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## Underestimates of methane from intensively raised animals could undermine goals of sustainable development

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E-mail: [Matthew.hayek@nyu.edu](mailto:Matthew.hayek@nyu.edu)**Keywords:** livestock, sustainable development, methane budget, climate change mitigation, agriculture, GHG modeling, dietary change**Abstract**

Greenhouse gas emissions from meat and dairy production are often highly uncertain; these emissions are typically estimated using inventory-based, ‘bottom-up’ models, which contain uncertainties that are difficult to quantify. Modeled emissions estimates can be corroborated using atmospheric measurements—taken above and downwind of animal production regions—to produce ‘top-down’ emissions estimates. Top-down and bottom-up estimates of animal methane show good agreement when considering global emissions. However, in the US, where animal production is predominantly highly intensified with confined feeding operations, animal methane emissions may be 39%–90% higher than bottom-up models predict (expressed as mean differences across studies). Animal emissions may grow in the future as meat and dairy demand increases in developing countries. We examine East and Southeast Asia as a test case, where emissions from increased meat and dairy production are expected to be offset by improved efficiency from intensive methods. We adjust the share of direct emissions projected to come from intensive systems by the intensities derived from US top-down estimates. We find that region-wide emissions from meat and milk production could reach 1.52 (1.41–1.62) GtCO<sub>2</sub>eq by 2050, an amount 21% (13%–29%) higher than previously predicted. Therefore, intensification may not be as effective in mitigating emissions in developing countries as is commonly assumed.

**1. Introduction: the role of intensification in sustainable development**

Global meat consumption is expected to increase by 50% in the coming decades (FAO 2018a). This increase is primarily driven by rising affluence in developing low- and middle-income countries. Animal agriculture has been reported to represent 15.6% of total annual greenhouse gas (GHG) emissions globally (FAO 2017 using a 100 year global warming potential (GWP<sub>100</sub>) for non-CO<sub>2</sub> GHGs), with emissions expected to increase in the coming decades.

Limiting GHGs from agriculture is urgent because business-as-usual agricultural growth is likely incompatible with limiting warming below 1.5 °C (Clark *et al* 2020). Reducing emissions from food systems is therefore urgent, even in scenarios in which fossil fuel usage is rapidly phased out.

Sustainable intensification offers a possible solution; it is defined as the production of more meat and dairy with fewer resources, like land and feed, and lower emissions per animal (Godfray *et al* 2010).

The majority of global agricultural emissions are estimated to come from lower efficiency, pastoral systems (Steinfeld *et al* 2006). These systems include long-range grazing, backyard, and mixed crop-livestock systems, although the latter category has wide variation, and could count as intensive or semi-intensive depending on the specifics of the system (Robinson *et al* 2011). These forms of animal agriculture are referred to as extensive production systems because of their larger land footprint, and they are becoming less common globally (Gerber *et al* 2013).

Conversely, intensive production systems, like those in North America and the EU, are likely

responsible for fewer direct and indirect emissions, in total and per unit of meat or dairy produced (Gerber *et al* 2013). Intensively managed animals in these regions are often raised in ‘landless’ systems (as defined by the UN Food and Agriculture Organization, FAO), where feed inputs are traded commodities and are largely detached from the immediate local resource base (Robinson *et al* 2011). Animals like pigs and chickens are highly confined for all of their lives, while beef cattle spend the latter part of their lives being ‘finished’ on enriched feeds in feedlots. These systems in the US are also referred to as concentrated animal feeding operations (CAFOs). However, this term is a more stringent categorization—as CAFOs may not include a number of smaller-scale facilities that are nonetheless highly industrialized and entail confinement.

Animal feeding in intensive, landless systems is oriented toward rapid weight gain and productivity (Robinson *et al* 2011). Animals live shorter lives relative to extensively-raised animals to optimize feed requirements and other input resources (Herrero *et al* 2013). Intensive systems for raising chickens, pigs, and dairy cows, and facilities for fattening or ‘finishing’ mature beef cattle to reach market weight in the North America restrict animal movement. Intensive landless systems typically do not allow for the expression of natural behaviors, access to pasture, or the ability to graze or forage for food. This type of animal agricultural production, often referred to as ‘industrialized production’, is now proliferating globally through developing low- and middle-income countries.

Intensively-raised animal-sourced foods have lower land and feed requirements, relative to extensively-raised animal-sourced foods (Swain *et al* 2018). The benefits of improved feed requirements are commonly assumed to include fewer GHG emissions: both lower indirect emissions from feed production and grazing, and fewer direct emissions from wastes like manure, urine, and belches, per unit of food produced. However, the aggregate air and water pollution from landless facilities, along with resource depletion of fresh water, can create local problems and contribute to environmental degradation (Carrel *et al* 2016, Richter *et al* 2020).

