# WHITE PAPER:

# THE BIOFILTRO BIDA® WASTEWATER TREATMENT SYSTEM

**Hydro Focus/ BioFiltro**

**Sabina Dore, Steve Deverel, Andrés Iacobelli, Matías Sjögren**

**March 2019**

## DESCRIPTION OF THE BIDA® SYSTEM

This paper aims to describe the BIDA® (**BI**ofilter **D**ynamic **A**erobic) System and its different applications in the United States (US), quantify its multiple benefits, and report plans and priorities for research and development. The BioFiltro BIDA® System was patented in 2009 as a system for treating organic contaminants in water by vermifiltration, a biological treatment process using earthworms and microorganisms to degrade the organic load of wastewater. The BIDA® System filters water onsite, converting wastewater into a reusable asset and contaminants into a natural and nutritious fertilizer.

To address the need for treatment of wastewater for agriculture and rural areas, vermifiltration has been demonstrated to be a decentralized, low-cost, and low-maintenance solution (Lazcano and Dominguez 2011). Vermifiltration has been adopted recently to treat domestic wastewater in developing countries (Xing et al 2015), industrial waste (Lin et al 2013), and livestock waste (Li et al 2008, Luth et al 2011, Wang et al 2014). Studies have reported high nutrient removal by vermifiltration of nitrogen (N), ammonia ($NH_3$) and hydrogen sulfide ($H_2S$), which are nuisance gases and respiratory irritants (Wang et al 2014), and nitrous oxide ($N_2O$), carbon dioxide ($CO_2$) and methane ($CH_4$), powerful greenhouse gases (GHGs) (Lai et al 2018). The BioFiltro BIDA® System has been proven to reduce water biological oxygen demand (BOD), total suspended solids (TSS), volatile solids (VS), total dissolved solids (TDS), fat oils and grease (FOG), total Nitrogen (N), Ammonia ($NH_3$), and Phosphorus (P).

The BIDA® System is a sustainable wastewater management strategy. By removing N from wastewater at its source, the BIDA® System also prevents downstream environmental and public health issues related to excess N volatilization and leaching into soil and groundwater and movement to surface waters, which can result in eutrophication, soil acidification, and groundwater pollution (Galloway et al 2003).

Earthworms mechanically degrade organic input and excrete mucus and castings which are enriched in organic matter, N, and P, thus making these nutrients readily available to microorganisms (Aira et al 2007; Zhao et al 2010). Earthworms' tunneling activity contribute indirectly to nutrient removal by increasing the porosity and aeration of the media, which facilitate the growth of aerobic microorganisms (Zhao et al 2010; Wang et al 2011a).

The BIDA® System can be classified as a biological system for aerobic treatment of organically contaminated wastewater based on a pre-straining filter modification. It is superior to the activated sludge and pre-strained filter techniques, which are the most common aerobic wastewater treatment systems because these systems produce solid by-products (sludge) that require secondary disposal. Also, these systems have higher operational costs because of the continuous use of aerators and the need of pumps to remove sludge. In pre-straining filters, finally, the filtering media becomes increasingly impermeable considerably reducing its efficiency. Compared to these traditional aerobic wastewater treatment systems, the BIDA® System: does not produce unstable sludge because the system degrades all of the organic soluble solids and transforms them into a stable, odorless humus material. The filtering material remains permeable because the bacterial flora and earthworms are constantly moving the solids, tunneling the medium and allowing the medium to remain porous. The operational costs are low because the system only needs impulsion pumps to distribute homogeneously the contaminated water over the surface of the BIDA® System, whereas other treatments require air or oxygen injection, or addition of chemicals and other substances. Finally, earthworms produce vermicompost from degraded organic material and castings that is a high-quality natural fertilizer.

Constraints of the BIDA® System include the need to separate solids before entering the system and its sensitivity to composition changes in contaminated water. Variations in wastewater quality will affect and can potentially damage the earthworms and bacterial flora.

The BIDA® System includes layers of cellulose-based and inert materials that filter the wastewater, provide habitat for the earthworms, and maintain conditions aerobic.

The surface layer is inoculated with an industry-specific mix of worms (*Eisenia fetida and Eisenia andrei*) and bacteria to achieve wastewater treatment. The burrowing worms create air channels and digest suspended solids, achieving densities up to 12,000 worms per cubic meter (yard). When liquid residues are in contact with the worm humus, a bacterial flora forms and degrades the organic material present in the contaminated water. Because this process adapts to the specific organic waste, it can be used in many different applications such as cleaning domestic sewer water and wastewater from slaughterhouses, dairies, livestock, vineyards, and food processing industries.

To avoid impermeability of the cellulose medium and subsequent anaerobic conditions, it is preferable to separate the majority of solids — in particular, fixed inorganic solids that cannot be degraded — from the liquid waste.

Due to the rapid four-hour processing time, the BIDA® System is virtually odorless and requires minimum storage capacity. The system permits discontinuous or seasonal operations because during the offseason the worm and bacteria biomass can survive off of the cellulose media.

## BIOFILTRO EXPERTISE AND BIDA® SYSTEM PERFORMANCE

The development of the BIDA® System for filtering organic liquid waste at an industrial scale started in the 1990s in Chile. Since the first BIDA® System built in 1995 to process 50,000 liters (13,000 gallons) per day of sanitary waste in a remote community in Chile, BIDA® Systems have been installed in 8 countries including US, New Zealand, Spain, Brazil, Mexico, Perú and Australia. Applications range from an operation in the harsh conditions of Antarctica, to the largest system treating 7.6 million liters (2 million gallons) of water per day, for a total of 150 worldwide full-size plants in 8 different industries. The BIDA® Systems have been operated for 25 years by the international wastewater filtration company BioFiltro which maintains plants in the US (13 plants), New Zealand (9 plants), Chile (112 plants), Spain (7 plants) Brazil, (4 plants) Mexico (2 plants) Australia (1 plant) and Perú (1 plant) and are currently serving:

1) Dairy farms (7 plants);
2) Sanitary waste generators (67 plants);
3) Wineries (12 plants);
4) Food processors (20 plants);
5) Milk processors (22);
6) Slaughterhouses (8 plants);
7) Livestock (5 plants);
8) Aquaculture (9 plants).

In the US, BIDA® System is used in dairies, food processing industries, municipal waste generators, slaughterhouse, and wineries. The system is used mainly to remove putrescible organic material (expressed as Biological Oxygen Demand or BOD) in food industries and municipal plant wastewater with removal efficiencies ranging from 87% to 95% (Table1). It is also used to remove suspended particles that are not dissolved in water (expressed as total suspended solids or TSS) from wastewater at removal efficiencies of 76% to 93% (predominantly in dairy farms, slaughterhouses, and in wineries) (Table 1). The BIDA® System removes 80% to 97% of organic N and ammonia (TKN) from dairy and food processing industry wastewater (Table 1). Current operational full-size plants in the US vary between 500 and 80,000 square feet and treat between 38,000 to 76 million liters (10 thousand to 20 million gallons) per year.

**Table 1: Characteristics of the BIDA® System and its efficacy**

| USA Sector | N plants | Size (sq feet) | Flow design (gpd) | Electricity consumption (kWh gallon$^{-1}$) | Start (year) | REMOVAL | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | | | | | BOD | TSS | TKN | TP |
| Dairy | 5 | 15,066 | 37,000 | 0.002 | 2013 | 52% | 85% | 80% | 83% |
| Food | 6 | 59,644 | 18,000 | 0.003 | 2014 | 87% | 85% | 97% | |
| Municipal | 2 | 3,010 | 28,000 | 0.001 | 2014 | 95% | 93% | | |
| Slaughterhouses | 1 | 118 | 300 | 0.006 | 2017 | 91% | 86% | | |
| Wine | 6 | 4,271 | 19,000 | 0.004 | 2014 | 90% | 91% | | |

**Removals reflects goals set at each plant based on specific needs, with values ranging between 50% and 90% of specific industry parameters. Operations include both, pilot and full-size projects**.

## DAIRY WASTEWATER CHALLENGES

The use of BIDA® System in dairy farms provides numerous co-benefits in addition to the treatment of wastewater, including reduction of GHG emissions of $CH_4$ and $N_2O$, the reduction of soil and air pollution from N and $NH_3$, and the production of organic fertilizer for improvement of soil health and soil carbon sequestration (Figure 1). The BIDA® System can be applied to farms of variable size with any confined livestock and results in large benefits compared not only to the anaerobic lagoons traditionally used to store and treat waste but also to anaerobic digesters. The multiple, synergic benefits merit a holistic examination.

There is much interest in understanding effects of manure management on GHG emissions, as manure contains substantial quantities of N, carbon, and water, three essential factors controlling processes leading to production and emissions of $N_2O$ and $CH_4$. Whilst manure is a source of GHGs, the management practice selected by farmers influences the magnitude of gaseous losses and has the potential to reduce those emissions. Dairies with confinement housing, especially for large herd sizes, face challenges in handling manure in environmentally sound, economically sustainable ways. In the US, the general trend of the industry is that animal populations have become more concentrated in certain areas of the country and the number of animals per facility has increased. These areas of concentration, such as California, New Mexico, and Idaho, tend to utilize more liquid-based systems (anaerobic lagoons) to manage and store manure. Also, new regulations controlling the application of manure nutrients to land have shifted manure management practices at smaller dairies from daily spread systems to storage and management of the manure on site. Thus, anaerobic lagoons are increasingly used to manage manure in the US (in California they were used on 58% of the dairy farms in 2016, CARB 2018). However, these systems cause the highest $CH_4$ emissions (IPCC 2006, Owen and Silver 2014). In addition, anaerobic lagoons produce undesirable odors unless provisions are made to oxidize the escaping gases. They require a relatively long detention time for organic stabilization due to the slow growth rate of the methane formers and sludge digestion and the ability to control the process is limited because environmental conditions directly impact the efficacy of the operation (e.g. lagoons are sensitive to temperature). If unlined, infiltration of wastewater can result.

