Solar drying in the vineyard: a sustainable technology for the recovery of nutrients from winery organic waste

F. X. Prenafeta-Boldú, L. Burgos, J. Noguerol, M. Mercader, J. Soler and B. Fernández

ABSTRACT

The present study describes a pilot-scale experimental validation of a forced-convection greenhouse solar dryer, combined with a biofilter, for controlled atmospheric emissions. This set-up was applied to the dewatering of sewage sludge from a biological plant that treated process wastewater in a commercial Mediterranean winery. Experiments were performed after the harvest, from September onwards, during the peak generation of sludge. The average drying rate during the first 10 days of operation ranged from 1.17 to 2.24 kg m^{-2} d^{-1}, depending on the measurement method, during which the water content of the sludge was reduced from 90% down to 67%. Biofiltration was quite inefficient against greenhouse gases (methane and dinitrous oxide), and direct emissions during the drying process were on average 57 g CO_2-eq m^{-2} d^{-1}. Ammonia and volatile organic compounds were removed with average efficiencies of 71% and 35%, but ammonia losses through volatilization represented less than 2% of the initial nitrogen content. The sludge was dried further during November, to the lowest possible water content of 14%. Both the intermediate and final sludge dried materials were characterized for their agronomical value as organic fertilizers.

Key words | bioeconomy, organic fertilizers, organic waste, sewage sludge, solar drying

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HIGHLIGHTS

- Solar drying successfully stabilized winery sewage sludge.
- Performance in autumn conditions, after vintage, was satisfactory.
- Biofiltration effectively minimized the emission of ammonia and VOC.
- The dried material complied with quality parameters for organic fertilizers.

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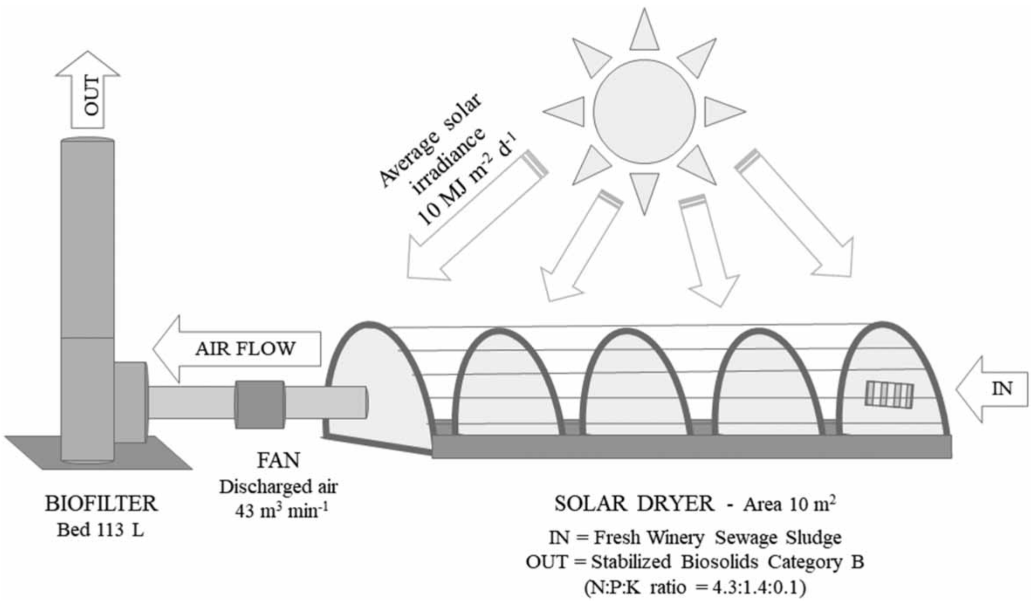
GRAPHICAL ABSTRACT



