions regarding the ENV impacts of the same type of WTR. To illustrate the differences, we go through the chain of impacts from ΔWT to ΔENV.

Each step in the ΔWT–Δ(income)–Δ(expenditure)–ΔENV causal chain is complex. First, the statistical relationship between ΔWT and Δ(income) tends to be weak because of large differences between hourly wages[27]. Second, annual income and expenditure indicators can be quite different, e.g. because of changes in net savings, income that is not (properly) measured (e.g. capital income[28]), and transfers between households. Third, the relationship between total expenditures and ENV may depend on personal values and choices regarding the composition of consumption.

Which of these indicators is supposed to change due to WTR has important implications for the ΔWT–ΔENV relationship. Some articles (Nässén and Larsson 2015, Buhl and Acosta 2016) assume that WTR changes WT *and* expenditures simultaneously and proportionately (for a discussion of representing the financial impacts of WTR as a change of expenditures vs. a change of incomes, see the supplementary information). The aim of these studies is to calculate two separate effects: an 'income effect', which stems from the financial impacts of WTR, and a 'time use effect', which stems from the changes in time use patterns (figure 2). Separating these would ideally require detailed longitudinal information on both the expenditures of a large sample of households and the time uses of household members. Then the consequences of changes in either just expenditures or time use could be studied using subsamples in which the other variable is unchanged.

However, such data is generally not available. In both studies using this approach, WT and expenditure data come from different datasets. The problem is that WT and income are correlated (Devetter and Rousseau 2011), so time use surveys contain some income effects (e.g. people with higher WT are more likely to have higher incomes and more expensive time uses) and expenditure surveys include some time use effects (e.g. people with higher expenditures likely have higher WT and some expenditures resulting from a time squeeze). It is not enough to consider only one of these effects because the WT–income relationship that is implicit in time use and expenditure surveys is almost certainly different from the WT–income relationship that characterises the studied type of WTR (e.g. a linear relationship)[29]. On the other hand, adding the two effects together without double counting is complicated. Nässén and Larsson (2015) recognized this[30] and attempted to calculate a pure time use effect by not only allocating different types of expenditures to time uses, but also adjusting some of the expenditure intensities to keep total expenditures constant in this part of the calculation. As the original matching between time uses and expenditures is already somewhat arbitrary (Schipper *et al* 1989, Jalas 2002, 2008, Druckman *et al* 2012), the adjusted version is bound to be very uncertain. What's more, calculating an income effect that does not include any time use effects has not been attempted, which draws into question the usefulness of efforts based on separate datasets.

Perhaps the best available data source, which contains both WT and expenditures for the same households, is used by Fremstad *et al* (2019). As detailed time use data is not included in this dataset, time use information cannot be used to better understand the pathways of ENV impacts (i.e. *why* certain expenditures change), but a straightforward WT–ENV regression is possible. By doing this, Fremstad *et al* (2019) make no assumptions about how incomes or expenditures change. (They separately calculate income and expenditure elasticities of ENV, which would correspond to assumptions of WTR changing these instead of WT. A discussion of these is included in the supplementary information.) However, they use a cross-sectional WT–ENV relationship to draw conclusions regarding the longitudinal effects of a WTR scheme, which is very questionable. The relationship between WT and ENV is so indirect that it would be very difficult to theoretically justify this approach. In the case of expenditure elasticities, one could argue that longitudinal changes of ENV may be approximated using cross-sectional data because the expenditure–ENV relationship is 'tight' (at any level of total expenditure the variation of ENV is not very

[27] There may be orders of magnitudes between the incomes of households working the same number of hours, see e.g. data from Fremstad *et al* (2019).

[28] Which can be particularly important in the group that is most likely to prefer WTR.

[29] The same limitation is true even if expenditure or time use data is longitudinal because it is unlikely that past changes have occurred under the conditions that characterize any given WTR scheme.

[30] Buhl and Acosta (2016) did not.

Table 4. Summary of specific concerns about country level studies, excluding general concerns.

