Environmental impact of insect rearing

Insects as animal feed: novel ingredients for use in pet, aquaculture and livestock diets

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7 Environmental Impact of Insect Rearing

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7.1 Introduction

One of the first publications on using insects as feed came out approximately a century ago (Lindner, 1919). Scarcity of fats at the end of the First World War caused interest in alternative sources. In the aforementioned publication several species that are currently still investigated received attention, including yellow mealworms (YM; *Tenebrio molitor*) and common house fly (HF; *Musca domestica*). The primary idea was to convert underused resources towards useful compounds via insects. Whereas a shortage of fat was the concern at that time, nowadays protein scarcity drives the more recent investigations towards using insects as bio-converters and protein concentrators. Commercially-farmed insects have been perceived as sustainable alternatives to conventional animal products for quite some time (Oonincx, 2015). However, studies aiming to quantify their environmental impact have only become available during the last decade. This chapter provides a general overview of the currently available studies.

After this introductory paragraph, first, direct greenhouse gas (GHG) emissions from insects used as feed or food are discussed. Subsequently, data from life cycle assessments (LCAs) on commercially farmed insects are discussed per species. This is followed by a paragraph on the relevance of the utilized feed on the environmental impact of insects and their derived products, including suggestions to lower this impact. Then, the limitations in the available data are highlighted followed by a concluding paragraph, where the most relevant conclusions of this chapter are summarized.

7.2 GHG Emissions

GHG emissions, as a driver of climate change, have received ample attention in recent years. There are various GHGs differing in their potency (global warming potential) which is expressed as CO_2 equivalents, with CO_2 being used as the benchmark. Two of the other well-known GHGs are methane and nitrous oxide which are produced by bacteria and considered to be ~21 and ~310 times, respectively, more potent than CO_2 (Krey *et al.*, 2014; Buendia *et al.*, 2019).

Several insect groups, including cockroaches, termites and certain beetles have methane-producing bacteria in their gut (Hackstein and Stumm, 1994). The amount of methane produced in these species varies: sun beetle (*Pachnoda marginata*) larvae emit 4.9 g of methane/kg gain, whereas this is 1.4 g/kg for the Argentinean cockroach (*Blaptica dubia*) and only 0.1 g/kg for larvae of YM (Oonincx *et al.*, 2010). While methane production is environmentally disadvantageous, these bacteria (e.g. *Blattabacterium* sp.) can facilitate highly efficient use of dietary protein by converting uric acid to amino acids (Sabree *et al.*, 2009; Oonincx *et al.*, 2015).

Nitrous oxide emissions seem primarily associated with bacterial digestion of dead insects, rather than insect rearing per se (Oonincx *et al.*, 2010). Indeed, nitrous oxide emissions reported for house crickets in the latter study were associated with high cricket mortality, whereas a Thai study without high mortality deemed the nitrous oxide emissions negligible (Halloran *et al.*, 2017). In moist rearing substrates, such as those used for black soldier flies (BSF; *Hermetia illucens*) and HF, bacteria and fungi

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present in the substrate can contribute to direct GHG emissions. These contributions can be large: 34% of the emitted CO_2 in a BSF system and up to 92.5% for sun beetle larvae (Oonincx *et al.*, 2010; Parodi *et al.*, 2020). The latter value is probably an overestimate due to fungal development in substrate without larvae, whereas larval presence greatly reduces fungal development. Nitrous oxide emissions might well be associated with the feeding of decaying materials, rather than being associated with insect metabolism (Mertenat *et al.*, 2019). Even though only a few insect species have been studied, the methane and nitrous oxide emissions of farmed insects generally seem far lower than for conventional livestock (Oonincx *et al.*, 2010; Parodi *et al.*, 2020). The contribution of direct GHG emissions to the total emissions of insect farming systems is low when considered for the YM (0.3%), but higher for BSF (10–15%) (Oonincx and de Boer, 2012; Mertenat *et al.*, 2019; Parodi *et al.*, 2020).

7.3 Life Cycle Assessment (LCA) Methodology

LCA is a method that enables the comparison of environmental, social or economic impacts of similar products or services. It starts with defining the system border: the boundary of a production system (Oonincx and de Boer, 2012). This can include all system inputs and end at the farm gate ('cradle to farm gate'). Alternatively, some inputs can be excluded, or the system can be extended to include post-farm processing.