INTRODUCTION

Wine is generally regarded as a traditional handcrafted product, with strong cultural implications and a reputation as a beverage that contributes to the preservation of historical landscapes. However, in the Mediterranean, vines are often cropped on exposed alkaline soils that contain very little organic matter and are highly degraded either because of steep slopes or by land-levelling works. Such environmental and management conditions make these vineyards very vulnerable to erosive processes and to nutrient runoff (Ramos & Martínez-Casasnovas 2006b). Further, due to their potential to reduce the cost of fertilisation, the use of organic fertilizers comes with several benefits concerning the preservation of soil quality. The application of composted cattle manure to vines in a winery has been shown to significantly increase water infiltration rates in the soil, thus reducing runoff and preventing sediment losses (Ramos & Martínez-Casasnovas 2006a). However, manure might not be readily available in extensive vineyard areas, and must be subjected to biological stabilization and hygienization processes, such as the thermophilic temperatures achieved during composting, in order to prevent microbial contamination with faecal pathogens (Moral *et al.* 2009).

The wine manufacturing process is also an important source of organic waste that could be valorized as agronomical fertilizers for vine crops. A survey of the winery sector in Spain, the largest producer in the European Union, concluded that for every litre of bottled wine roughly 0.6 kg of waste is produced, and that 80–85% of this waste is organic (Ruggieri *et al.* 2009). Grape pomace and lees are the main organic waste products (76%), but these materials are currently valorized as by-products, while stalks (12%) and dewatered sewage sludge (12%) are still being incinerated or disposed of in landfills. Another specific problem related to winery organic waste is that it is primarily produced shortly after the grape harvest and lasts for a brief period between September and October, and so intensive treatment technologies are not economically feasible for such short operational periods. The co-composting of dewatered sludge and stalks has been proposed as a suitable valorization option with several economic and environmental benefits because of the substitution of mineral fertilizers by in-house produced compost. Yet, the dewatering of sewage sludge down to the optimal ranges for composting (below 60% in water content) still requires complex energy-demanding dewatering systems (Christensen *et al.* 2015).

Extensive reviews have been published on the use of solar drying techniques for the dewatering of urban sewage sludge (Bennamoun *et al.* 2013; Pirasteh *et al.* 2014), but using them to treat agricultural organic wastes is still relatively rare. A recent study has demonstrated the viability of combined solar greenhouse drying and composting to produce organic fertilizer from olive mill wastewater (Galliou et al. 2018). We have also tested the feasibility of drying pig slurries in a greenhouse, combined with acidification and an air biofiltration system for controlling gaseous emissions (Prenafeta-Boldú et al. 2020). Solar driers have also been applied in vineyards to produce raisins (Jairaj et al. 2009). Yet, even though solar energy is an abundant resource in most wine-producing areas, the literature on the technical viability of applying solar driers for the treatment and valorization of winery organic waste is lacking. The benefits associated with solar drying include the need for only relatively simple infrastructure, a low carbon footprint, microbial hygienization, and improvement of the agronomical value of the dried products as organic fertilizers. On the downside, depending on the climatic conditions, solar driers might require a large area and cause undesired gaseous emissions.

In this study, we describe a pilot-scale application of a greenhouse-based solar drier with forced aeration for the treatment of activated sludge produced by a wastewater treatment plant from a commercial winery. Assays were performed during the sludge production peak after harvest, during autumn, under Mediterranean conditions. A biofilter was also implemented in order to minimize the potential gaseous emissions associated with the thermal and biological processes.

METHODS

Experimental setup

The pilot-scale solar drier was installed at the wastewater plant (41°20′55.20″N; 1°39′37.80″E) of the winery Bodegas Torres (Vilafranca del Penedès, Catalonia, Spain). The drier was composed of a greenhouse equipped with a forced aeration system (66 W; 1 m^3 min^{-1} nominal flow) and a biofilter for the treatment of emitted gases (Figure 1). The greenhouse was designed to have a Quonsep shape (LWH: 8.4 × 1.2 × 0.7 m). A structural frame of PVC tubes (32 Ø mm) supported a 200 μm low density polyethylene film greenhouse cover (PET) sheet, with a 400 μm PET sheet at the base for sludge containment. The greenhouse was thermally isolated from the concrete floor by a geotextile cover. Indoor and outdoor temperature (T) and relative humidity (RH), as well as the weight of the sludge sample inside the greenhouse, were monitored online. The extracted air was forced into a biofilter packed with a mixture of ripe compost and pine bark (1:5 mass ratio; 113 L total volume), which was encased in a PVC tube (diameter 31.5 cm).