Today, anaerobic digesters are the instrument designated to mitigate the large $CH_4$ emissions from anaerobic lagoons, especially in dairy farms. However, anaerobic digesters typically require large investments of financial and human capital and substantial managerial expertise. The capital cost for a typical farm can reach close to $5 million and the digester's operating costs are larger than any other operating cost except for feed and replacement cows (Lee and Sumner 2018). In addition, combustion of biogas during electricity generation emits $N_2O$, a substance regulated by the federal Clean Air Act (EPA 2017). Recent studies showed anaerobic digestors increase $NH_3$ concentrations in treated waters (Leytem et al 2018, Holly et al 2017), which require additional emission control technologies. The need to comply with federal regulations has made anaerobic digesters an expensive strategy for reducing dairy $CH_4$ emissions. And consequently, despite government support, anaerobic digesters are practical only for large operations and not extensively adopted.

The dairy industry faces many economic and policy challenges, none of which is more vital than how to deal with environmental concerns and related regulations. States are now developing strategies and regulations to control pollution and emissions from animal agriculture. For example, in California, the nation's top dairy state with substantial overseas exports of milk products, the California Department of Food and Agriculture (CDFA) and the California Air Resources Board (CARB) among other agencies, are in the midst of an aggressive attempt to reduce GHG emissions from agricultural production and processing (Lee and Summer 2018). $CH_4$ emissions associated with manure produced at California livestock operations will be subject to detailed regulations which will become effective in 2024 (Lara 2016).

In 2017, agricultural activities were responsible for 8.4% of total US GHG emissions. $CH_4$ emissions from manure management represented approximately 11% of $CH_4$ emissions from agriculture. Manure management can also be an important source of $N_2O$ emissions, which is one of the longest-lived GHG and has an estimated radiative forcing 298 times that of $CO_2$ (while $CH_4$ is currently considered 25 times that of $CO_2$, IPCC 2007). Manure from livestock production systems contributed 8% of the $N_2O$ emissions from US agriculture (EPA 2019a). In addition to its contribution to global warming, $N_2O$ also plays an important role in stratospheric ozone depletion (Ravishankara et al 2009).

Overall, US GHG emissions from manure management increased by 57.1% between 1990 and 2017. This encompassed an increase of 66% for $CH_4$ emissions, from 37.1 million tons of carbon dioxide equivalent ($tCO_2eq$) in 1990 to 61.7 million $tCO_2eq$ in 2017, and an increase of 33% for $N_2O$ in 2017. The majority of the increase observed in $CH_4$ emissions resulted from swine and dairy cattle manure, where emissions increased 29% and 134%, respectively, from 1990 to 2017 (EPA 2019a).

Soil N supply is often limited, so farmers increase the amount of N fertilizers to achieve better crop yield, and dairy manure application is a possible way to add N to soils. However, as plants are not able to absorb excess N from fertilizers, it has been estimated that 50% - 70% of the N provided to the soil is lost (Hodge et al 2000). Nitrate leaching, soil denitrification, and volatilization are the main processes for excess N loss and contribute to environmental pollution. Nitrate leaching contaminates groundwater and other bodies of water, which may contribute to eutrophization. In addition, volatilized N contributes to global warming by releasing $N_2O$. The increase of N in vegetable leaves is an important factor in the development of several human diseases (Park et al 2012).

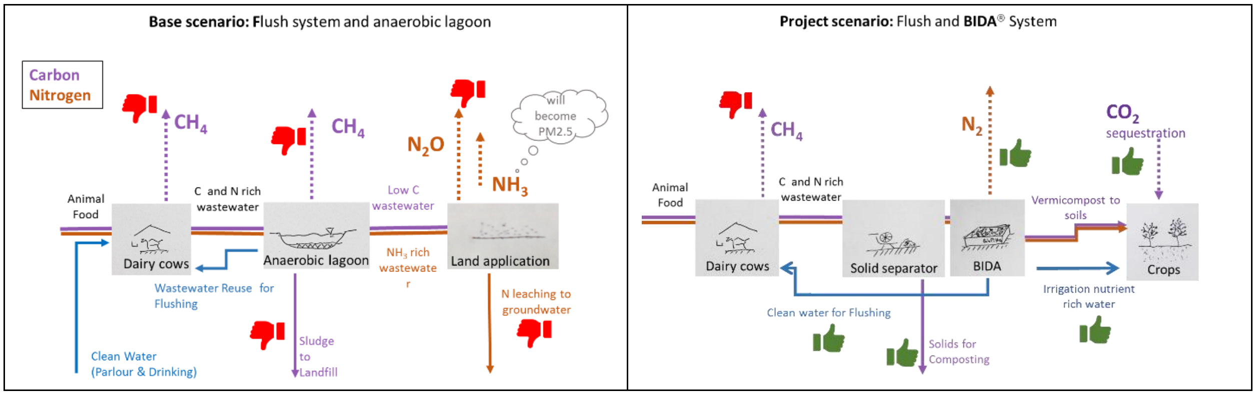


**Figure 1: Comparison of carbon and nitrogen dynamics in the treatment of dairy wastewater using anaerobic lagoon or the BIDA® System**

**Table 2: Characteristics of the BIDA® System applications and its efficacy in the US dairy sector.**

| Dairy | Phase | Size | Flow design | Electricity consumption | Start year | REMOVAL | | |
|---|---|---|---|---|---|---|---|---|
| | | (sq. feet) | (gpd) | (kWh gallon $^{-1}$) | (year) | TSS | TKN | TP |
| CSU Fresno Dairy | Pilot | 538 | 2,000 | 0.0028 | 2014 | 64% | 92% | |
| Fanelli | Pilot | 5,800 | 13,500 | 0.0011 | 2014 | 83% | 77% | |
| G. Deruyter | Pilot | 109 | 300 | 0.0047 | 2016 | 73% | 54% | |
| J. Deruyer | Pilot | 109 | 300 | 0.0047 | 2016 | 98% | 85% | |
| Royal Dairy | Pilot | 2,000 | 5,000 | 0.0008 | 2015 | 97% | 93% | |
| | 2nd phase | 81,838 | 200,000 | 0.0003 | 2017 | 93% | 78% | 83% |

**Values are averages of monthly values for the duration of the operation, including periods of malfunctioning. Removal of total suspended solids (TSS), Total organic nitrogen and ammonia (TKN) and phosphorous (TP).**

Fine particulate matter PM2.5 can form from the emission of gases — including N containing gasses such as oxides of nitrogen (NOx) and $NH_3$ — that turn into fine particulates in the atmosphere through chemical reactions or condensation. $NH_3$ budgets in the troposphere show that about 90% of emissions in the US results from animal and crop production (Davison and Cape 2003). Because $NH_3$ plays a significant role in the formation of PM2.5 and because it is a proxy for odor issues, $NH_3$ is subject to environmental regulations. Anaerobic digesters are currently supported for their ability to reduce net $CH_4$ emissions by using $CH_4$ to produce energy, but result in a GHG emission tradeoff as they can increase $NH_3$ emissions by up to 80% and increase available N (Bernet et al 2000, Holly et al 2017, Leytem et al 2018, Nkoa 2014).

One of the most valuable benefits of the BIDA® System, compared to other wastewater treatment systems and particularly the anaerobic digesters is the removal of organic nitrogen and ammonia (TKN). This is of great importance for dairy farmers often limited by the need to safely dispose of their waste. For this reason, the crops planted on dairy farms are usually low-value crops that can receive the maximum amount of N fertilizer. The BIDA® System's removal of N in dairy wastewater translates to an increase in the number of animals allowed on a farm, the ability to switch to high values crops and to change land use, the reduction of water usage, and finally, the reduction in odor from manure stores.

Because of its efficacy in reducing both $CH_4$ emissions and N loads from dairy wastewater, the BIDA® System can be used as an alternative to anaerobic digesters to reduce $CH_4$ emissions, with the added benefit of reducing N loads. The BIDA® System can also be used in conjunction with anaerobic digesters. Treating already digested wastewater greatly reduces N loads as well as the residual $CH_4$ emissions. This setup allows a small sized BIDA® System to treat a large quantity of wastewater with a low additional cost compared to the cost of a biodigester.

## CURRENT RESEARCH AND FINDINGS

In the US, the use of the BIDA® System to treat dairy wastewater started in 2013. Several pilot studies preceded full-scale operations. BIDA® System plants range from 45 $m^2$ (500 sq. feet) to 7400 $m^2$ (80,000 sq. feet) at Royal Dairy in Washington State. These systems are designed to treat around 100 to 1,500 cows daily, up to 760,000 liters (200,000 gallons) of water per day, and work continuously throughout the year (Table 2). Suspended solids (TSS) are reduced on average by 85%. The BIDA® System in the dairy sector is limited by the capacity to accommodate a large quantity of TSS in the wastewater. For this reason, the BIDA® System requires upstream solid separation and monitoring of TSS levels to assess the need for dilution of wastewater by recycling treated water.