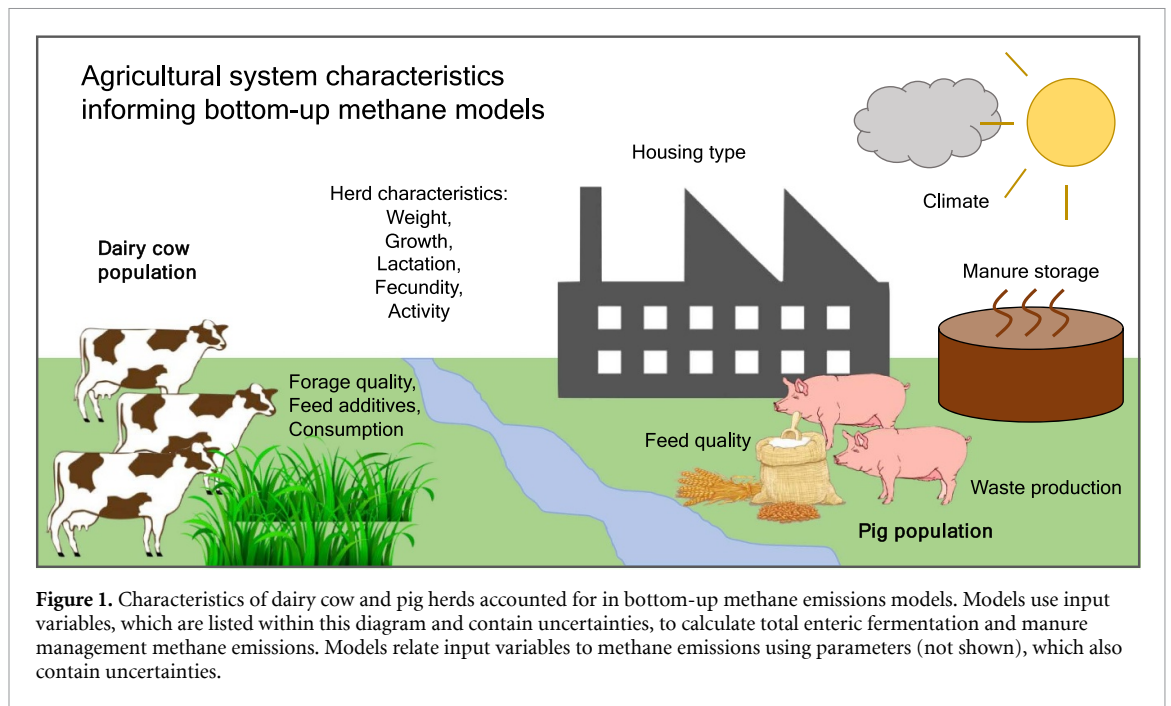
Researchers have therefore recommended intensifying existing extensive and pastoral systems to reduce GHG emissions (e.g. Steinfeld *et al* 2006, Thornton and Herrero 2010, Swain *et al* 2018). North American meat and dairy companies frequently claim that their emissions are improving, directly attributing this trend toward more intensive management practices (Cattlemen’s Beef Board 2018, McDonald’s Canada 2018, National Pork Board 2020). In unity with

these messages, governments and nongovernmental organizations alike have promoted intensification for food security, environmental improvements, and GHG mitigation in low and middle-income countries (De Oliveira Silva *et al* 2018, FAO 2018b, Enahoro *et al* 2019, ILRI 2019). International development banks further incentivize intensive methods through conditional lending and low-interest loans for confined production infrastructure (Wasley and Heal 2020, World Bank International Finance Corporation 2020).

Extensive production methods currently predominate in low- and middle-income regions. Animal-sourced food consumption is also likely to grow as these countries develop economically (Alexandratos and Bruinsma 2012), with rising population and rising income-related dietary shifts. However, there is an ongoing shift toward intensification in these countries, both as an economic response to rising meat and dairy production and government agendas to develop economies through ‘green revolution’ type programs of achieving efficiency through intensification.

This shift toward intensification is expected to offset a substantial fraction of GHG emissions. In the greater East and Southeast Asia (ESA) region, an area of both rapid consumption growth and industrialization, these two trends are expected to approximately counter-balance one another, with 39% more meat consumption in 2050 relative to present day, but little net change in total emissions (FAO 2018a), credited to the efficiency of intensification.

However, the hypothesis that intensification leads to GHG emissions mitigation, in particular that this mitigation is large enough to offset projected increases in production and consumption, has not been corroborated through independent assessments. The following review describes the model-based methods in which GHGs from animal-sourced food production are typically assessed and forecasted. We then contrast this approach with recent atmospheric measurement-based methods, along with a discussion of both methods’ inherent uncertainties. Lastly, we conclude with a test case analyzing commonly-accepted emissions projections, adjusting the projection from ESA intensive systems by the same degree to which present-day emissions from North American intensive systems are possibly biased. These prospective results imply that emissions may increase in the region through 2050, contrary to existing predictions that they will plateau or decrease after 2030. This case study has important implications for the standard paradigm of agricultural development in low- and middle-income countries, as it implies that the GHG mitigation benefits of intensification may be overestimated.



## 2. Quantifying progress toward sustainable emissions

### 2.1. Bottom-up estimates of animal emissions

Present-day estimates of total GHG emissions in a given region are rarely measured directly. Rather, emissions estimates are based on animal emission models. Such models collect and integrate detailed information from multiple animals and production regions through counting, interviewing, and census-taking. Emissions models are referred to as ‘bottom-up’ estimates, because detailed information is collected at the ground-level, then multiplied and added up to calculate total emissions.

The predominant GHG from animal agriculture is methane. Methane is produced by enteric fermentation and manure management. Methane represents 50% of all animal emissions (direct + indirect) and 92% of direct animal emissions (FAO 2017).

Bottom-up estimates of methane are created by tallying up populations of each type of animal, known as activity data. These data are then multiplied by emissions factors—estimates of gas emitted by each animal daily or each operation annually. Separate emission factors are generally used for (a) enteric fermentation (belches) and the (b) handling and storage of their manure. Animal populations are subsequently multiplied by animal- and region-specific emissions factors, yielding an estimate of direct emissions produced. Bottom-up models are often also used to estimate indirect emissions from feed production and manure application to fields. However, this review mostly concerns direct emissions of animal agriculture, with some exceptions.