Study	Main concerns	Further comment
Schor (2005)	Cross-sectional, no control variables, small sample, etc.	Self-describes as thought provoking, not as serious analysis, no conclusions possible
Rosnick and Weisbrot (2007)	Cross-sectional	No conclusions possible
Hayden and Shandra (2009)	Cross-sectional	No conclusions possible
Knight *et al* (2013)	Structural break in the middle of the period for 6 out of 29 countries. Model results suspicious: impact of population is very large	Suspicious that elasticity is 1.3 for the carbon footprint and 0.5 for territorial CO_2
Fitzgerald *et al* (2015)	Model results suspicious: impact of employed population ratio on TPES is negative. Statistical method problematic	Very large fluctuations in the WT–ENV relationship do not seem very likely
Shao (2015)	Model results suspicious: impact of population is very small. Most results on the role of WT are not statistically significant	Very large differences between EU regions and time periods in terms of WT–ENV
Shao and Rodríguez-Labajos (2016)	Structural break in first study period (collapse of socialism). Few explanatory variables or sensitivity analyses	Effect size small, interpretation not cautious enough
Shao and Shen (2017)	Inappropriate grouping of countries. Model results extremely questionable: impact of population on TPES and territorial CO_2 is negative	Data discrepancies for the countries in the headline results highlighted
Fitzgerald *et al* (2018)	Territorial emissions problematic for small, open economies. Main driver of emissions (change of electricity mix) not included in the model. Structural break in the middle of the study period	Results without outliers not shown

wide) and because characteristic expenditure patterns are followed by specific groups of people. In contrast, the WT–ENV relationship is 'weak' (at any level of WT the variation of ENV is wide) and there are no characteristic lifestyles according to WT[31]. The cross-sectional WT–income and WT–expenditure relationships in the sample used by Fremstad *et al* (2019) are very different from those in the collective WTR assumed by Nässén and Larsson (2015), in which hourly wages are kept constant. It is also incorrect to draw conclusions about the role of WT in explaining differences in ENV at the country level, not only because the cross-sectional assumption would imply a very specific—and not necessarily realistic—type of WTR, but also because effects at the macro level are neglected.

This makes it unclear whether taking annual WT and expenditure data from the same dataset is better than relying on expenditures alone to predict the impacts of a WTR scheme. It is likely that approximating income by expenditures leads to an overestimation of the ENV impacts of WTR while using annual WT or income data leads to underestimation (for the explanation see the supplementary information). Using both approaches is useful for a sensitivity analysis. Differences will depend on the datasets applied, the methods used and the type of WTR regarding which conclusions are sought.

A summary of the main concerns for each household-level study is given below (table 5).

5. Towards a robust understanding of WT (reductions) and the potential effects on resource use and emissions

In section 3, figure 2 gave a technical overview of selected variables connecting ΔWT and ΔENV, but did not account for all important causal mechanisms. To propose directions for further research, deeper

[31] Fremstad *et al* (2019) tried to reduce heterogeneity by analysing subsamples and using a number of control variables, but the proportion of the variance in the dependent variable that is predictable from the independent variables is still quite low in all of their models ($R^2 < 0.5$). Controlling for all potentially relevant factors that influence how expenditures would change as a result of WTR looks very difficult (Hanbury *et al* 2019). If there are systematic differences between unobserved drivers of ENV according to WT, which is fully possible, then longitudinal changes can be expected to differ from the cross-sectional correlation. Path-dependence (e.g. rigidities of consumption patterns) can also invalidate cross-sectional results. A further comment is that preferences to change WT under given conditions may differ substantially between different groups in the same sample, so conclusions regarding concrete WTR schemes (collective or individual, compensated or uncompensated, weekly or annual, etc.) may be different even if the same dataset is used.

Table 5. Summary of specific concerns about household level studies, excluding general concerns.