### Water recycling

Agriculture accounts for 80% of water use and contributes to water shortages, groundwater overdraft and depletion (Pimentel et al. 2004). Climate models predict droughts will increase in some regions, such as California, due to anthropogenic warming (Diffenbaugh et al 2015). These predictions, in conjunction with the economic impacts of drought on agriculture, underscore the importance of designing agricultural management strategies that conserve water.

The BIDA® System allows on-farm recycling of water. Because of the BIDA® System's ability to remove 85% of TSS and 80% of N load from dairy wastewater (Table 2), the high-quality water resultant from treatment can have a wide range of applications. Water can be reused in the dairy process or used for irrigation on a broad range of crops. Assuming a conservative 50% N removal, farmers could reduce land surface used for wastewater disposal by 50%, or about 0.2 acres per cow.

In the foreseeable future, a market for cleaner water is also possible. The Environmental Protection Agency (EPA) is supporting the use of water quality trading offsets and similar programs for achieving compliance with regulatory requirements to maximize pollutant reduction efforts and improve water quality. The use of market-based programs would aim to reduce water pollution at a lower overall cost and incentivize implementation of technologies and land use practices that reduce pollution in water (EPA 2019b).

The BIDA® System's large and consistent reduction of solids in (Table 3) proves its capacity to reduce dairy $CH_4$ emissions. $CH_4$ is produced during the anaerobic degradation of the volatile solid (VS) component of the wastewater. The constant ratio between TSS and VS measured in multiple US dairy farms (Table 5) implies that a reduction of TSS will result in a correspondent 85% reduction of VS, thus strongly limiting the capacity of dairy wastewater to produce $CH_4$ emissions.

### Low power requirement

The BIDA® System energy requirement is minimal in that it is merely the electricity needed for irrigation of wastewater on the system surface. Irrigation pumps activate only when facilities discharge and, due to the intermittent irrigation schedule, typically demand energy for a total of 5 hours a day for the days the facility is active. Compared to the status quo, no fossil fuel usage is needed to remove sludge nor to periodically clean the lagoons. Typical plants consume 0.0003 to 0.006 kWh per gallon water treated and full-size operations use up to 11,000 kWh a year, approximately 10 times the average US household (EIA 2015).

### GHG emissions and N load reductions

The majority of research in vermifiltration has focused on optimizing vermifilter performance as measured through nutrient removal efficiency in domestic wastewater (Liu et al. 2012, Wang et al., 2011, Wang et al. 2014) and elucidating the microbial communities responsible for these effects (Li et al 2013, Liu et al 2012, Wang et al 2016). Few studies examined vermifiltration of livestock wastewater.

The study of Lai et al. (2018) measured the effects of the BIDA® System on N compounds, GHG emissions, and microbiota at a commercial dairy. The studies concluded that the BIDA® System reduced $NH_3$ emission by 90.2% without substantially increasing emission of $N_2O$, $CO_2$, $CH_4$ compared to the untreated lagoon water (Table 4), confirming similar results obtained from Luth et al. (2011). Similarly, nutrients and organic compounds removed from the wastewater were not converted into ethanol or other volatile organic compounds (VOCs), which are atmospheric smog precursors. These authors determined that N was removed from wastewater via enhanced N-cycling in the BIDA® System, and didn't increase production of $N_2O$, a byproduct of incomplete denitrification (Table 3). In the scientific literature, earthworms showed increased (Drake et al 2007, Lubber et al 2013) or decreased emissions of $N_2O$ and $CH_4$ (Li et al 2008, Luth et al 2011). The Lai et al study (2018) suggests that earthworms reduced $NH_3$, $N_2O$ and $CH_4$ emissions when concentration of these gases is high, as seen in the BIDA® System. In contrast, earthworms increase emissions in soils, where background gas concentrations are low (Lai et al 2018).

The water quality analysis in the Lai et al paper (Table 3) revealed a 2-fold reduction in total organic bounded N (TKN) and a 3-fold reduction in $NH_3$ coupled with only a slight increase in $NO_3$ and $N_2O$. The study demonstrated that the BIDA® System enriched the wastewater microbiome in taxa known to participate in nutrient degradation and cycling, specifically bacteria and archaea capable of N transformations. Minute emissions of $N_2O$ despite the overall reduction in $NH_3$ and TKN, indicate complete denitrification to $N_2$ as opposed to incomplete denitrification to $N_2O$ (Table 3). The study concluded that the BIDA® System provided an ideal ecosystem to facilitate the microbial decomposition and removal of organic N from dairy wastewater, decreasing downstream N load on the soil or groundwater without increasing GHG emissions

A preliminary assessment of the GHG benefit derived from using the BIDA® System is based on measurements at a typical mid-size dairy farm in CA: the BioFiltro dairy operation in Merced County, CA described in the paper by Lai et al. (2018). GHG emission reductions were calculated as the difference between baseline conditions and GHG fluxes resulting from the BIDA® System treatment. Manure was collected by a flush system and stored in an anaerobic lagoon. To estimate the GHG emission reduction, 150 cows of the total 1105 on the farm were considered, as this was the number of animals the system was designed to support. To calculate the GHG emissions per cow, a population of 118 dairy cows was used, considering the proportion between excretion produced by dairy and non-dairy cows. GHG emissions resulting from fossil fuels for electricity and equipment related to the management of manure were not considered. The BIDA® System was not directly treating flushed water, but water pumped from the lagoon. Consequently, the fluxes measured in the BIDA® System were likely, in part due to processes started in the lagoon and may be different if wastewater was treated immediately.

.

**Table 3**

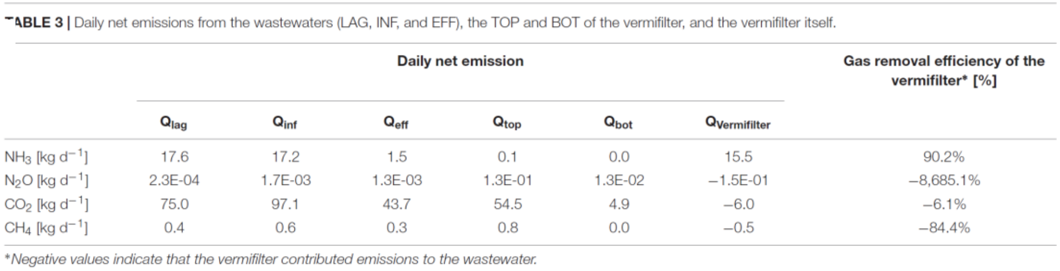


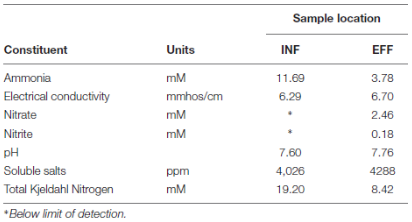
**TABLE 3 |** Daily net emissions from the wastewaters (LAG, INF, and EFF), the TOP and BOT of the vermifilter, and the vermifilter itself.

| | Daily net emission | | | | | | Gas removal efficiency of the vermifilter* [%] |
|---|---|---|---|---|---|---|---|
| | $Q_{lag}$ | $Q_{inf}$ | $Q_{eff}$ | $Q_{top}$ | $Q_{bot}$ | $Q_{Vermifilter}$ | |
| $NH_3$ [kg $d^{-1}$] | 17.6 | 17.2 | 1.5 | 0.1 | 0.0 | 15.5 | 90.2% |
| $N_2O$ [kg $d^{-1}$] | 2.3E-04 | 1.7E-03 | 1.3E-03 | 1.3E-01 | 1.3E-02 | −1.5E-01 | −8,685.1% |
| $CO_2$ [kg $d^{-1}$] | 75.0 | 97.1 | 43.7 | 54.5 | 4.9 | −6.0 | −6.1% |
| $CH_4$ [kg $d^{-1}$] | 0.4 | 0.6 | 0.3 | 0.8 | 0.0 | −0.5 | −84.4% |

*Negative values indicate that the vermifilter contributed emissions to the wastewater.*

Adapted from Lai et al 2018. Profiling of the microbiome associated with nitrogen removal during vermifiltration of wastewater from a commercial dairy. Frontiers in microbiology, 9, p.1964.

**Table 4: Wastewater chemistry analyss of influent and effluent**

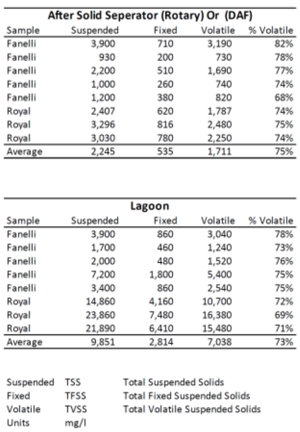


| | | Sample location | |
|---|---|---|---|
| Constituent | Units | INF | EFF |
| Ammonia | mM | 11.69 | 3.78 |
| Electrical conductivity | mmhos/cm | 6.29 | 6.70 |
| Nitrate | mM | * | 2.46 |
| Nitrite | mM | * | 0.18 |
| pH | | 7.60 | 7.76 |
| Soluble salts | ppm | 4,026 | 4288 |
| Total Kjeldahl Nitrogen | mM | 19.20 | 8.42 |

*Below limit of detection.*

Adapted from Lai et al 2018. Profiling of the microbiome associated with nitrogen removal during vermifiltration of wastewater from a commercial dairy. Frontiers in microbiology, 9, p.1964.