Experimental runs were performed during autumn in 2018. The total amount of initial sewage sludge and the final dewatered product was weighed in a load cell. Samples from these materials were taken in triplicate for subsequent physicochemical analysis in the laboratory. Gas samples from the biofilter inlet and outlet were taken throughout the operational period (four samples) for the analysis of volatile compounds in the laboratory, within 48 hours. For volatile inorganic compounds (VIC), gas samples were collected by means of a calibrated sampling pump (flow of 1 L min^{-1}) and stored in a 3 L volume gas sampling bag (SamplePro FlexFilm, SKC Ltd., UK). For the analysis of volatile organic compounds (VOC), samples were transferred to pre-evacuated 12.5 mL vials (Labco Ltd., Buckinghamsire, UK).

Meteorological data on the daily temperature (mean, maximum and minimum), RH and solar radiance were obtained from the weather station of La Granada (41°21′58.28″N; 1°43′42.85″E), located about 6 km from the pilot plant. This station belongs to the Meteorological Service of Catalonia and the historical weather parameters are freely accessible for consultation (METEOCAT 2020).

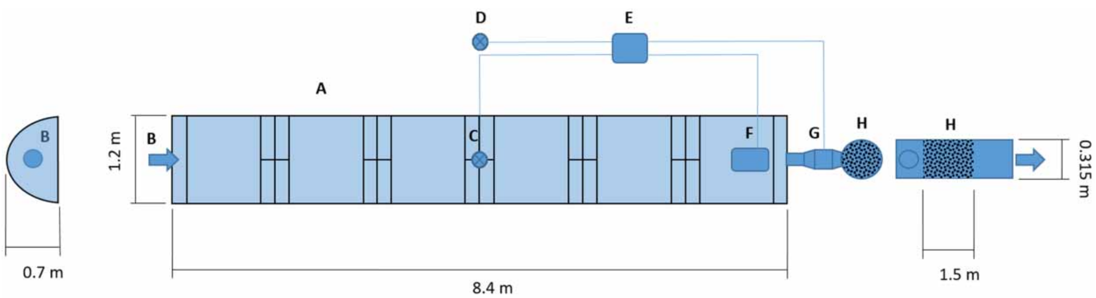


Figure 1 | Schematic representation of the solar drying plant design and dimensions: greenhouse (A), air inlet (B), indoor and outdoor temperature and RH sensors (C and D), data logger and programmable logic controller (E), electronic mass scale (F), suction ventilation system (G), and biofiltration unit (H).

Monitoring and analytical methods

A sensor for monitoring the air temperature and RH (EWHS 284, Eliwell Ibérica, Spain; accuracy: $\pm 0.1\ ^\circ C$ for T and $\pm 3\%$ for RH) was placed in the middle of the greenhouse, and a second identical sensor was installed outdoors. An electronic scale was also installed in the final section of the greenhouse, close to the air exhaust, for the continuous weighing of a 20×30 cm tray containing the sample of sewage sludge (DVP02LC-SL, Delta Electronics Inc., Taiwan; accuracy: ± 2 g). All sensors were connected to a datalogger (DOP-B03E211, Delta Electronics Inc., Taiwan), and data measurements were recorded every 15 min. Air flow was monitored with a portable thermal anemometer (TA4, Airflow Instruments, NJ).