Study	Main concerns	Further comment
Devetter and Rousseau (2011)	WT indicator very crude, ENV indicator not comprehensive.	Only qualitative conclusions possible regarding the WT–ENV relationship.
Nässén and Larsson (2015)	Two non-independent effects added together—part of the problem is not recognized.	No conclusions beyond the expenditure–ENV relationship possible.
Buhl and Acosta (2016)	Two non-independent effects added together—the problem is not recognized. ENV indicator based on too few expenditure categories.	Unclear reporting of results. No conclusions possible.
Fremstad *et al* (2019)	Cross-sectional WT–ENV analysis not informative about ENV impacts of WTR in general and about country differences.	Conclusions beyond the expenditure–ENV relationship very questionable.

theoretical understanding is helpful. We start from economic changes associated with WTR and proceed towards the social structures that shape the economic system and its dynamics, drive WT related decisions, and influence the impacts of given WTR schemes. Finally, we discuss potentials and limitations of various research directions.

5.1. Economic pathways between ΔWT and ΔENV

Any type of WTR has two immediate impacts: some labour input (working hours) and some labour output (e.g. products or services) disappear from the economic system. A number of indirect impacts can be expected both on the production and the consumption side.

When labour input is reduced, production-side impacts may occur through labour productivity, production processes, and labour markets. Impacts on *labour productivity* are due to individual level changes such as less fatigue and organizational effects like the changing effectiveness of work groups (Golden 2012, Collewet and Sauermann 2017). The magnitude of these effects depends on the types of jobs as well as the type and participants of WTR. *Production processes* change because different factors of production—labour, capital, energy, and resources—can substitute or complement each other (Berndt and Wood 1979). Even if the direct labour–energy relationship is close to neutral (Cox *et al* 2014), longer term effects through substitution or complementarity with capital can be relevant (Fallon and Layard 1975, Apostolakis 1990, Frey and Osborne 2017). These relationships depend on production processes and characteristics of producers like skill levels. Effects through *labour markets* may include a variety of changes. Total employment may change, e.g. if new workers are hired to keep up production despite WTR, thereby reducing un(der)employment. The strength of this effect depends on broader economic conditions like the level of un(der)employment among workers suitable for the jobs. Besides, sectoral shifts may occur if the organizations where WTR takes place attract labour from other organizations. To what extent and on what time scale this happens depends on the relative positions of employers in the labour market and the availability of skilled labour.

Moreover, the previous three effects interact, e.g. changes in productivity and substitution by capital may influence how much labour is sought in the labour market. In addition to production-side effects, the time use of all impacted producers will change, with effects on the consumption side. The impacts of changing production on revenues/incomes at indirectly affected organizations should also not be forgotten.

Effects due to the loss of labour output on the production-side may propagate through supply chains and markets. If some organizations produce less, others may have to scale back too (e.g. buyers) or may see new opportunities for growth (e.g. competitors). On the consumption side, the total exchange value of labour is lost, which includes the compensation of workers as well as the net profits of the organizations[32]. As discussed earlier, income reductions have complex effects on consumption.

Finally, production-side and consumption-side impacts interact. Which effects dominate depends on the markets through which they are connected. Elasticities of supply and demand play important roles here. Supply-side and demand-side economists will likely see the overall impacts differently. Given that impacts have different time scales and no equilibrium is guaranteed, the least one can say is that overall impacts are very complex (figure 4).

5.2. Societal structuring of work, time uses and environmental effects

Before turning to the question of what this complexity means for future research, we look at the larger societal structures in which these economic pathways are embedded. We begin with broader theoretical

[32] Plus there is a shift of taxes and contributions between the state and the organization.