**Table 5: Relationship between total and volatile suspended solids**



**Total Volatile Suspend Solids**

**After Solid Seperator (Rotary) Or (DAF)**

| Sample | Suspended | Fixed | Volatile | % Volatile |
|---|---|---|---|---|
| Fanelli | 3,900 | 710 | 3,190 | 82% |
| Fanelli | 930 | 200 | 730 | 78% |
| Fanelli | 2,200 | 510 | 1,690 | 77% |
| Fanelli | 1,000 | 260 | 740 | 74% |
| Fanelli | 1,200 | 380 | 820 | 68% |
| Royal | 2,407 | 620 | 1,787 | 74% |
| Royal | 3,296 | 816 | 2,480 | 75% |
| Royal | 3,030 | 780 | 2,250 | 74% |
| Average | 2,245 | 535 | 1,711 | 75% |

**Lagoon**

| Sample | Suspended | Fixed | Volatile | % Volatile |
|---|---|---|---|---|
| Fanelli | 3,900 | 860 | 3,040 | 78% |
| Fanelli | 1,700 | 460 | 1,240 | 73% |
| Fanelli | 2,000 | 480 | 1,520 | 76% |
| Fanelli | 7,200 | 1,800 | 5,400 | 75% |
| Fanelli | 3,400 | 860 | 2,540 | 75% |
| Royal | 14,860 | 4,160 | 10,700 | 72% |
| Royal | 23,860 | 7,480 | 16,380 | 69% |
| Royal | 21,890 | 6,410 | 15,480 | 71% |
| Average | 9,851 | 2,814 | 7,038 | 73% |

| | | |
|---|---|---|
| Suspended | TSS | Total Suspended Solids |
| Fixed | TFSS | Total Fixed Suspended Solids |
| Volatile | TVSS | Total Volatile Suspended Solids |
| Units | mg/l | |

**Table 6: Analysis of contaminant removals at Fanelli farm - March 2019.**

| Analyte | Influent (mg/L) | Effluent (mg/L) | Reduction |
|---|---|---|---|
| Ammonia | 590 | 27 | 95% |
| BOD | 2600 | 560 | 78% |
| TKN | 990 | 140 | 86% |
| TN | 990 | 150 | 85% |
| Total suspended solids | 9000 | 610 | 93% |
| Total volatile Solids | 18000 | 2000 | 89% |

First, <u>baseline GHG emissions</u> of a generic dairy farm in California, characterized by the presence of an anaerobic lagoon and the absence of any practice aimed to reduce GHG emissions were estimated. Baseline GHG emissions (Table 7) were calculated using, as follows:

1) The annual California GHG emissions inventory for the livestock sector for 2016 (EPA 2019a). The annual GHG emission rate per cow from manure stored in anaerobic lagoons derived from the inventory was 8.3 $tCO_2eq$ $yr^{-1}$. Direct $N_2O$ emissions from anaerobic lagoons were 0.3 $tCO_2eq$ $yr^{-1}$ $head^{-1}$ and indirect $N_2O$ fluxes from soils were 1.1 $tCO_2eq$ $yr^{-1}$ $head^{-1}$. Thus, $N_2O$ emissions were circa 20% of the CH4 emissions.
2) The emission factors from the Owen and Silver (2014) study, which compiled field-scale measurements of GHG emissions from working and research dairies. Estimated emissions from the anaerobic lagoons were 9.2 $tCO_2eq$ $yr^{-1}$ $head^{-1}$ for $CH_4$ and 0.3 $tCO_2eq$ $yr^{-1}$ $head^{-1}$ for $N_2O$.
3) The CDFA Alternative Manure Management Practice (AMMP) GHG Calculator Tool (CARB 2019). The CDFA calculation tool is based on the guidelines for measuring GHG sink and sources of the International Panel on Climate Change (IPCC 2006). The CDFA calculation tool sets most of the parameters and emission factors for California. Baseline $CH_4$ emissions estimated using the CDFA tool were 4.1 $tCO_2eq$ $yr^{-1}$ $head^{-1}$, which were lower than the previous baseline estimates because the tool assumes that a fraction of daily excreted manure is left on the ground and is thus excluded from producing $CH_4$. The tool also estimates lower baseline GHG emissions due to specific conversion factor (MCF) and temperature effects calculations.

The GHG emission reduction is the difference between the baseline scenario and GHG emissions due to a BIDA® treatment system (Table 7). <u>Project BIDA® GHG emissions</u> were calculated as follows:

1) Using the CDFA Calculator Tool assuming that the BIDA System was an additional high-efficiency separator able to retain 83% of the VS before entering the anaerobic lagoon and with the separated solids turned aerobically to compost. For this scenario GHG emissions were 0.5 $tCO_2eq$ $yr^{-1}$ $head^{-1}$.
2) Measured daily fluxes In Lai et al. 2008, were scaled up to annual fluxes and corrected for effects of temperature on $CH_4$ and $CO_2$ production. In this case, the BIDA® System's capacity to reduce emissions of $N_2O$ were included, which is not included in the CDFA Calculator Tool. Annual BIDA® System GHG emissions per cow were circa 0.4 $tCO_2eq$ $yr^{-1}$ $head^{-1}$ (Table 7).

Details on baseline and BIDA® System GHG emission estimations are described in the Appendix.

Measured BIDA® System emissions were in agreement with $CH_4$ emissions estimated by the CDFA Calculator Tool. They were much lower than baseline emission, and also 25% lower compared to the 2.1 $tCO_2eq$ $yr^{-1}$ $head^{-1}$ reported for anaerobic digesters by the CA GHG inventory. Thus, GHG emissions were reduced by the BIDA® System circa 3.5 t $CO_2eq$ $yr^{-1}$ $head^{-1}$ at Fanelli Dairy and by circa 8. 5 t $CO_2eq$ $yr^{-1}$ $head^{-1}$ compared to a generic, same sized farm in California (Table 7).

$N_2O$ emissions for the BIDA® System were estimated using the IPCC guidelines (2006). Baseline GHG emissions included direct $N_2O$ emissions and volatilization losses from the management system and direct and indirect $N_2O$ emissions resulting from the application of dairy manure to cropped soils. This baseline emission was compared to the direct and indirect $N_2O$ emissions resulting from the use of the BIDA® System. Considering the 80% N load reduction in wastewater (Table 4), this emission component decreased from 1.1 $tCO_2eq\ yr^{-1}\ head^{-1}$ to 0.2 $tCO_2eq\ yr^{-1}\ head^{-1}$, adding to the total GHG mitigation benefit.

Substituting vermicompost produced by the BIDA® System to a generic crop-irrigated soil instead of synthetic fertilizers or for mulching was estimated to add 3.4 to 4.4 $tCO_2eq\ yr^{-1}$ (see Appendix for details).

Based on preliminary calculations, the BIDA® System results in a substantial reduction of net GHG emissions relative to conventional treatment methods.

**Table 7. GHG emissions from livestock manure treatment systems in California**

| | | GHG gas | Source | Methods | GHG emission (tCO$_2$eq year$^{-1}$ head$^{-1}$) |
|---|---|---|---|---|---|
| **BASELINE** | Anaerobic lagoon | CH4 | A. lagoon | CA Inventory | **8.3** |
| | | | | Owen | **9.2** |
| | | | | IPCC/CDFA | **4.8** |
| | | N2O | A. lagoon | CA Inventory | **0.3** |
| | | | | Owen | **0.3** |
| | | | Cropped soils | CA Inventory | **1.1** |
| | Fanelli | CH4 | A. lagoon | IPCC/CDFA | **4.1** |
| **PROJECT** | BioFiltro BIDA® System | CH4, N20, and CO2 | BIDA® System | Gas measurements | **0.4** |
| | | N2O | Cropped soils | CA Inventory | **0.2** |
| | | CH4 | BIDA® System | IPCC/CDFA | **0.5** |
| | Anaerobic Digestor | CH4 | Digestor | CA Inventory | **2.1** |

**GHG emission and emission reduction are expressed as ton $CO_2$ equivalent per cow. Global warming potential for $CH_4$ is 25 and for $N_2O$ is 298, by current IPCC standards (IPCC 2007). Emissions from separated solids are not included from both baseline and project (treatment) emissions.**

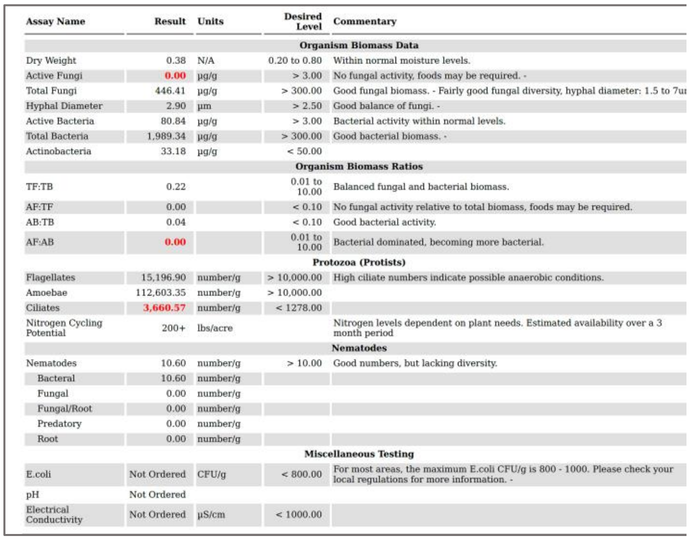
## CARBON SEQUESTRATION, HEALTH, AND N AVAILABILITY IN SOILS

Cattle manure is a valuable resource as a soil fertilizer as it provides high contents of organic matter (OM) and nutrients for crop growth and is a low-cost alternative to mineral fertilizers. However, overproduction of this waste substance has led to inappropriate disposal practices that can cause serious environmental problems, including an excessive input of potentially harmful trace metals, inorganic salts, and pathogens; increased nutrient loss from soils through leaching, erosion, and runoff; and the emission of $NH_3$ and other toxic gases (Hutchison et al 2005). Composting and vermicomposting reduce environmental problems associated with manure management. Manure can produce high-quality mulches for agricultural use and, with further maturation and elimination of phytotoxic compounds, high-quality organic fertilizers (Carr et al 1995).