Emissions factors are calculated using detailed equations, provided in the IPCC guidelines of the

Fourth Global Assessment Report (Dong *et al* 2006) and its 2019 refinement report (Gavrilova *et al* 2019). Multiple levels of detail or ‘tiers’ of estimates are provided, corresponding to different levels of aggregation and coarseness. Emissions factors are calculated at Tier 1 with fixed tabular values, Tier 2 with the more detailed equations that are standardized by the IPCC, or Tier 3 which are considered rigorous and detailed methods that are country-specific but nonetheless subjected to international peer review (Dong *et al* 2006). Each level reflects processed-based detailed input information on how animals are raised in each region, differing mainly in what level of detail is included by the modeler. The resulting emissions factors describe how much methane and nitrous oxide an average animal from each herd directly emits per day or year.

The development of these emissions factors requires information on how animals are raised in each region, including feed, physical activity, manure disposal, and local climate, along with their productivity, such as weight gain, lactation amount, and breeding rate (figure 1). Most of these local production and environmental characteristics are required inputs for all tiers of IPCC methods—including Tier 1 methods that rely on IPCC emissions factors and Tier 2 and 3 methods that provide detailed models that allow users to calculate their own emission factors. The remainder of this discussion will focus predominantly on Tier 2 emissions to provide a standard, if simplified framework for understanding, but there are commonalities among all three tiers.

The details used to develop methane emission factors can be incredibly fine-grained and include information on how much an average animal moves during a day, whether or not a dried crust is allowed

to form on top of the slurry tanks that handle liquid manure, the temperatures of a given animal operation and the seasonal cycle of weather conditions, and other highly specific details. This detailed information can be challenging to collect, even for countries with detailed agricultural censuses (Hristov *et al* 2017).

Detailed input information in bottom up emissions estimates contributes to their precision—the degree to which estimates can detect changes between one system or another. However, this detail does not guarantee estimates that are accurate—close to the true, real-world value. As more details are added and multiplied within the models, the model errors tend to increase. The errors in totaled emissions estimates are therefore considerably more uncertain than any single input. Ultimately, bottom-up estimates require simplifying assumptions and extrapolation from representative systems. They therefore may contain large uncertainties and compounding errors (Miller *et al* 2014, Miller and Michalak 2017) that make emissions difficult to accurately quantify, including estimates contained in national inventory reports.

The uncertainty envelopes in bottom-up emissions inventories may be underestimated, and are difficult to compare and scrutinize in existing sources. Uncertainties are required in the reporting from all UNFCCC annex-1 countries, including the United States. However, uncertainty ranges are not easily discoverable, are missing from a number of UNFCCC reports and web portals, and do not clearly identify which types of the multiple types of model uncertainty they consider. The United States' inventory includes mention of uncertainty analysis performed on 185 input variables and categories, and reports resulting errors for enteric fermentation methane of approximately  $\pm 15\%$  (US EPA O 2021). The details of the assessment are vague, stating that uncertainties for input variables 'were collected from published documents and other public sources; others were based on expert opinion and best estimates'. Conversely, the IPCC methodology states that Tier 1 and 2 uncertainty estimates for enteric fermentation may be on the order of 50% (Dong *et al* 2006), a substantially larger range than the US EPA and most other bottom up emissions inventories report. The 2019 update states that uncertainty estimates for enteric fermentation contains 'no refinement' (Gavrilova *et al* 2019). Uncertainty ranges in bottom-up estimates are possibly too small, even though they are intended to be fully comprehensive.

Furthermore, bottom-up inventories uncertainties often do not reflect uncertainties arriving from model structure and parameter selection, referred to as epistemic uncertainty (Sykes *et al* 2019), instead reflecting uncertainties arriving from input data. Bottom-up models structurally contain dozens of parameters that describe how inputs and outputs relate to one another. For instance, one

commonly-used equation in Tier 1 and 2 methods for estimating the food consumption of cows uses multiple constant coefficients to relate live dairy cow body weight to quantity of dry matter ingested per day (equations (10.18) in Dong *et al* 2006, Gavrilova *et al* 2019). These and other parameters are gleaned from small-scale studies, dating back to the early 1980s and 1990s (NRC 1989, 2001, references therein). Analyses of epistemic uncertainties studies have been done at the farm level (Sykes *et al* 2019) but are not explicitly utilized in the development of national inventories.

While these uncertainties have been detected across models may be common to all bottom-up estimates as they rely on similar assumptions and model parameters, introducing systematic biases that elude uncertainty analysis. The multiple sources of uncertainties in bottom-up estimates (input data, geographic distribution and allocation, model and parameter selection) are rarely analyzed holistically by existing studies. A recent analysis discovered that, while three different total US inventories of methane agree within 20%, their spatial discrepancies exceed 100% across certain US states (Hristov *et al* 2017). Altogether, the information used to develop bottom-up models, including input data and model parameters, may therefore lead to biases in reported estimates outcomes, due to their outdated nature, or due to a lack of representative information from the wide national and global variation in contemporary production systems (Wolf *et al* 2017).

In summary, bottom-up methods allow researchers to estimate animal emissions with a high level of precision, or fine-grained resolution, parsing out how GHG emissions differ between animal species, geographic regions, production systems, and food products, but not necessarily an accordingly high level of accuracy. The accuracy of these methods in different global regions remains poorly characterized, requiring independent methods to verify emissions estimates from animal agriculture.