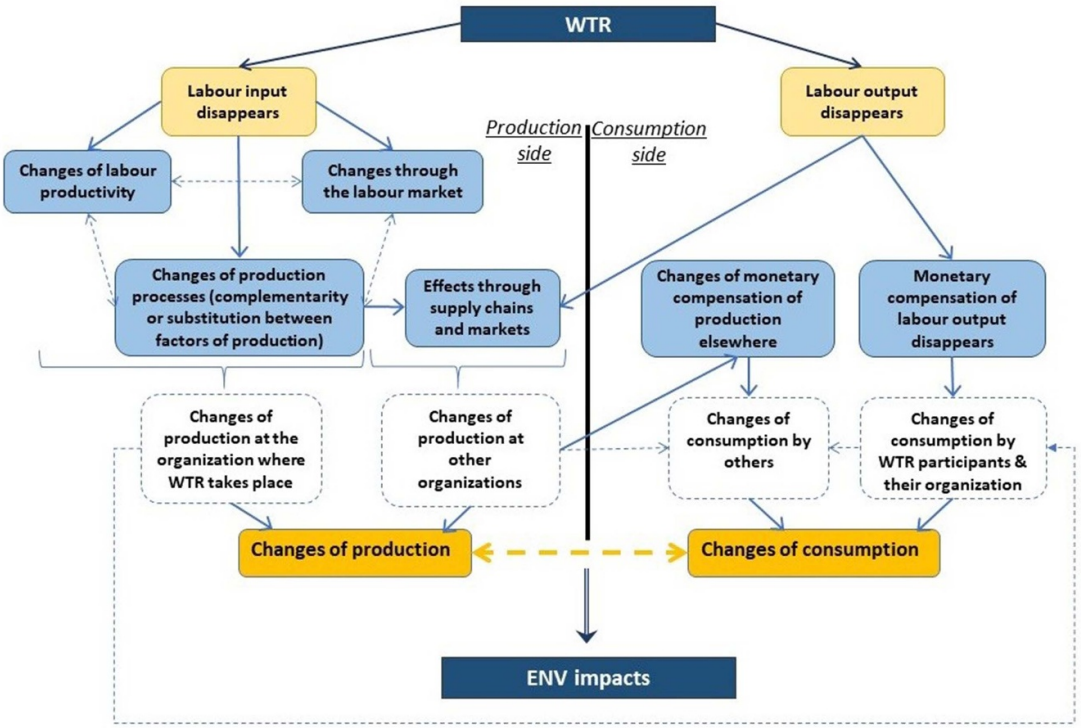

Figure 4. Economic pathways through which working time reductions influence environmental impacts.

perspectives, then mention theories that are relevant for specific causal effects in figure 4. Given the large amount of literature on the sociology of time use, we only select a few strands that are particularly focused on how people view time and how this is changing.

One broad narrative that focuses on our changing relationship with time is that of Hartmut Rosa (2013). He identifies three types of acceleration. Technical acceleration is about the increasing speed of goal-directed processes, following the economic idea that 'time is money'. Social acceleration means that social beliefs and actions have shorter periods of validity and are co-existing with often radically different other beliefs and actions. The acceleration of the pace of life is a response of modernity to cope with the ultimate limitedness of the human lifetime, which drives people to exhaust as many options as possible. These trends influence production processes, people's aspirations, and time uses. However, such general philosophical approaches have to be further specified to reach quantitative conclusions regarding the ENV impacts of WTR.

A somewhat more concrete strand in the literature talks about treadmill effects (Binswanger 2006). The 'positional treadmill' refers to the constant strive for social status relative to others (Frank 1985), the 'hedonic treadmill' describes the adaptation to higher levels of income and consumption (Stutzer 2004), and the 'time-saving treadmill' means that time-saving innovations tend to have large rebound effects and do not mitigate time pressure. A concrete example for the latter is that faster travel tends to result in longer distances covered instead of travel-time savings. Between 1965 and 2000, average travel-time budgets in affluent countries were invariant with both income and other time uses, including WT (Schäfer *et al* 2009). If travel-time budgets are still stable, then travel emissions are more strongly influenced by the costs of different modes of transportation than WT, so this theoretical prediction is informative for the ENV impacts of WTR.

However, not all time use categories and environmentally relevant behaviours show consistent patterns, so building up a general theory regarding the ENV impacts of various types of WTRs currently seems unfeasible even at the level of households. Working hours structure everyday lives and strongly influence income, so they likely shape the nature of non-work activities and associated environmental impacts, which occur within specific material settings and social obligations (e.g. Wiedenhofer *et al* 2018). Yet these behaviours can be expected to be dependent on the context, including family circumstances, worldviews, money matters, living environments, etc. (Hanbury *et al* 2019, Lindsay *et al* 2020). The diversity of these influences may be a reason why so little empirical evidence has been found so far for direct impacts of work-life balance on environmental behaviour (Kennedy *et al* 2013, Melo *et al* 2018). We also note that a number of environmentally important decisions, e.g. on larger investments, have not been investigated as a function of

WT so far. In addition, various rebound effects may cancel out significant parts of initial environmental benefits (Sorrell *et al* 2020). At the systemic level, effects are even more diverse. Seemingly beneficial lifestyles like telework have ambiguous ENV effects (Hook *et al* 2020) and WTR schemes may substantially differ from each other (King and van den Bergh 2017). Therefore, instead of general theories we suggest a series of context-specific investigations.