OM in the soil is critical for maintaining balanced soil biological communities, as these are largely responsible for maintaining soil structure, increasing water infiltration, and building the soil's ability to store and release water and nutrients for crop use. Addition of OM to agricultural soil increases sustainability because it recycles organic waste, especially if produced on-farm; reduces GHG emissions because of long term sequestration of C in soils and because of the reduced need to produce fertilizers. Addition of OM to soil is hence a land management strategy for mitigation of climate change. Past studies have shown that the application of OM with a low dry matter content (i.e. livestock slurry and digestate) had very little impact on soil OM levels (Wrap 2016). In addition, the rate of mineralization is highest when the OM is first incorporated in the soil and can differ from the pattern of plant uptake. Therefore, it can be challenging to schedule OM application so that the rate of available N supply coincides with the rate of N uptake. Crops with high uptake rates for a short time are not considered well adapted to be fertilized solely by OM. Progressive accumulation of N from the addition of OM can increase the risk of soil N losses through nitrate leaching, denitrification and $NH_3$ volatilization (Rodrigues et al 2002). A tight plant-soil-microbe N cycling system is a desirable scenario which reduces tradeoffs among yields, water quality, and GHG emissions. In addition to the general benefit of OM additions to soils, vermicompost has its unique benefits. As a result of the different processes involved in the production of compost and vermicompost, soils exhibit different physical and chemical characteristics and thus properties in both the bacterial community composition, fungal abundance, and plant growth (Lazcano and Dominguez 2011).

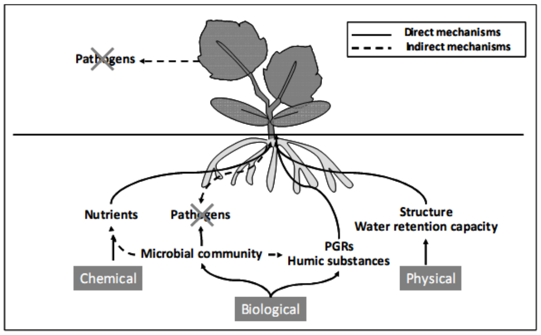
Vermicomposting involves the bio-oxidation and stabilization of organic material by the joint action of earthworms and microorganisms. Although it is the microorganisms that biochemically degrade the organic matter, earthworms are the crucial drivers of the process. They aerate, condition and fragment the substrate, thereby drastically altering the soil microbial activity which determines soil chemical and physical status. The C:N ratio is gradually reduced and surface area exposed to microorganisms increased– thus making it much more favorable for further microbial activity and decomposition (Domínguez et al 1997). Vermicompost is, therefore, a complex mixture of earthworm feces, humified organic matter and microorganism and it has been found to have beneficial effects when used as a total or partial substitute for mineral fertilizer. Some studies show that vermicomposting leachates or vermicompost water-extracts used as substrate amendments or foliar sprays can also promote plant growth. Positive effects of vermicompost include stimulated seed germination, growth, plant flowering, increased number and biomass of flowers, fruit yield, and nutritional quality of some vegetable crops. Results may depend on the characteristics of vermicompost, the plant species used, and the cultivation conditions.

**A**



| Assay Name | Result | Units | Desired Level | Commentary |
|---|---|---|---|---|
| **Organism Biomass Data** | | | | |
| Dry Weight | 0.38 | N/A | 0.20 to 0.80 | Within normal moisture levels. |
| Active Fungi | 0.00 | µg/g | > 3.00 | No fungal activity, foods may be required. - |
| Total Fungi | 446.41 | µg/g | > 300.00 | Good fungal biomass. - Fairly good fungal diversity, hyphal diameter: 1.5 to 7u |
| Hyphal Diameter | 2.90 | µm | > 2.50 | Good balance of fungi. - |
| Active Bacteria | 80.84 | µg/g | > 3.00 | Bacterial activity within normal levels. |
| Total Bacteria | 1,989.34 | µg/g | > 300.00 | Good bacterial biomass. - |
| Actinobacteria | 33.18 | µg/g | < 50.00 | |
| **Organism Biomass Ratios** | | | | |
| TF:TB | 0.22 | | 0.01 to 10.00 | Balanced fungal and bacterial biomass. |
| AF:TF | 0.00 | | < 0.10 | No fungal activity relative to total biomass, foods may be required. |
| AB:TB | 0.04 | | < 0.10 | Good bacterial activity. |
| AF:AB | 0.00 | | 0.01 to 10.00 | Bacterial dominated, becoming more bacterial. |
| **Protozoa (Protists)** | | | | |
| Flagellates | 15,196.90 | number/g | > 10,000.00 | High ciliate numbers indicate possible anaerobic conditions. |
| Amoebae | 112,603.35 | number/g | > 10,000.00 | |
| Ciliates | 3,660.57 | number/g | < 1278.00 | |
| Nitrogen Cycling Potential | 200+ | lbs/acre | | Nitrogen levels dependent on plant needs. Estimated availability over a 3 month period |
| **Nematodes** | | | | |
| Nematodes | 10.60 | number/g | > 10.00 | Good numbers, but lacking diversity. |
| Bacteral | 10.60 | number/g | | |
| Fungal | 0.00 | number/g | | |
| Fungal/Root | 0.00 | number/g | | |
| Predatory | 0.00 | number/g | | |
| Root | 0.00 | number/g | | |
| **Miscellaneous Testing** | | | | |
| E.coli | Not Ordered | CFU/g | < 800.00 | For most areas, the maximum E.coli CFU/g is 800 - 1000. Please check your local regulations for more information. - |
| pH | Not Ordered | | | |
| Electrical Conductivity | Not Ordered | µS/cm | < 1000.00 | |

**B**



**Chemical, biological and physical mechanisms by which vermicompost may directly or indirectly influence plant growth and development (Reprinted from Lazcano and Dominguez, 2011. The use of vermicompost in sustainable agriculture: impact on plant growth and soil fertility. Soil nutrients, 10(1-23), p.187).**

**C**

| Ph | Organic matter (%) | Organic Carbon (%) | Electric Conductivity (uS/cm) | C:N | N total (%) | Phosphorus (%) | Potassium (%) | Calcium (%) | Magnesium (%) | Iron (mg/Kg) | Copper (mg/kg) | Zinc (mg/kg) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 7.0-7.8 | 55-75 | 20-40 | 0.6-2.8 | 9-13 | 1.5 -2.5 | 1.2-2.0 | 0.3-0.7 | 2.0-3.0 | 0.5-1.5 | 6000-7000 | 20-40 | 100-150 |

**Figure 2 Vermicompost properties**

Vermicompost influences plant growth directly or indirectly through different chemical, physical and biological mechanisms (Figure 2 C). It constitutes a source of plant macro- and micronutrients. It supplies plant growth hormones as there is strong evidence of hormonal activity associated with the earthworms and with the humic substances present in vermicompost. Vermicompost has also been found to have a wide range of indirect effects on plant growth such as the mitigation or suppression of plant diseases, insect pests, plant parasitic nematodes and fungal diseases (Lazcano and Dominguez 2011). Disease suppression by vermicompost may be attributed to either direct suppression of pathogens or to the induction of systemic resistance in the plant. Because vermicompost increases microbial biomass in soil and changes the diversity and abundance of soil fauna (Gunadi et al 2002; Arancon et al 2006), a broader range of organisms may act as biocontrol agents. Vermicompost promotes the establishment of a specific microbial community in the rhizosphere different from that of plants supplemented with mineral fertilizers or other types of organic fertilizers such as manure (Aira et al 2010). Because of their favorable microbial composition and the beneficial action of worms, vermicompost is a very suitable substrate for agronomic purposes. Characteristics and quality of the vermicompost produced by BIDA® System are described in Figure 2.

## PRELIMINARY ECONOMIC ANALYSIS IN DAIRIES

### 1) Increase of profitability from selling worm casting

The vermicompost produced from the BIDA® System can be sold as a soil and plant amendment. In addition, worm castings can produce liquid soil amendments and vermicompost teas that can be applied as a foliar spray, or through existing fertilization and irrigation systems.

**Table 8 Economics of BIDA® System**

| Project Economics | Income per cow per year |
|---|---|
| **Incomes** | **$85 - $205** |
| Worms casting | $70 - $100 |
| Carbon Credits | $15 - $105* |
| **Costs** | **$200 - $230** |
| Construction | $160 - $190 |
| Operation | $40 |
| **Net Cost** | **$0 - 145** |

**The difference is due to different prices of Voluntary or Compliance Carbon market.**

### 2) Increase of profitability from selling carbon offsets.

In the current effort to mitigate global climate change, a special emphasis is given to the agricultural sector which is the second largest emitter of human GHG emissions (EPA 2019) and is characterized by a large potential for GHG emission reduction. In the cap and trade programs, the carbon offsets result from the GHG emission reduction of projects that are quantified and verified using standardized methodologies and sold in the carbon market. A GHG emission reduction of a project or activity is the quantification of the change compared to GHG emission levels of baseline or status quo. Carbon offsets obtained using the BIDA® System can be sold in the carbon market and represent an additional income for farmers or project developers.