## 2.2. Top-down estimates: evaluating animal emissions from the atmosphere

The GHG emissions estimates produced by bottom-up models can be compared against measurements taken from the air above animal operations, using sensors on tall towers, airplanes, and satellites.

Researchers use meteorological models and observations of wind direction, speed, and turbulence to link the measured GHG concentrations in the air back to their sources on the ground. These methods can be used to gauge the quantity of emissions from sources in the upwind footprint of the measurements over months or years—by measuring for prolonged periods of time and from multiple wind directions. Atmospheric estimates start by measuring GHGs in the sky, then trace those GHGs back to their source regions on the ground; these methods are referred to as 'top-down' estimates.

One limitation of top-down estimates is that these methods can only broadly distinguish among emission sources. The partitioning of sources typically depends upon differences in the spatial distribution of different source types, requiring accurate information regarding where these sources are distributed in geospatially gridded bottom-up data (e.g. Miller and Michalak 2017). For example, a top-down approach would struggle with this source partitioning if cattle are grazed amidst natural gas wells, or pig manure lagoons and natural wetlands are located in the same region.

The level of disaggregation or detail in emissions inventories also determines how finely top-down emissions estimates can be used to distinguish among source types. For example, top-down studies often employ a 'scaling factor inversion' to statistically partition sources—by scaling each emissions sector in a bottom-up inventory to match atmospheric observations. The number of these scaling factors is limited by the number of emissions categories present in the bottom-up gridded inventory and by the information content of the atmospheric observations used in the inverse model. The geospatially gridded inventories of animal emissions from EDGAR or the US EPA that are publicly available and used in the top-down analyses reviewed here are not disaggregated by type of animal or production type. Such disaggregated gridded bottom-up products would be useful for examining potential errors and biases, but could introduce further partitioning errors as the number of scaling factor coefficients would increase.

Another limitation is that top-down atmospheric estimates rely on wind observations and models with their own errors. Importantly, these atmospheric transport errors are different from the errors in bottom-up models (Pandey *et al* 2019). These errors are commonly reported in the literature and are non-compounding.

An additional challenge of top-down modeling is that these models are often under-constrained or under-determined by atmospheric observations. The number of atmospheric observation sites may be limited, while the spatial and temporal patterns of emissions can show complexity that exceeds the detection abilities of existing atmospheric observations. As a result, multiple configurations of the emissions or multiple solutions to a top-down estimation problem can reproduce the atmospheric observations within a specified error tolerance. Existing top-down models therefore have finite utility in constraining emissions at fine spatial scales. With that said, the uncertainties in top-down emissions estimate typically decrease at regional and national spatial scales (e.g. Miller *et al* 2013, 2019). The reported uncertainties at these scales are often smaller than the differences between bottom-up and top-down estimates, implying that top-down methane estimates are sufficiently robust to compare and contrast with bottom-up models at

regional to continental scales (e.g. Turner *et al* 2015, Saunois *et al* 2020).

Top-down atmospheric estimates are therefore an independent way of testing the accuracy of the estimates produced by bottom-up emissions models across both small and large spatial scales, albeit with their own inherent uncertainties and limitations. To date, however, they cannot reliably distinguish between various sources that are co-located.

### 3. Comparing bottom-up and top-down estimates of US animal methane emissions

#### 3.1. Quantifying differences between emissions estimates

We synthesized reports in the literature regarding top-down methane emissions estimates in regions of predominantly intensive production. We conducted literature searches on ISI Web of Science and Google Scholar for terms pertaining to top-down atmospheric estimates of methane, and filtered search results by spatial domain (state/province or larger) and whether they statistically partitioned livestock sector emissions from the total estimate. We present the results in table 1, alongside comparisons with bottom-up estimates.

Estimates from atmospheric measurements indicate that global animal methane emissions may be only slightly higher than the bottom-up models predict—about 5% higher—and within the margin of error (e.g. Turner *et al* 2015). A more recent comparison of methane estimates from total agricultural and waste sources from 2000 to 2017, which included all direct livestock emissions but also include rice crops and landfills, also found that bottom-up and top-down emissions estimates agree well within statistical uncertainty (Saunois *et al* 2020).

The difference is much greater in the United States; at least four top-down estimates of the contiguous US, representative of emissions occurring over at least one full year, indicate that direct animal methane emissions are 39%–90% higher than bottom-up models predict (Miller *et al* 2013, Wecht *et al* 2014, Turner *et al* 2015). Such comparisons are currently lacking for other predominantly intensified production regions, e.g. Europe.

This tendency is mirrored in smaller geographic regions within North America. Of five regional top-down studies, four found significantly higher methane emissions from areas of confined animal production than the bottom-up models suggest (Jeong *et al* 2016, Chen *et al* 2018, Desjardins *et al* 2018, Yu *et al* 2021). The exception is the Southeastern US, where there was close agreement between aircraft measurements and a short-term campaign (Sheng *et al* 2018). This and the Ontario Desjardins *et al* (2018) study were both conducted over an aircraft campaign lasting less than 2 months, meaning that these results may not be representative of the full annual

**Table 1.** Direct methane emissions from farmed animals in Tg CH<sub>4</sub>. Scale factor refers to the relative magnitude of the top-down emissions estimate relative to the respective bottom-up estimate (e.g. 200% means top-down estimate is two times higher than bottom-up estimate). Emissions model abbreviations are EDGAR—Emissions Database for Global Atmospheric Research; EPA—Environmental Protection Agency; CALGEM—California GHG emissions measurement project. Bottom-up models reflect the models used to develop geospatially gridded datasets for comparison in the respective top-down analyses.