5.3. Implications for further empirical research

Considering the limitations of the approaches used so far, we suggest several research strategies that may help to better understand the environmental and climate change mitigation potentials of WTRs.

At the level of countries, the most important step forward is the assessment of the reliability of statistical approaches using aggregate data. One main problem is that many types of ΔWTs occur simultaneously, and only the cumulative effects of these changes appear in aggregate data. Therefore, both the patterns of ΔWT (whose WT changes and how) and the extent to which these ΔWTs differ from each other in terms of ENV impacts must be understood better. The former could be achieved through labour market investigations focusing on the types and conditions of ΔWT, while the latter requires smaller-scale case studies (see below). Another main problem is that current statistical studies do not control for key drivers of ENV indicators. To assess how this affects reliability, the role of ΔWT in ΔENV could be studied in individual countries. Substantial changes in WT over relatively short periods could be analysed first, considering the various sectoral drivers of ΔENV and assessing the role of ΔWT in the overall change. Natural experiments in which different regions of a country have different regulations could also be useful to study if appropriate disaggregated data is available (Chemin and Wasmer 2009).

One possible outcome is that case studies find mechanisms that completely invalidate country-level statistical approaches. For example, simultaneous ΔWTs with opposite ENV effects would make aggregate indicators useless. Country-level statistical comparisons are also inadequate if uncertainties in sectoral drivers of ΔENV turn out to be substantially larger than the contribution of ΔWT. The other possible outcome is that no such prohibitive difficulties are identified. Then the limitations discussed in section 4 should be addressed as much as possible. A longitudinal approach is necessary and structural breaks should be avoided. Countries should be sufficiently similar in terms of ΔWT and ΔENV in the sample: checking and arguing these will be easier after the case studies. Internationally comparable datasets should be used for WT (Bick *et al* 2018, Fuchs-Schündeln 2019), and various ENV indicators should be tested in the same study. Changes of public and private debt should be monitored.

At the household level, a main reason for the lack of reliable studies is the limited availability of consistent data. Too often, expenditure surveys and time-use surveys are conducted separately and matching these requires many uncertain assumptions. Therefore, collecting and using data on the same households for WT and expenditures is strongly preferable. Such data must be longitudinal to be informative regarding impacts of WTR. In particular, a series of 'before-after' case studies from different contexts would be extremely useful. Building on early efforts (Promberger *et al* 1999), information should be collected on expenditures and time uses simultaneously, considering savings and loans, as well as changes of aspirations. Using relatively small samples, it might be possible to explore the WT–income–expenditure–ENV chain in detail. Like in the country-level case, this would also help to estimate the reliability of results based on larger samples but less detailed information. One could assess the importance of various neglected factors like capital incomes, a safety net through inter-household relations, etc.

In these suggested case studies, key behaviours and associated expenditure categories—such as travel, which may have become an important leisure activity, attracting a disproportionate share of discretionary income (Oswald *et al* 2020)—need special attention. Better understanding these categories and their ENV impacts using both monetary and physical terms may worth separate studies. Similarly, using physical terms for household energy consumption is advisory. These could then feed into more comprehensive assessments using a sufficient number of consumption categories.

Another useful direction is to study cases where WT is reduced while incomes are preserved, like in a Swedish trial a few years ago (Oltermann 2017). These may reveal a pure time use effect. If income is registered in time use diaries, that may also help modelling the effect. For a first step towards estimating a pure income effect, it could be useful to use a combined WT and expenditure dataset (e.g. from the USA) and compare the income–expenditure relationship at given WTs (this is feasible because there are many households at standard levels of WT) with the income-expenditure relationship of the whole sample. To collect longitudinal data, case studies of WT-neutral salary increases and decreases could be beneficial. Whether pure time use and income effects can be combined to understand overall effects at households is a further topic to be explored.