Fresh and dried sludge samples were characterized in terms of pH, electrical conductivity (EC) total and volatile solids (TS, VS), chemical oxygen demand (COD), total Kjeldahl nitrogen (TKN), total ammonia nitrogen (TAN), total phosphorous, (TP), total potassium (TK) and sulphate (SO_4), following the Standard Methods for the Examination of Water and Wastewater (APHA *et al.* 2005). Additionally, the heavy metals copper (Cu) and zinc (Zn) were measured by acid extraction and optical emission spectrometry (U.S. Department of Agriculture 2018).

Different methods were used for the analysis of VIC from collected gas samples. Methane was quantified with a Thermo 2000 gas chromatograph (Thermo Finnigan, CA) equipped with a flame ionization detector, in accordance with the method described by Palatsi *et al.* (2010). The simultaneous analysis of carbon dioxide and nitrous oxide was carried out with an Agilent 7890A gas chromatograph (Agilent Technologies, CA) equipped with an electron capture detector, similar to that described by Martínez-Eixarch *et al.* (2018). The concentration of ammonia at the biofilter inlet and outlet was measured in situ with a portable electrochemical sensor (VRAE, RAE Systems, CA). The concentration of VOC was analyzed by adapting the protocols of NIOSH 1500 (Eller & Cassinelli 1994) and EPA 325 (Boulding 2019) using a Thermo 2000 gas chromatograph.

Calculations

The total content of specific compounds in the loaded sludge and the recovered dried product was calculated as the product of the measured mass and concentration average values. The mass water balance was determined from the cumulative products of the air inlet and outlet absolute humidity values (15 min pace readings) by the measured air flow blown by the ventilator during operation. The air absolute humidity was calculated from the saturation vapour pressure (e_S; kPa) according to the Tetens formula, as a function of the air temperature (T; °C) (Equation (1)).

$$e_S = 0.6108 \exp[17.27T/(T + 237.3)] \quad (1)$$

The vapour pressure (e; kPa) at a given relative humidity (RH; %) was then derived from Equation (2). Absolute humidity (χ; g m^{-3}) was finally determined from vapour pressure (e; kPa) and temperature (T; °C) (Equation (3)).

$$e = (e_s\ RH)/100 \quad (2)$$

$$\chi = 2165e/(T + 273.16) \quad (3)$$

RESULTS AND DISCUSSION

Operational runs

Experiments were performed at the height of the grape harvest season, when the sewage sludge production in the winery was at its maximum. On 17 September 2018, the greenhouse was loaded with 278 kg on a wet basis of a sewage slurry that contained 10.44% total solids (specific area loading 27.8 kg m^{-2}). The internal online weight measurement system contained 5 kg on a wet basis of a sludge sample (equivalent to 83.3 kg m^{-2}). The ventilation was then turned on by an automatic working command between 09:00 and 20:00, if the online RH readings from the sensor in the middle of the greenhouse were above 50%. This automatic ventilation programming ensured that the exhaust air was close to the RH saturation most of the time (condensates were observed continuously at the air exhaust during aeration). The continuous monitoring of T and RH revealed strong daily fluctuations (Figure 2), with minimum and maximum T inside the greenhouse falling between 16.8 °C and 20.1 °C, and 27.1 °C and 48.6 °C, respectively, while the daily minimum and maximum RH ranged between 62.4–78.6% and 93.8–97.9%. Online weight measurements of the sludge sample inside the drier displayed a characteristic 'saw-tooth' decreasing regular pattern, with an average evaporation rate of 2.24 kg m^{-2} d^{-1} (Figure 3). No weight loss occurred during the night, and some mass was even briefly gained during the morning hours, just after the aeration was turned on. This phenomenon could be explained by the mixing of saturated air