Region	Year	Bottom-up model	Bottom-up estimate	Top-down estimate	Scale factor	Reference
Globe	2009–2011	EDGAR	111	116 ± 10	105%	Turner <i>et al</i> (2015)
USA			8.9	14.8 ± 2.2	166%	
USA	2004	US EPA EDGAR	8.8 8.5	12.2 ± 1.3	139% 144%	Wecht <i>et al</i> (2014)
USA	2007–2008	US EPA EDGAR	9.3 8.9	17.0 ± 6.7	181% 190%	Miller <i>et al</i> (2013)
US Midwest	2016–2017	US EPA EDGAR	2.6 2.7	4.8 ± 1.5	185% 178%	Chen <i>et al</i> (2018)
US Upper Midwest	2017–2018	US EPA	3.3	4.11 ± 0.54	124%	Yu <i>et al</i> (2021)
California, USA	2013–2014	CALGEM	0.90	1.33 ± 0.40	149%	Jeong <i>et al</i> (2016)
E. Ontario, Canada	April–May 2011	IPCC Tier 2	0.013 ± 0.002	0.027 ± 0.006	217%	Desjardins <i>et al</i> (2018)
US Southeast	August–September 2013	US EPA	3 ± 0.5	2.9 ± 0.3	97%	Sheng <i>et al</i> (2018)

cycle. The Southeastern US is also an area of abundant wetland emissions (8–16 Tg CH<sub>4</sub> yr<sup>-1</sup>), with close proximity to animal production areas, making accurate source partitioning challenging (Miller *et al* 2015).

The bottom-up models included in these sources use agricultural survey and census data on farmed animal populations, and use IPCC standards (predominantly Tier 2) to calculate their emissions factors. The bottom-up estimates (table 1, column 4) correspond to various base years and bottom-up models (table 1, columns 2 and 3 respectively) and model versions. Future work is therefore needed to compare bottom-up and top-down emissions estimates using the most recent bottom-up model versions across their respective regions and base years. While the models differ in assumptions made to calculate emissions factors, i.e. assumptions reflecting how animals are fed, housed, and raised, feeds tend to be energy-rich and animal production methods are intensive and confined throughout the US and Canada.

The discrepancies between bottom-up and top-down estimates in the US and Canada imply that models may under-predict the emissions intensity of animals raised in intensive, predominantly confined systems. This pattern furthermore suggests that a greater proportion of total global methane emissions from animals is coming from intensive systems than is routinely reported.

### 3.2. Why North American emissions may be underestimated by models

It is unclear why the modeled methane emissions from farmed animals in the US are often lower than emissions inferred from atmospheric measurements. Discrepancies do not seem specific to one animal or system. For example, Jeong *et al* (2016) analyzed airborne measurements from California and were able to partition emissions between dairy cattle and other animals. They found dairy cattle and other non-dairy animals' emissions were 45% and 69% higher, respectively, than the bottom-up model predicted. Therefore, emissions from multiple animal species may be underestimated by the bottom-up emissions models.

Bottom-up models also appear to routinely underpredict emissions from manure. Models of manure emissions are based on laboratory experiments within controlled test chambers. When methane is measured outside of the lab, in the air directly above manure tanks, pits, and piles, emissions tend to be greater than models predict, sometimes by more than 300% (Owen and Silver 2015). However, because manure represents approximately 11% of animal methane emissions, this difference is still not large enough to explain the gap in US animal methane emissions (Wolf *et al* 2017).

High methane emissions may also stem from animal disease. Infections in animals are widespread in intensive production systems; approximately one

quarter of US dairy cows have mastitis—a swelling and infection of udders accompanied by secretion of somatic cells into the milk (USDA 2016). Among beef cattle, nearly all feedlots have at least some cattle affected by bovine respiratory disease complex (BRDC or ‘shipping fever’) or lameness; BRDC has been detected in 16%, digestive disorders in 4%, and lameness in 2% of the entire US feedlot beef cattle population (USDA 2013). Digestive disorders and infectious parasites have multiple adverse effects on enteric fermentation emissions; they (a) decrease digestion efficiency, hence increasing overall feed requirements and methane output per unit of production (Houdijk *et al* 2017, Fox *et al* 2018) and (b) increase methane yield per unit of feed. The former tendency may be missed by bottom-up models if entirely healthy cattle are being taken as representative of the entire herd. The latter tendency cannot be represented at all by Tier 2 emissions methods models (Dong *et al* 2006, Gavrilova *et al* 2019), and would require new methods that could account for unhealthy/infected proportions of the herd. A full accounting for animal diseases, which are prevalent in modernized intensive systems, may result in bottom-up emissions that are higher than commonly reported.

The current body of top-down estimates is insufficient to understand precisely why bottom-up models may underpredict animal emissions. The metabolic processes that generate emissions (e.g. manure and belches) are highly co-located relative to the large and distant spatial scales at which atmospheric measurements for top-down estimates occur. Multiple inputs and metabolic processes contribute to the rate of methane conversion in an individual animal or manure storage system. In the future, a greater number of atmospheric measurements performed at smaller spatial scales, located in closer proximity to facilities where manure storage is separated animals enteric fermentation, could be utilized to better partition which types of animals, processes, and production systems are most responsible for driving the discrepancies.