In summary, the project economics show (Table 4) how an income of $85-$205 per cow per year from the carbon market and by selling vermicompost will mostly cover the 230$ to $200 per head costs needed to build and operate the BIDA® System.

In summary, the project economics show (Table 4) how an income of $85-$205 per cow per year from the carbon market and by selling vermicompost will mostly cover the $200 to $230 costs needed to build and operate the BIDA® System.

## Path forward for gathering and solidifying evidence to demonstrate BIDA® System benefits.

After decades of efforts to optimize the success of removal of contaminants, the inclusion of technologies, the efficiency of operation, and decreasing costs, BioFiltro priority is now to quantify co-benefit provided by the BIDA® System for the different applications in addition to the ability to reduce organic waste from water. Specifically, quantify GHG emissions reductions, especially but not exclusively on dairy farms; C sequestration, soil health and fertilization resulting from vermicompost addition to soils; N pollution reduction, and economic benefits from carbon, vermicompost, and possibly water markets.

## References

Aira, M., Monroy, F., Dominguez, J. (2007). Earthworms strongly modify microbial biomass and activity triggering enzymatic activities during vermicomposting independently of the application rates of pig slurry. Sci. Total Environ. 385, 252–261.

Arancon, N.Q., Edwards, C.A. Bierman, P. (2006). Influences of vermicomposts on field strawberries: Part 2. Effects on soil microbiological and chemical properties. Bioresource Technology, 97, 831-840.

Bernet, N., Delgenes, N., Akunna, J.C., Delgenes, J.P., Moletta, R. (2000). Combined anaerobic-aerobic SBR for the treatment of piggery wastewater. *Water Research*, *34*(2), pp.611-619.

California, Air Resources Board (2018) California Greenhouse Gas Emissions for 2000 to 2016. Available at https://www.arb.ca.gov/cc/inventory/data/data.htm (accessed March 25, 2019).

California, Air Resources Board (2019) CCI Quantification, Benefits, and Reporting Materials. Available at (https://ww2.arb.ca.gov/resources/documents/cci-quantification-benefits-and-reporting-materials (accessed March 25, 2019).

Carr, L., Grover, R., Smith, B., Richard, T., Halbach, T. (1995). Commercial and on-farm production and marketing of animal waste compost products. In: Steele, K. (Ed.), Animal Waste and the Land-Water Interface. Lewis Publishers, Boca Raton, pp. 485–492

Davison, A.W. and Cape, J.N. (2003). Atmospheric nitrogen compounds—issues related to agricultural systems. *Environment International*, *29*(2-3), pp.181-187.

Diffenbaugh, N.S., Swain, D.L., Touma, D., (2015). Anthropogenic warming has increased drought risk in California. *Proceedings of the National Academy of Sciences*, *112*(13), pp.3931-3936.

Domínguez, J., Aira, M. Gómez Brandón, M. (2010). *Vermicomposting: earthworms enhance the work of microbes*. In: H. Insam, I. Franke-Whittle, and M. Goberna, (Eds.), Microbes at Work: From Wastes to Resources (pp. 93-114). Springer, Berlin Heidelberg,

Drake, H.L. and Horn, M.A. (2007). As the worm turns: the earthworm gut as a transient habitat for soil microbial biomes. *Annu. Rev. Microbiol.*, *61*, pp.169-189.

EIA- US Energy Information Administration (2015). Available at https://www.eia.gov/ (accessed March 25, 2019).

EPA-Environmental Protection Agency (2019a) Draft Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2017.Available at https://www.epa.gov/ghgemissions/inventory-us-greenhouse-gas-emissions-and-sinks. (accessed March 25, 2019).

EPA-Environmental Protection Agency (2017). Air Actions in the San Joaquin Valley — Ozone. www3.epa.gov/region9/air/sjvalley/ (accessed March 25, 2019).

EPA-Environmental Protection Agency (2019b) https://www.epa.gov/sites/production/files/2019-02/documents/trading-policy-memo-2019.pdf (accessed March 25, 2019).

Galloway, J.N., Aber, J.D., Erisman, J.W., Seitzinger, S.P., Howarth, R.W., Cowling, E.B., Cosby, B.J. (2003). The nitrogen cascade. *Bioscience*, *53*(4), pp.341-356.

Gunadi, B., Edwards, C.A., Arancon, N.A. (2002). Changes in trophic structure of soil arthropods after the application of vermicomposts. *European Journal of Soil Biology*, 38, 161-165.

Hodge, A., Stewart, J., Robinson, D., Griffiths, B.S., Fitter, A.H. (2000). Spatial and physical heterogeneity of N supply from soil does not influence N capture by two grass species. *Functional Ecology*, *14*(5), pp.645-653.

Holly, M.A., Larson, R.A., Powell, J.M., Ruark, M.D., Aguirre-Villegas, H. (2017). Greenhouse gas and ammonia emissions from digested and separated dairy manure during storage and after land application. *Agriculture, Ecosystems & Environment*, *239*, pp.410-419.

Hutchison, M.L., Walters, L.D., Avery, S.M., Munro, F., Moore, A. (2005). Analyses of livestock production, waste storage, and pathogen levels and prevalences in farm manures. Appl. Environ. Microb. 71, 1231–1236.

IPCC (2006). Guideline for National GHG Inventories. Eggleston H S, Buendia L, Miwa K, Ngara T, Tanabe K. IGES, Japan. Available at: http://www.ipccnggip.iges.or.jp/-public/2006gl/index.html. (accessed March 25, 2019).

IPCC (2007) AR4 Climate Change 2007: Synthesis Report available at. https://www.ipcc.ch/assessment-report/ar4/ (accessed March 25, 2019).

Lai, E., Hess, M., Mitloehner, F. (2018). Profiling of the microbiome associated with nitrogen removal during vermifiltration of wastewater from a commercial dairy. *Frontiers in microbiology*, *9*, p.1964.

Lara R. Chapter 395, Statutes of (2016). www.leginfo.ca.gov/ pub/15-16/bill/sen/sb_1351-1400/sb_1383_bill_20160919_chaptered.htm (accessed March 25, 2019).

Lazcano, C. and Domínguez, J. (2011). The use of vermicompost in sustainable agriculture: impact on plant growth and soil fertility. Soil nutrients, 10(1-23), p.187.

Lee, H. and Sumner, D., (2018). Dependence on policy revenue poses risks for investments in dairy digesters. California Agriculture, 72(4), pp.226-235.

Leytem, A.B., Bjorneberg, D.L., Al Rotz, C., Moraes, L.E., Kebreab, E., Dungan, R.S., (2018). Ammonia emissions from dairy lagoons in the western US.

Li, X., Xing, M., Yang, J., Lu, Y. (2013). Properties of biofilm in a vermifiltration system for domestic wastewater sludge stabilization. Chem. Eng. J. 223, 932–943.

Li, Y.S., P. Robin, D. Cluzeau, M. Bouché, J.P. Qiu, A. Laplanche, et al. (2008). Vermifiltration as a stage in reuse of swine wastewater: Monitoring methodology on an experimental farm. Ecol. Eng. 32: 301-309.

Lin, H., S.A.W. Diemont, T. Toland, W. Tao, D. Daley, D.L. Johnson. (2013). Vermifiltration Ecological Treatment for the Re-Use of Food Waste Digestate. Water Environ. Res. 85: 2184- 2193.

Liu, J., Lu, Z., Yang, J., Xing, M., Yu, F., Guo, M. (2012). Effect of earthworms on the performance and microbial communities of excess sludge treatment process in vermifilter. Bioresour. Technol. 117, 214–221.

Lubbers, I.M., Van Groenigen, K.J., Fonte, S.J., Six, J., Brussaard, L. and Van Groenigen, J.W., (2013). Greenhouse-gas emissions from soils increased by earthworms. *Nature Climate Change*, *3*(3), p.187.

Luth, P. Robin, P. Germain, M. Lecomte, B. Landrain, Y. Li, et al. (2011). Earthworm effects on gaseous emissions during vermifiltration of pig fresh slurry. Bioresour. Technol. 102: 3679-3686.

Nkoa, R. (2014). Agricultural benefits and environmental risks of soil fertilization with anaerobic digestates: a review. Agronomy for Sustainable Development, 34(2), 473-492.

Owen, J.J. and Silver, W.L. (2015). Greenhouse gas emissions from dairy manure management: a review of field-based studies. *Global change biology*, *21*(2), pp.550-565.

Parks, S.E.; Irving, D.E.; Milham, P.J. (2012). A critical evaluation of on-farm rapid tests for measuring nitrate in leafy vegetables. *Sci. Hort. 134*, 1–6.

Pimentel, D., Berger B., Filiberto D., Newton M., Bet al. (2004). Water Resources: Agricultural and Environmental Issues, *BioScience*, Volume 54, Issue 10, Pages 909–918.

Ravishankara, A.R., Daniel, J.S. Portmann, R.W. (2009). Nitrous oxide (N2O): the dominant ozone-depleting substance emitted in the 21st century. *science*, *326*(5949), pp.123-125.

Rodrigues, M.A., Coutinho, J., Martins, F. (2002). Efficacy and limitations of triticale as nitrogen catch crop in a Mediterranean environment. Eur. J. Agron 17 (3), 155–160.

Wang, L., Guo, F., Zheng, Z., Luo, X., Zhang, J. (2011). Enhancement of rural domestic sewage treatment performance, and assessment of microbial community diversity and structure using tower vermifiltration. Bioresour. Technol. 102, 9462–9470.