Then, the indicators to be quantified are chosen. In environmental LCAs this can include GHG emissions, which fall under the indicator global warming potential (GWP). Other commonly used indicators are energy use (EU) as a measure of fossil fuel depletion, and land use to quantify the amount of arable land used in the production chain. Associated land use changes (LUC) can lead to deforestation, and thereby to GHG emissions, which are often mentioned separately. Other commonly used indicators are water depletion, effects on fresh and marine waters, acidification, and many more (Halloran *et al.*, 2017).

The environmental impact is subsequently coupled to a functional unit (FU), a quantitative measure indicating the function of a product and this is often expressed based on weight. For insects, commonly used FUs are kilograms of fresh weight, kilograms of protein, or kilograms of edible protein (Oonincx and de Boer, 2012). The environmental impact arising at different steps within the system border is then summed up, resulting in the total impact for a certain environmental indicator. Lastly, this is divided by the number of FUs and gives the environmental impact per FU. This value can then be used to compare the environmental impact between products with a similar function.

Production systems sometimes yield more than one product. For instance, when producing beer, the spent grains are considered a by-product. In such cases part of the environmental impact is allocated to the by-product based on relative weight, or on relative economic value.

At the time of writing the following insect species have been evaluated in an environmental LCA:

- YM (Oonincx and de Boer, 2012; Miglietta *et al.*, 2015; Thévenot *et al.*, 2018)
- superworms (SW; *Zophobas morio*) (Oonincx and de Boer, 2012);
- house crickets (HC; *Acheta domesticus*) (Halloran *et al.*, 2017);
- banded crickets (BC; *Gryllodes sigillatus*) (Suckling *et al.*, 2020);
- two-spotted crickets (*Gryllus bimaculatus*) (Halloran *et al.*, 2017; Suckling *et al.*, 2020);
- BSF (Salomone *et al.*, 2017; Mertenat *et al.*, 2019; Smetana *et al.*, 2019); and
- HF (van Zanten *et al.*, 2015).

Mealworm LCAs

The YM, together with the SW, were part of the first LCA on edible insects (Oonincx and de Boer, 2012). A farm, producing 83 t/year, was assessed from cradle to farm gate excluding buildings and equipment. Both fresh weight and weight of edible protein were used as FUs. Per kilogram of fresh weight, the GWP for these mealworms was 2.65 kg CO_2-eq., the EU amounted to 33.68 MJ and land use was quantified at 3.56 m^2. Feed production and transportation were the main drivers for GWP (56%), EU (43%) and land use (99%). These mealworms were considered an alternative form of animal protein for human consumption, and were therefore compared to milk, pork, chicken and beef. The EU per kilogram of edible protein was higher for mealworms than for milk and chicken, and similar to values published for pork and beef. Both the GWP and the land use per kilogram of edible protein were lower for the mealworms than for the four benchmarks. The relatively high EU was due to the use of fossil fuels for heating the farm to suitable ambient temperatures. The impact of these heating requirements was largely offset in the GWP by an efficient feed utilization, which concomitantly limited the required amount of arable land. In a subsequent study, the water use of the aforementioned mealworm production system was quantified (Miglietta *et al.*, 2015). When expressed as litres per gram of animal live weight, the water use for the mealworms was higher than for chicken and pigs, but lower than for beef cattle. However, when expressed as litres per gram of edible protein, the water use was lower for mealworms (23 l/g) than for chicken (34 l/g), pork (57 l/g) or beef (112 l/g). The difference in ranking between these two units is caused by a higher protein content and greater edible portion of mealworms (100%), compared to the chosen benchmarks.

A second assessment of YM was based on a farm producing 17 t of larvae/year and used a cradle-to-mill-gate system border, including impacts of feed, farm and equipment (Thévenot *et al.*, 2018). Environmental impact was economically allocated over insect meal (88.5%) and oil (11.5%) based on yields and sales prices and expressed per kilogram of larvae meal, and per kilogram of protein (FUs). Feed production was a major driver of environmental impact (land use 87%, eutrophication potential (82%), acidification potential (66%) and GWP (48%)). The farming process was associated with 29% of the EU and 19% of the GWP. Most of the energy (56%) was used for drying the larvae. The impact of mealworm production in this assessment was lower than for the first (EU 24.29 vs 33.68 MJ, climate change 0.99 vs 2.65 kg CO_2-eq. and land use 1.60 vs 3.56 m^2). These differences are likely to reflect differences in the energy source (nuclear vs natural gas) and feed composition (wheat bran vs mixed grains with carrots).