## 4. Animal emissions in an international development context

### 4.1. Case study: intensification in East and Southeast Asia

Intensification in ESA, which includes mainland China, is expected to reduce marginal emissions substantially (Wang *et al* 2017, Yu *et al* 2018, FAO 2018a). A recent analysis of animal emissions in China recommends increasing per-capita production of animal products (Yu *et al* 2018), finding that improved efficiency will offset the increase in production and provide net methane emission reductions overall. The benefits of intensification, however, vary case by case, and by the proportions of direct and indirect

GHG emissions, which can vary widely geographically (Poore and Nemecek 2018).

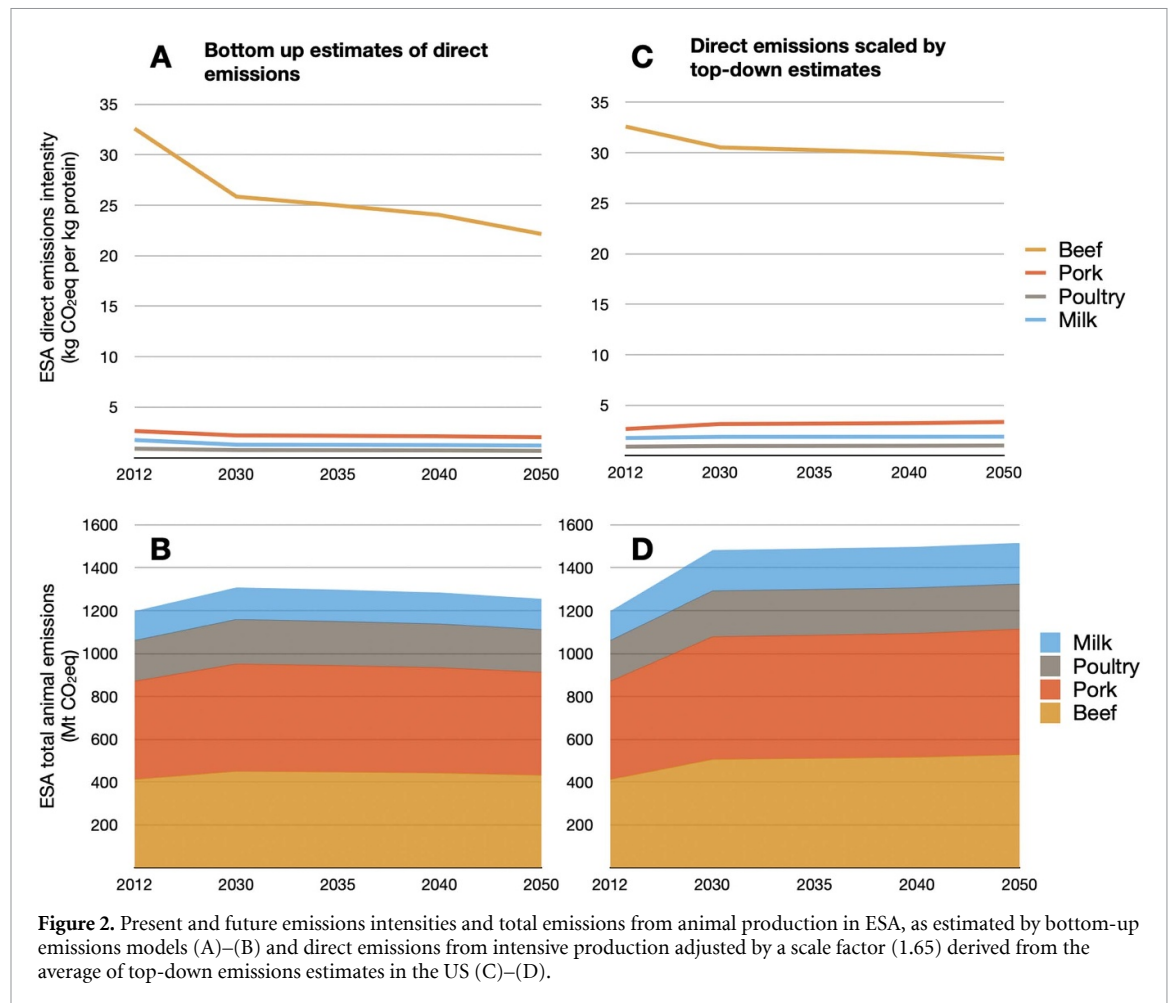
The following synthesis aims to approximately correct or scale future GHG emissions projections in the ESA region for direct animal emissions. We adjust the proportion of direct emissions in bottom-up emissions estimates by a mean and range of scaling factors derived from the USA top-down emissions scaling factors. This analysis serves as a prospective correction, outlining a possible consequence of business-as-usual development, but it is not a definitive prediction. The specific metabolic processes and source categories that can fully explain the discrepancy in North American emissions (table 1) are still unknown, including but not limited to manure methane fluxes influenced by climate and geography (Dong *et al* 2006). However, we expect that this analysis will approximately account for changes in the increased use of landless, intensive systems in the region, with common industrial production characteristics that are increasingly deployed in the ESA region and throughout the developing world (Wasley and Heal 2020).

The 2050 projections were taken from FAO, using regionally appropriate emissions factors from IPCC (FAO 2018a). The data are available on the online FAO data portal ‘Food and Agriculture Projections through 2050’ ([www.fao.org/global-perspectives-studies/food-agriculture-projections-to-2050/en/](http://www.fao.org/global-perspectives-studies/food-agriculture-projections-to-2050/en/)). We used agricultural emissions and livestock herd size data from the ‘business-as-usual’ projection (FAO 2018a) and disaggregated further where necessary by supplementing this data with public data from FAO GLEAM, which further partitions emissions into production categories GHGs, and lifecycle processes (FAO 2017).