Before jumping to conclusions, household level studies must be complemented by estimates of impacts at the company and macro-economic levels. Whether WTR changes productivity in given sectors is one question. Combining quantitative productivity indicators with qualitative data looks suitable to explore this. Another question is about indirect

impacts on other workers' WT (including overtime hours) and employment, which could help to understand how different WTR schemes change WT at the aggregate level. Recent modelling efforts represent a useful step towards understanding this (Cieplinski *et al* 2021). Analysing the substitution or complementarity of labour, capital and resources in various production processes is not only useful to develop such models, but also to investigate production-side effects of WTR. In each of these cases, specifying the types of WTR under study will be important.

6. Conclusions

Current research on the potentials of WTR as a demand-side climate change mitigation measure remains inconclusive. While the positive relationship between income/expenditure on the one hand and resource use/emissions on the other is established (e.g. Hubacek *et al* 2017, Oswald *et al* 2020), the complexity of the ΔWT–ΔENV relationship and the lack of sufficiently detailed data on both time use effects and systemic feedbacks (including production-side effects) precludes strong conclusions. This review summarized the existing literature, uncovered a number of methodological limitations, and suggested ways forward.

In the case of all reviewed country-level studies, fundamental questions were raised about the validity of conclusions. Some studies used cross-sectional samples with severe comparability issues, which makes them inadequate. Longitudinal studies based their results on very noisy data without controlling for the main confounding drivers of environmental indicators, also potentially confusing different types of ΔWTs whose ENV impacts may differ. To test the usefulness of country-level statistical approaches, we proposed country-level case studies focusing on individual countries.

The reviewed household-level articles that aimed to comprehensively assess ENV impacts are also questionable. They either calculated total ENV impacts as a sum of two non-independent effects (the income effect and the time use effect), or used a cross-sectional WT–ENV relationship to draw conclusions regarding longitudinal effects of WTR, even though the respective WT–income–expenditure–ENV relationships are likely to be different. The most important step towards rectifying these problems would be the simultaneous, longitudinal collection of time use (or WT) and expenditure data. With this approach, a series of context-specific investigations could provide very valuable insights. Studies on particularly important behaviours and impacts, like those related to mobility, could help to make assessments more reliable, especially if they also collect physical, not just monetary, data. In addition, separate investigations of production-side effects on productivity, the substitution or complementarity between factors of production, and labour market impacts, as well as investigations of macro-level feedbacks should complement these analyses. Taken together, these suggestions represent a significant shift of research directions in this field.

One should keep in mind that understanding current WT–ENV relationships is not all that is needed to estimate the potential environmental impacts of WTRs. At least at the household level, environmentally beneficial WTR looks possible (Hanbury *et al* 2019), so the other main question regards the conditions under which such benefits are achievable. ENV impacts of WTR schemes depend on various factors that may be changed, such as social norms and urban structures (Kennedy *et al* 2013, Kallis *et al* 2013). Research on these drivers is as important as research on the past or current relationship itself.

Due to the Covid-19 pandemic, both working lives and environmentally relevant behaviours are changing. Exploring how, under these new circumstances, different types of WTRs may affect environmental and climate-relevant indicators is a very important task for the future. Unlike currently dominant supply-side and technological solutions to environmental problems, WTR may provide a new vision of more socially and environmentally sustainable societies.

Data availability

No new data were created or analysed in this study.

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Author contributions

MA designed the research, searched the literature, participated in coding & led the analysis and paper writing. BP led coding, participated in analysis and

paper writing. JM participated in coding and statistical discussions, provided technical help. DW participated in research design, analysis and paper writing.

ORCID iDs

Miklós Antal https://orcid.org/0000-0003-3426-9916
Barbara Plank https://orcid.org/0000-0002-0306-3715
Dominik Wiedenhofer https://orcid.org/0000-0001-7418-3477

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