The environmental impact of mealworm meal was higher for all investigated parameters compared to the utilized benchmarks for soya bean meal and fishmeal.

Cricket LCAs

Data from a Thai company, producing HC and two-spotted crickets in approximately equal proportions, was assessed from cradle to farm gate, including building construction materials (Halloran *et al.*, 2017). Edible mass and edible protein were used as FUs and the frass (insect faeces, often mixed with undigested feed and moulds) was considered a by-product replacing mineral fertilizer. The cricket farm was compared to a local broiler producer based on edible protein. Crickets have a higher crude protein content than broiler meat, therefore this FU is beneficial for the cricket production system. The GWP, including LUC was higher for broiler meat than for crickets (8.21 vs 4.35 kg CO_2-eq./kg edible protein). The eutrophication potential (freshwater, marine water and terrestrial) was approximately twice as high for broiler meat compared to crickets. Also, water depletion was higher for broiler meat than for crickets (0.94 vs 0.71 m^3/kg edible protein), whereas resource depletion, including minerals, fossils fuels and renewables, were similar (0.041 vs 0.043 g Sb-eq./kg edible protein). Feed production was the major driver of environmental impact.

A recent assessment based on a UK company, producing two-spotted crickets and BC for the UK pet food market, used a similar system border (cradle to farm gate) and also considered the produced frass as a by-product replacing mineral fertilizer (Suckling *et al.*, 2020). Weight of the live crickets was used as the FU. Feed production and cricket rearing were the main drivers of environmental impact. The GWP was 21.1 kg CO_2-eq./kg of cricket, of which 59% was attributed to the cricket rearing process, and 19% was due to heating. The far lower GWP of the Thai cricket farm (2.57 kg CO_2-eq./kg of cricket) (Halloran *et al.*, 2017) is due to the colder climate and higher control over climate conditions in the UK farm requiring more energy, thereby increasing the GWP. Furthermore, the UK study assumed that all carbon contained in the frass was emitted as CO_2 and, together with CO_2 from cricket respiration, these emissions were included in the GWP. These were excluded in the Thai study. Most studies exclude direct CO_2 emissions of insects and their frass as this carbon was first taken up from the air and stored as plant biomass, subsequently used as feed (Oonincx *et al.*, 2010; Oonincx and de Boer, 2012). Hence it is not a net contribution (emission) to GWP.

Furthermore, the UK system had a higher water resource depletion for crickets than the Thai system (0.82 vs 0.42 m^3/kg cricket), of which 99% was due to feed production. Here, the difference is likely due to the high feed conversion ratio (FCR) in the UK system (9.09 vs 2.50) indicating a very poor feed utilization rate. This factor potentially also underlies the six times higher freshwater ecotoxicity and 12 times higher freshwater eutrophication value for the UK system compared to the Thai system. Improvement of the FCR from the current 9.09 to 1.47 (Lundy and Parrella, 2015) would decrease freshwater eutrophication by 44%, LUC by 66%, and water resource depletion by 82%, indicating the large potential improvement due to better feed utilization.