Wang, L., Y. Zhang, X. Luo, J. Zhang, Z. Zheng. (2016). Effects of earthworms and substrate on diversity and abundance of denitrifying genes (nirS and nirK) and denitrifying rate during rural domestic wastewater treatment. Bioresour. Technol. 212: 174-181.

Wang, L., Z. Guo, Y. Che, F. Yang, J. Chao, Y. Gao, et al. (2014). The effect of vermifiltration height and wet:dry time ratio on nutrient removal performance and biological features, and their influence on nutrient removal efficiencies. Ecol. Eng. 71: 165-172.

Wrap (2016). DC-Agri field experiments for quality digestate and compost in agriculture. Wrap UK Final report 2016.

Xing, M., X. Li , J. Yang. (2015). Treatment performance of small-scale vermifilter for domestic wastewater and its relationship to earthworm growth, reproduction and enzymatic activity. Afr J Biotechnol 9: 7513-7520.

Zhao, L., Wang, Y., Yang, J., Xing, M., Li, X., Yi, D., et al. (2010). Earthworm-microorganism interactions: a strategy to stabilize domestic wastewater sludge. Water Res. 44, 2572–2

# APPENDIX

## CALCULATION OF GHG EMISSION REDUCTION OF THE BIDA® SYSTEM AT FANELLI DAIRY

This Appendix describes in detail the preliminary assessment of the greenhouse gas (GHG) emission reduction obtained by the BioFiltro BIDA® System at Fanelli Dairy in Merced County, CA. GHG emission reduction was obtained using two independent approaches: using the California Department of Food and Agriculture (CDFA) calculation tool and using published GHG emissions measurements data.

In this typical midsize dairy farm in CA (Fanelli Dairy, details are described in Lai et al 2018) manure was managed with the flush system and anaerobic lagoon. The free-stall barns were flushed daily 3 times for 6 minutes to remove manure from the barn floor. Flush water was stored in an uncovered anaerobic lagoon with approximately 5.7 million liters of holding capacity. A portion of this water was recycled during the next flushing period and eventually, all water was applied to surrounding cropland as fertilizer. The BIDA® System was designed to support 150 cows of the total 1,105 on the farm. To calculate the GHG emissions per cow, a population of 118 dairy cows was used, considering the proportion between excretion produced by dairy and non-dairy cows at the farm. The BIDA® System treated lagoon water before it was recycled back to the free stall barns as flush water, and thus was not treating directly flushed water. The BIDA® System bed consisted of a concrete basin (49 m x 11 m x 1.5 m) filled with a 30 cm bottom layer of river cobble to improve drainage, topped with a 1.2 m layer of semi-sterile woodchips made from hearts of Douglas fir, White fir, and Ponderosa pine. The surface layer of woodchips was seeded with 300 kg of earthworms (*Eisenia fetida*). Twenty peripheral PVC exhaust pipes (20 cm diameter) allowed air exchange between the bottom layer and ambient air to maintain aerobic conditions in the BIDA® System profile.

Net GHG emission reduction was calculated by subtracting BIDA® System GHG emissions from baseline GHG emissions from the anaerobic lagoon. Both baseline and BIDA® System GHG emissions were calculated only for the number of cows supported by the BIDA® System. GHG emissions resulting from fossil fuels were not included.

### 1) GHG emission reduction estimated using the CDFA Calculation Tool

The CDFA Alternative Manure Management Program (AMMP) Quantification Methodology is adapted from the California Air Resources Board (CARB) 2014 Compliance Offset Protocol for Livestock Projects adopted by CARB in 2011 and updated in 2014 (CARB 2016). This methodology accounts for the net GHG benefit based on a calculation of the avoided methane ($CH_4$) emissions from anaerobic manure decomposition after the adoption of a livestock digester in order to generate offset credits for use in the Cap-and-Trade Program. While the focus of the Protocol is the installation of a digester, the equations used to calculate emissions are broadly applicable to any livestock operation with anaerobic manure management systems. The CARB protocol also contains equations for quantifying $CH_4$ emissions from a variety of manure management practices and for quantifying fossil carbon dioxide ($CO_2$) emissions associated with manure management. These equations form the basis of this AMMP Quantification Tool (CARB 2019).

Practices for treating/storing manure that did not have corresponding $CH_4$ conversion factors (MCFs), as for the BIDA® System, factors for closely related practice were selected based on the definitions in the Benefits Calculator Tool. Co-benefits such as soil carbon sequestration from compost additions to soils are included and followed the Methodology for Soil Health and Conservation (CARB 21018).

Methane production from manure depends on the quantity of volatile solids (VS), i.e. the biodegradable organic material in manure that can decompose anaerobically, air temperature, and the retention time of manure during treatment and storage. This methodology combines project-specific data with default factors to establish both a baseline scenario and a project scenario. Baseline scenario methane emissions ($BE_{CH4}$) represent the emissions within the Project Boundary that would have occurred without adoption of alternative manure management practices for the previous 12 months of dairy operation. To calculate project BIDA® System emissions using the CDFA calculator tool, the BIDA® System was considered as a high-efficiency separator capable of retaining most of the VS (83% obtained by monthly measurements of TSS between 2015-2017 in Table 2) before entering the anaerobic lagoon. In addition, the VS retained by the BIDA® System were considered aerobically composted.

Methane is produced from both anaerobic storage (lagoon) and from predominantly aerobic systems where separated solids are collected.

$$BE_{CH4} = BE_{CH4AS} + BE_{CH4nonAS} \qquad \text{(Equation 1)}$$

$CH_4$ emission is estimated for each cow population category, month, and storage/treatment system as:

$$BE_{CH4AS} = f(VS \times B_0 \times GWP \times MCF \times MS) \qquad \text{(Equation 2)}$$

Where the proportion of the VS excreted during the month, adjusted for the non-degraded VS stored from previous months that are available for conversion to $CH_{4,}$ is corrected by the effect of temperature by the van't Hoff-Arrhenius factor f. $B_O$ is the maximum potential $CH_4$ production of the cow category, *MS* is the fraction of waste going through the specific storage/treatment system, and *MCF* is the VS-to-$CH_4$ conversion efficiency for each storage/treatment used for non-anaerobic practices. *GWP* is the warming potential of $CH_4$ compared to $CO_2$. Current IPCC GWP values of 25 for $CH_4$ and 298 for $N_2O$ were used in these calculations. For the annual non-anaerobic $CH_4$ emissions estimate, the same function in Equation 2 was applied on the annual sum of daily excreted VS.

Values and parameters set as inputs and used in the tool are shown in Table S1 A, B, and C

**A**

| Livestock population | N animals | Bo (m$^3$ $CH_4$/kg VS) | VS (kg/cow/day) | VS deposited on land (%) |
|---|---|---|---|---|
| Lactating Dairy Cows (freestall) | 88 | 0.24 | 7.7588 | 20 |
| Dry cows | 14 | 0.24 | 3.80304 | 70 |
| Heifers (on feed) | 48 | 0.17 | 3.43508 | 70 |
| Bull | 1 | | | 70 |

**B**

| Treatment Practice MCF (16.5 °C ) | MCF Values |
|---|---|
| Anaerobic lagoon | Calculated using the van't Hoff-Arrhenius factor f *0.80 on the available Vs |
| Solid storage | 4% |
| Dry lot | 1.5% |
| Liquid slurry (with natural crust cover) | 19% |
| Composting -passive windrow /BIDA® System vermicomposting | 1% |

**C**

| | Solid separation | Type | Separation | Fraction sent to anaerobic lagoon |
|---|---|---|---|---|
| Baseline | Solid separator | Vibrating screen | 15% | |
| | Fraction sent to anaerobic lagoon | | | 68 % |
| Project-BIDA® System | Solid separation | Vibrating Screen | 15% | |
| | Secondary solid separator | Roller drum | 25% | |
| | Third separator: BIDA® System | Biofilter | 83% | |
| | Fraction sent to anaerobic lagoon | | | 9 % |

**Tables S1 A,B,C. Specific values for the farm and storage/treatment systems. The factors describing the fraction of the potential $CH_4$ production emitted by storage type are set for the annual average temperature of 16.5 °C.**

Thus, the tool calculates monthly temperature-corrected $CH_4$ emissions. Temperatures are set at the county level and derived from monthly values from NOAA (NCDC) Western Regional Climate Center

For the dairy cows at this farm, 20% of excrement is left on the ground by default, and 12% is removed by a first separator. These separated VS are kept in a liquid slurry with a natural cover while the rest goes to the anaerobic lagoon. Separated solids and residual solids from previous months produce $CH_4$ proportionally to the f values (Table S1 B). For the other livestock categories, the initial proportion of manure left on the ground is 70%. In total, estimated baseline annual emissions are 488 $tCO_2eq$ yr$^{-1}$, and additional 32.5 non-anaerobic $CH_4$ emissions from the separated materials.

For project conditions, the first solid separation and storage methods are unchanged. However, a second separator filters 25% of the remaining VS, and then the BIDA® System retains 83% of the remaining VS. In this scenario, only 9% of VS remain in wastewater after treatment (Table S1 C). Because the VS separated by the BIDA® System are composted by worms in predominantly aerobic conditions, MCF for the BIDA® System has been set to 1%, as for the most similar storage treatment method: the composting in passive windrows (with infrequent turning for mixing and aeration).