Total demand for animal-sourced foods is increasing in ESA concurrently with intensification. Emissions of each unit of production are assumed to decrease in the near future according to prospective bottom-up estimates (figure 2(A)). As a result of these two counterbalancing forces, animal emissions are expected to increase by 2030, then gradually go down until they return to early 2000s levels by 2050 (figure 2(B)). However, these bottom-up emission projections may overestimate the benefit of intensification.

If the share of direct GHG emissions projected to come from intensive systems is scaled in proportion with top-down US estimates (scaled up by 65%, the average inferred from top-down estimates, provided in table 1), future emissions per unit of meat and dairy will not decrease as much as FAO bottom-up estimates predict (figure 2(C)). Multiplying this emission intensity by total consumption, we find that the ESA region could reach 1.52 (1.41–1.62 total range) GtCO<sub>2</sub>eq by 2050, an amount 21% (13%–29%) higher than previously predicted (figure 2(D)).





Indirect emissions account for approximately 40%–50% of animal agriculture’s emissions. This prospective analysis leaves indirect emissions unaltered (e.g. nitrous oxide from fertilizer applied to animal feeds and CO<sub>2</sub> emissions from land use change) while adjusting direct CH<sub>4</sub> emissions and also N<sub>2</sub>O emissions from manure management; N<sub>2</sub>O emissions appear to be similarly or more dramatically higher than bottom-up estimates when assessed using top-down methods (Owen and Silver 2015). Estimates of CO<sub>2</sub> from fossil fuel and land use change, which are indirect emissions processes in animal agricultural production, generally show good agreement among bottom-up and top-down assessments (Friedlingstein *et al* 2020). However, the implications for animal agriculture in developing contexts is highly uncertain, due to large geographic and seasonal variation in the discrepancies between emissions patterns and because patterns of feed crop trade through 2050 are uncertain. These complexities would make assessing the effects of indirect emissions estimates adjustments substantially more speculative, hence an assessment of indirect emissions estimate corrections was outside of our scope. By not altering indirect emissions, we produce a conservative estimate.

This case study is limited in terms of the confidence with which we can project and scale such

differences, however. Uncertainties are not reported by FAO in their ESA regional emissions through 2050, which is a common tendency in bottom-up emissions estimates and projections. We discuss measures needed to improve uncertainty analysis in section 2.1. A fuller analysis regarding how uncertainties in this and other bottom-up emissions compound is outside of the scope of this review and prospective analysis, but is urgently needed.

We express emissions in our ESA case study in CO<sub>2</sub> equivalence (CO<sub>2</sub>e) using GWP<sub>100</sub>, although we acknowledge that this is limited in by placing short- and long-lived gases on the same scale. Recent analyses and commentary have proposed a new approach of GWP\*, which includes information about the rate of change in shorter-lived pollutants like methane (Collins *et al* 2020, Lynch *et al* 2020). We share the concerns motivating the development of an alternative CO<sub>2</sub> equivalence metric. However, we did not adopt this new method for recalculating CO<sub>2</sub>e due to (a) maintaining consistency for a comparison with previous FAO summary findings and (b) a lack of consensus regarding the appropriateness of assumptions embedded in GWP\*, still inspiring active debate on topic (e.g. Rogelj and Schlessner 2019, Smith and Balmford 2020). Ultimately, there are limitations to any method of calculating an equivalence of all

climate pollutants to just one chemical species, rather than directly calculating their effects on temperature. However, this is only a limitation of our ESA case study and not the discrepancies found in our review of methane emissions themselves (table 1), in which we compare Tg of CH<sub>4</sub> directly.

Our prospective analysis demonstrates that animal emissions in the ESA region may not plateau in by 2030, as has previously been projected. While the majority of emissions increase is expected to occur by 2030 (figure 2(B)), this increase may be higher than expected (figure 2(D)) and total emissions may still slightly increase for multiple decades as consumer demand in the region continues to rise.

Despite this uncertainty, our illustrative example of ESA demonstrates that widescale adaptation of intensive production methods in low- and middle-income countries may have limited benefits to GHG emissions, especially if demand continues to grow.

#### 4.2. The limits of intensification for sustainable development

Recommendations to intensify animal production are ubiquitous in the scientific literature (Thornton and Herrero 2010, Gerber *et al* 2013, Swain *et al* 2018, Grossi *et al* 2019). Such suggestions shape policies that disseminate intensive production technologies to low- and middle-income countries. Such policies include sustainable lending criteria from international development banks (World Bank International Finance Corporation 2012), and government subsidies and incentives, which in turn influence and are influenced by the actions of multinational meat and dairy producers (Lazarus *et al* 2021).

Policies that incentivize landless and industrialized animal production methods risk amplifying other public health and environmental harms. Low-interest loans for such animal production operations cover the capital costs of building large shed-like facilities that confine tens to hundreds of thousands of animals indoors (Wasley and Heal 2020) in a manner that often highly restricts their movement and expressions of natural behavior (Prunier *et al* 2010, Grandin 2016, Widowski *et al* 2016). Such facilities require high levels of subtherapeutic antibiotics (Silbergeld *et al* 2008), leading to antibiotic resistance, and encourage the rapid transmission and mutation of viruses with pandemic potential (Alders *et al* 2014, Millman *et al* 2017, Lycett *et al* 2019) including highly pathogenic avian influenzas (HPAI, also known as ‘bird flu’). Intensified animal production poses a litany of zoonotic disease risks (UNEP and ILRI 2020). Large concentrated quantities of production and storage also create point and non-point sources of groundwater pollution and can degrade air quality (Secchi and McDonald 2019, Domingo *et al* 2021).