Fly LCAs

Two species of flies have been assessed, the BSF (Salomone *et al.*, 2017; Mertenat *et al.*, 2019) and HF (van Zanten *et al.*, 2015). Data from an Italian pilot facility processing organic food waste with BSF larvae was assessed based on cradle-to-farm-gate data, excluding machinery and equipment (Salomone *et al.*, 2017). Three FUs were used: (i) tons of processed food waste; (ii) fat to replace rapeseed for biodiesel; and (iii) protein to replace soya bean meal in aquafeed. Frass was considered as a by-product replacing inorganic fertilizer. The mass of the organic food waste was reduced by 67% and associated with a GWP of 30.2 kg CO_2-eq./t of food waste. When corrected for avoided soya bean meal and nitrogen fertilizer, the GWP was -432 kg CO_2-eq./t of food waste. The GWP per kilogram of protein was 2.1 kg CO_2-eq. This was primarily (57%) due to assumed direct GHG emissions, which were derived from mass-based emissions from the methane-producing sun beetle larvae (Oonincx *et al.*, 2010). When calculating based on mass-gain data for that species the GWP was lower – 1.1 kg CO_2-eq./kg of protein. Recalculating the emissions, based on direct GHG emissions for BSF (Parodi *et al.*, 2020), indicates that direct emissions are approximately 95% lower than for sun beetle larvae which leads to a GWP of 0.91 kg CO_2-eq./kg of protein. This discrepancy is likely caused by the fact that contrary to sun beetle larvae, BSF larvae do not produce methane via gut-associated bacteria (Mertenat *et al.*, 2019). Therefore, direct GHG emissions are far lower, and hence the contribution to GWP is primarily due to transportation and drying of the larvae. Compared to the reported benchmark soya bean meal, the reported GWP was higher for BSF (2.1 vs 1.7 kg CO_2 eq./kg of protein). However, if based on the more accurate calculations above, the GWP would be lower for BSF (0.91 kg CO_2-eq./kg of protein) than for soya bean meal. The EU per kilogram of produced protein via BSF was 15.1 MJ which is much higher than for soya bean meal (4.1 MJ). The land use, however, was far lower at 0.05 vs 8.65 m^2/kg of protein (Salomone *et al.*, 2017).

An assessment based on an Indonesian facility treating biowaste with BSF compared their GWP with composting (Mertenat *et al.*, 2019). The produced larvae were considered an alternative to Peruvian fishmeal and the avoided methane emissions arising from the composting were included in this study. The latter was excluded in Salomone *et al.* (2017). Waste sourcing and compost utilization were considered outside the system border and indirect emissions due to infrastructure, equipment and machinery, as well as direct CO_2 emissions, were excluded.

The BSF treatment resulted in a far lower GWP than composting (35 vs 111 kg CO_2 eq./t of food waste). This result was partially due to the limited climate control (ventilation only) and due to the avoided methane formation. However, the GWP was not expressed per kilogram of larvae, larvae meal, or protein, which impairs further comparisons.

Another assessment of BSF utilized a cradle-to-gate approach and included several processing steps to transform fresh BSF to defatted protein concentrate (Smetana *et al.*, 2019). Dried distiller grains with solubles (DDGS) and wheat by-products were used as feed ingredients. Pureed BSF and defatted concentrate were used as FUs, as were fertilizer production and fat production, the latter to be used in pig feed. Per kilogram of fresh larvae, the calculated GWP was 1.16 kg CO_2-eq., with an EU of 17.9 MJ and a land use of 0.48 m^2. Various other indicators were summed together, which impairs direct comparison with other insect LCAs. However, the authors do conclude that plant-based protein is currently the most sustainable. The greatest contributors to all categories of environmental impact of pureed BSF were feed production (43%) and energy use (37%).

HFs were assessed by LCA in a theoretical system utilizing chicken manure and food waste as feed for the larvae which subsequently would be utilized as a pig feed ingredient (van Zanten *et al.*, 2015). The FU, 1 ton of dried and milled HF larvae, was associated with 770 kg of CO_2-eq., an EU of 9329 MJ and land used of 32 m^2. This study also calculated the indirect consequences of the system. The larvae meal was assumed to replace soya bean meal and fishmeal on a 50:50 basis and the food waste, currently used to generate bioenergy, would no longer be available for that purpose. Incorporating these effects decreased land use by 1713 m^2 but increased GWP by 1959 kg CO_2-eq. and EU by 21.342 MJ/t of HF larvae meal.

7.4 Effect of Feed

LCAs for insect production systems clearly indicate that the feed utilized for production is a primary driver of environmental impacts in such systems. Feed production is associated with land, water, and energy use, and GHG emissions. Inefficient use of feed also leads to more eutrophication, ecotoxicity and acidification. Feed utilization, expressed as the FCR (kilogram of feed/kilogram of produced mass), can vary greatly even for similar species. Reported FCRs for HC and two-spotted crickets (2.50) (Halloran *et al.*, 2017), two-spotted crickets and BC (9.09) (Suckling *et al.*, 2020) and HCs (1.47) on broiler feed (Lundy and Parrella, 2015) indicate room for improvement. This could be achieved by better matching feed composition to the nutritional requirements of the insects, and by limiting feed losses. Also, harvesting at an optimal size, as suggested by Suckling *et al.* (2020), could decrease the FCR if not restrained by specific size requirements in the sales market.