Resulting BIDA® System related $CH_4$ anaerobic emissions are 62 $tCO_2eq$ yr$^{-1}$, and 43.4 $tCO_2eq$ yr$^{-1}$ non-anaerobic $CH_4$ emissions from the separated materials. Thus, the BIDA® System reduced 87% $CH_4$ anaerobic emissions and increased 33% the smaller $CH_4$ emissions from the non-anaerobic storage of the separated solids.

## 2) GHG emission reduction estimated using GHG flux measurement data

A second approach to estimate GHG emission from the BIDA® System was based on the GHG fluxes measured in summer 2015 published in Lai et al 2018 (Table 3 and 4). $CO_2$, $CH_4$, $N_2O$, and $NH_3$ emissions were measured over 24-hour periods from the lagoon (LAG); in the water entering the BIDA® System (INF); in the treated water exiting the BIDA® System (EFF); and at the surface (TOP) and the bottom gravel layer (BOT) of the BIDA® vermifiltration system while operating. The sum of the EFF, TOP, and BOT equal the vermifilters BIDA® System emissions and is compared with emission prior to treatment or in the lagoon.

The daily fluxes measured during the field campaign were scaled up to monthly sums. Monthly $CH_4$ fluxes were corrected for temperature effects using the same Arrhenius factor used in the CDFA quantification tool. The monthly flux of July was divided by the *f* factor of July to obtain a $CH_4$ basal flux. The basal flux was then multiplied for the *f* factor of each month and monthly fluxes were summed to obtain the annual flux. $CO_2$ monthly fluxes were scaled up to annual value using the same approach as for $CH_4$, but using a $Q_{10}$ function and the typical $Q_{10}$ values of 2 (van't Hoff, 1898). Temperature values were the same used for the CDFA tool approach.

This second estimate of BIDA® System related GHG emission was 42 $tCO_2eq$ yr$^{-1}$ (Table S2), much less than measured with the AMMP tool. It was comprised of 37% of $N_2O$ fluxes, 51% $CO_2$ fluxes, and 12% $CH_4$. $N_2O$ fluxes were very small but had a large effect due to a warming potential 298 times the warming potential of $CO_2$. Even if a single measurement campaign could be insufficient to quantify annual GHG emissions, the measurements were made during the warmest and most active month and show insignificant emissions from the BIDA® System. Even if measured GHG emission from the BIDA® System (on a volume of 50,000 liters of wastewater) were slightly higher compared to fluxes from the same amount of water in the lagoon (Table 3) the authors explained how they sampled only superficial, less $CH_4$ rich lagoon water. Moreover, slightly higher fluxes from a system requiring four hours to treat wastewater like the BIDA® System are still insignificant compared to the fluxes generated by the entire lagoon, 25 times larger than the BIDA® System and storing wastewater for months.

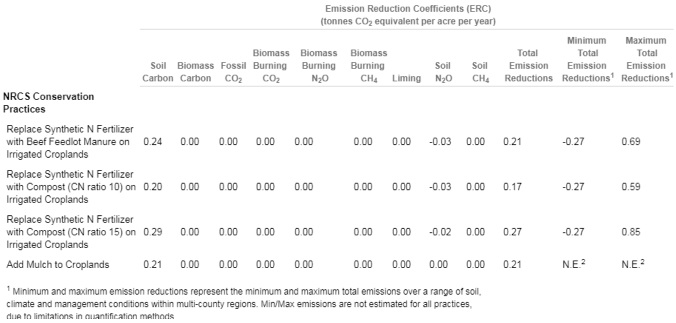
The capacity of the BIDA® System to reduce indirect $N_2O$ emitted from the soil of crops, which are excluded by the CDFA Calculator Tool, was also assessed. Baseline $N_2O$ emissions were estimated using the IPCC guidelines (2006). Emissions included indirect $N_2O$ emissions resulting from the application of dairy manure to cropped soils. Considering the 80% efficiency of the BIDA® System in reducing the N load of wastewater, this component decreased from 1.1 $tCO_2eq\ yr^{-1}$ per cow to 0.2 $tCO_2eq\ yr^{-1}$per year, adding to the total GHG mitigation benefit.

**Table S2 GHG fluxes**

| GHG | LAGOON | | BIDA® System (internal+effluent) | |
|---|---|---|---|---|
| | $t\ yr^{-1}$ | $t\ CO_2eq\ yr^{-1}$ | $t\ yr^{-1}$ | $t\ CO_2eq\ yr^{-1}$ |
| $N_2O$ | 0.00008 | 0.03 | 0.053 | 15.70 |
| $CO_2$ | 15.60 | 15.60 | 21.45 | 21.45 |
| $CH_4$ | 0.07 | 1.84 | 0.20 | 5.05 |
| TOTAL | | **17.47** | | **42.20** |

**Fluxes of the different components are calculated over the same volume (volume of water treated daily by the BIDA® System. However, the lagoon volume is 100 times larger than the volume of the BIDA® System.**

**Table S3: Emission Reduction Coefficients from http://www.comet-planner.com/**



Emission Reduction Coefficients (ERC)
(tonnes $CO_2$ equivalent per acre per year)

| NRCS Conservation Practices | Soil Carbon | Biomass Carbon | Fossil $CO_2$ | Biomass Burning $CO_2$ | Biomass Burning $N_2O$ | Biomass Burning $CH_4$ | Liming | Soil $N_2O$ | Soil $CH_4$ | Total Emission Reductions | Minimum Total Emission Reductions[1] | Maximum Total Emission Reductions[1] |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Replace Synthetic N Fertilizer with Beef Feedlot Manure on Irrigated Croplands | 0.24 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | -0.03 | 0.00 | 0.21 | -0.27 | 0.69 |
| Replace Synthetic N Fertilizer with Compost (CN ratio 10) on Irrigated Croplands | 0.20 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | -0.03 | 0.00 | 0.17 | -0.27 | 0.59 |
| Replace Synthetic N Fertilizer with Compost (CN ratio 15) on Irrigated Croplands | 0.29 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | -0.02 | 0.00 | 0.27 | -0.27 | 0.85 |
| Add Mulch to Croplands | 0.21 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.21 | N.E.[2] | N.E.[2] |

[1] Minimum and maximum emission reductions represent the minimum and maximum total emissions over a range of soil, climate and management conditions within multi-county regions. Min/Max emissions are not estimated for all practices, due to limitations in quantification methods

The AMMP quantification tool includes the quantification of the co-benefits derived by composting the separated solids. It assumes that the compost produced annually by diverting organic matter from waste stream (*compost*) is applied to improve soils outside the project area. The benefits are expressed as acres of application. The application area is calculated as:

*Application area = compost * Ag use/AR* (Equation 3)

Where *compost* is the quantity of compost produced (dry ton); *Ag use* is the percent of compost used (assumed 100%); *AR* is the application rate (4.65 dry tons per acre).

The BIDA® System decomposes the wood chips and the retained VS produces annually 0.47 $m^3$ per $m^2$ of system per year. At Fanelli Dairy, the 540 $m^2$ BIDA® System produces 114.8 dry tons of vermicompost (using a dry vermicompost density of 450 kg $m^3$ of and the same factor to transform wet compost to dry compost used by the tool), a quantity sufficient to amend 6.6 ha (16.3 acres) of soil.

To further assess the benefits of using vermicompost as soil amendment we used the Comet Planner tool (http://www.comet-planner.com/), an evaluation tool designed to evaluate potential carbon sequestration from adopting conservation practices, such as soil additions of compost or mulch. In this tool, the emissions reduction is calculated using emission reduction coefficient (*ERC*) relative to the conservation practice. The emissions reduction coefficient represents the average impact of a conservation practice compared to baseline conditions, over a range of soils, climate and cropland management within multi-county regions (USDA-NRCS 2006).

*Emission reduction = Area * ERC* (Equation 5)

The conservation practices selected were to use (vermi)compost to replace synthetic fertilizers on irrigated crops and to use the vermicompost as mulch. Table S5 lists the emission reduction coefficients used. Replacing synthetic fertilizer with BIDA® System produced vermicompost (C:N ratio 15) on irrigated croplands, for the area calculated above, results in an additional GHG emission reduction of 4.4 $tCO_2eq\ yr^{-1}$, and, 3.4 $tCO_2eq\ yr^{-1}$ if the vermicompost is used as mulch. This includes both the effects of C sequestration in soils and the release of $N_2O$ from the compost.

## REFERENCES

CARB (2016) Compliance Offset Protocol Livestock Projects. https://www.arb.ca.gov/cc/capandtrade/protocols/livestock/livestock.htm. Accessed March 25, 2019.

CARB (2019) CCI Quantification, Benefits, and Reporting Materials, Available at https://ww2.arb.ca.gov/resources/documents/cci-quantification-benefits-and-reporting-materials. Accessed March 25, 2019.

CARB (2018) Co-benefit Assessment Methodology for Soil Health and Conservation. https://www.arb.ca.gov/cc/capandtrade/auctionproceeds- / final_soil_am.pdf. Accessed March 25, 2019.

IPCC (2006) IPCC Guideline for National Greenhouse Gas Inventories. Eggleston H S, Buendia L, Miwa K, Ngara T, Tanabe K. IGES, Japan. Available at: http://www.ipcc-nggip.iges.or.jp/public/2006gl/index.html. Accessed March 25, 2019.

Lai, E., Hess, M. and Mitloehner, F., (2018). Profiling of the microbiome associated with nitrogen removal during vermifiltration of wastewater from a commercial dairy. Frontiers in microbiology, 9, p.1964.

Van't Hoff, J. H. (1898). Lectures on theoretical and physical chemistry. In Chemical Dynamics Part I (pp. 224–229). London: Edward Arnold