While the efficiency of feeding and fattening allowed by intensified animal production can reduce

indirect land use, confined forms of animal production including landless and industrialized systems Intensification can lead to further homogeneity of food systems and exacerbates zoonotic disease risks (UNEP and ILRI 2020) including interspecies transmissions of and other infectious viruses.

Public and private institutions must fully evaluate socio-environmental externalities and risks of intensified animal production systems in their policies that finance and incentivize their expansion. Continuing to export these technologies and practices to developing regions could achieve less GHG mitigation than commonly assumed while exacerbating other socio-environmental harms.

#### 4.3. The climate mitigation potential of agricultural development

The climate impacts of growing demand for meat and dairy have implications for keeping global warming within safe limits. Most countries have not committed to reducing non-CO<sub>2</sub> gases (Climate Action Tracker, Wang *et al* 2018), including methane and nitrous oxide from animals. Previous analysis has demonstrated that if countries aggressively limit CO<sub>2</sub> from fossil fuels, growth in meat and dairy consumption would be sufficient to cause warming of above 2 °C even with aggressive mitigation in other sectors (Harwatt *et al* 2020) and business-as-usual growth in animal sourced food production, efficiency, and consumption would exceed the entire GHG budget compatible with a 1.5 °C threshold of warming (Clark *et al* 2020). The aforementioned findings were all informed by bottom-up models, which may underestimate emissions from intensive production systems that are proliferating globally. Growth in meat and dairy production and consumption therefore may be even more incompatible with limiting global warming than previously reported.

Our prospective analysis demonstrates that GHG emissions could increase as developing regions intensify production. On one hand, intensive systems could achieve GHG mitigation with improved technologies such as anaerobic manure digestors that capture biomethane for fuel usage (Paolini *et al* 2018) and concentrated feeds supplemented with enteric fermentation suppressants such as 3-nitrooxypropanol and seaweeds (Machado *et al* 2016, Jayanegara *et al* 2018). On the other hand, ambitious interventions and improvements in efficiencies have the potential to mitigate less than a quarter of annual agricultural emissions relative to business-as-usual (Höglund-Isaksson *et al* 2020).

GHG emissions are not the only indicator of sustainability, and GHG suppression measures for intensively-raised animals could incentivize animal concentration at larger scales, exacerbating co-impacts such as water pollution load and antimicrobial resistance, especially if such technologies require additional subsidies to remain economically

viable (Eker *et al* 2019). Sustainability encompasses multiple dimensions, including multiple environmental impacts, social equity, and economic feasibility. Effective sustainable development strategies for food systems are uncommon because they must provide benefits across multiple dimensions and goals (Rasmussen *et al* 2018).

Socio-environmental benefits could be realized by providing resources in areas of pastoral production for intensification practices that do not involve confinement to similar levels of intensive US 'landless' systems. Such practices include improved animal veterinary care, improved forage seeds and pasture management, and more precise nutrient application. Improved coordination between smallholder and midsize producers, government, and NGOs can hasten these benefits (Swain *et al* 2018, FAO 2018b, IPCC 2019). Intensification may be effective in optimizing co-benefits in some circumstances, especially in regions where indirect emissions are large (e.g. ongoing land-use change, including deforestation).

Reducing externalities of further intensification also depends upon limiting overall consumer demand, particularly in regions where animal-sourced food consumption already exceeds nutritional requirements. Improved productivity and lower prices can lead to runaway effects of stimulating greater demand a feedback process known as Jevons Paradox (Garnett *et al* 2015), and stimulate more production by decreasing input costs (Kreidenweis *et al* 2018). Feedback effects between supply and demand can therefore undermine or worsen the environmental impacts that intensification efforts seek to mitigate. Mitigation policies should address the overconsumption of animal-sourced foods to avoid these feedback effects, rather than merely focusing on improving production.

Managing both supply and demand of animal-sourced foods must be considered within efforts to reform and develop food systems sustainably while mitigating climate change (IPCC 2019, Clark *et al* 2020, Harwatt *et al* 2020). Such efforts could include reforming dietary recommendations to reflect regional environmental impacts and planetary boundaries (Willett *et al* 2019, Springmann *et al* 2020), developing public-private partnerships to stimulate production and demand of plant-based proteins (Attwood *et al* 2019), and changing taxation and subsidy structures to reflect the true environmental costs of meat and dairy production across multiple impacts (IPCC 2019).

In addition to limiting overall consumption, more steps are necessary to ensure that intensification strategies maximize co-benefits and constrain potential externalities. These include (a) marrying intensification with strong environmental conservation policies (Garrett *et al* 2018, Kreidenweis *et al* 2018) (b) improving farmer livelihoods and disincentivizing

industry consolidation that can crowd out smallholder producers, and (c) avoiding extreme forms of confinement entailed in many landless and industrialized systems, which also promote zoonotic disease emergence and antimicrobial resistance (UNEP and ILRI 2020). This last criterion may prove an exceedingly difficult needle to thread if demand for animal-sourced foods continues to increase unabated. All of the above and more is required to ensure that the intensification of animal agriculture does not further amplify multiple environmental risks including climate change, biodiversity loss, and the emergence of potentially pandemic zoonotic diseases.

## Data availability statement

The data that support the findings of this study are available upon reasonable request from the authors.

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