Another way to decrease the environmental impact of insect production is to utilize underused feedstocks. Clearly, using waste products such as household waste or manure can hold much potential, if legally allowed and proven safe. Also using feed materials originally intended for conventional livestock but discarded due to contamination could improve the sustainability of insect production systems. As an example, grain products contaminated with mycotoxins are unsuitable for conventional production systems, but do not hamper the development of certain insect species (Bosch *et al.*, 2017; Camenzuli *et al.*, 2018). These mycotoxins seem to be catabolized by insects and therefore do not accumulate in the final product (Meijer *et al.*, 2019).

Similarly, some insect species are unaffected by certain heavy metals. Matching contaminated materials with species that efficiently excrete these heavy metals could allow the safe use of otherwise discarded substrates. For instance, cadmium-contaminated materials could be processed for YM and arsenic-contaminated materials could be used for BSF larvae (van der Fels-Klerx *et al.*, 2016). Further insights into the mechanisms utilized by these species to excrete or metabolize such contaminants are required, prior to utilizing these materials safely. However, when using troublesome waste products, it is essential that the insect species can use the material well to grow and develop. Studies on using polystyrene as feed for YM and wax moths indicate low growth rates and feed conversion efficiencies, indicating a low potential for commercialization (Billen *et al.*, 2020). Several by-products from the food industry have shown potential as insect diet ingredients and could decrease the environmental impact of insect production systems (Oonincx *et al.*, 2015; van Broekhoven *et al.*, 2015).

Besides utilizing by-products in insect production, the use of by-products from insect production can increase valorization and decrease environmental impact. If production is focused towards protein yields, the lipid fraction could be used as a biofuel (Wong *et al.*, 2018). Besides the insects themselves, the best known and most widely used by-product is insect frass.

Furthermore, frass might be a suitable substrate for anaerobic digestion and hence might function as an energy source, prior to being used as a fertilizer (Bulak *et al.*, 2020). Frass from mealworms, crickets and BSF larvae yielded 208–259 ml methane/g, which is comparable to benchmarks such as animal manures, organic wastes and sewage sludge, reported in that study. Subsequent utilization of the methane would reduce the need for fossil fuels as an energy source and thereby reduce EU and GWP, while retaining N, P and K in the substrate which could still be used as fertilizer.

7.5 Data Limitations

Within an LCA, decisions on the use of system borders, FUs, by-products and their allocation, and impact parameters are made. Different choices, for instance whether the purpose of a system is waste reduction or protein production, leads to different FUs, and the use of different benchmarks. Differences in system borders such as including construction materials, or utilizing residual materials as fertilizers, impair direct quantitative comparisons between publications. Also, variation in reported impact indicators and whether they are pooled or reported separately limits direct comparisons.

While the number of insect LCAs is limited, the aforementioned methodological differences made it necessary to include only the more commonly used indicators and focus on explaining the utilized system borders and FUs in this chapter. Hopefully in the coming decade, more detailed LCAs using similar system borders will become available allowing more direct comparisons. These would preferably utilize a cradle-to-farm-gate approach, several FUs and be based on large-scale production facilities. Moreover, in several LCAs on insects for food or feed, improved scenarios are explored (Halloran *et al.*, 2017; Smetana *et al.*, 2019; Suckling *et al.*, 2020). These provide an outlook on the future regarding the potential development of an environmental impact. However, they should be interpreted with extreme caution as they often contain unproven assumptions.

7.6 Conclusions

Even though direct comparisons between the conducted studies are hampered, some relevant conclusions can be drawn regarding the sustainability of insect production. Land use associated with insect production generally seems low, compared to conventional feed and food products. The EU (expressed as fossil fuel depletion) of insect production is often high compared to conventional products. To a large extent this is because several LCAs have been conducted for systems in temperate climates, which require extensive climate control. This also leads to an elevated GWP due to the emission of GHGs associated with the used energy. Besides energy consumption during the rearing process, a large part of the environmental impact is due to the production of feed for the insects. This effect can be mitigated by using lower impact feed sources, assuming that feed can be used efficiently, thereby decreasing the environmental impact associated with insect production.

As large-scale insect production systems are relatively new and rapidly developing, it seems reasonable to expect increased efficiency and thereby decreased environmental impact as the sector progresses.

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