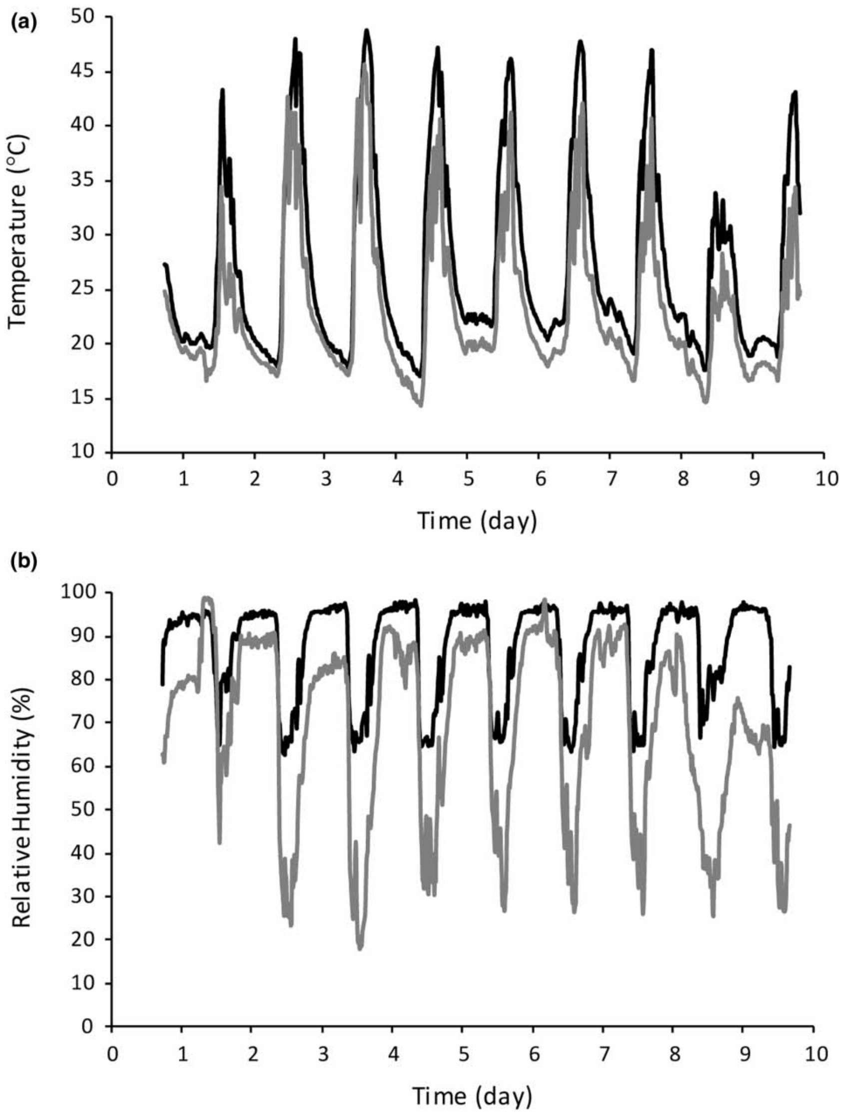


Figure 2 | Time course evolution of the temperature (a) and RH (b) recorded inside (black line) and outside (grey line) the greenhouse during the first period of operation.

inside the greenhouse (the formation of condensates all over the inner greenhouse surfaces was clearly visible at those times), and to the well-known hygroscopic properties of the sewage sludge material (Bougayr *et al.* 2018). The hourly drying rates during the sunny hours of the day ranged from 0.26 to 0.63 $kg\,m^{-2}\,h^{-1}$ ($n > 20$; $r^2 > 0.98$), depending on the meteorological conditions. When comparing the cumulative daily evaporation during this period with that resulting from the difference between the daily maximum and minimum water content, thus excluding sludge rehydration from the global balance, it became apparent that the hygroscopic behaviour of the sludge reduced the process efficiency by about 20%. Hence, the removal of condensates that are formed inside the greenhouse during the cool periods might significantly improve the drying process.

After nine days of operation, the sludge had a dry aspect and, therefore, it was decided to stop the process in order to collect and analyze this material (this first operational run has henceforth been referred to as Period 1). At that point, the ventilation system had been active for 106 hours (46% of the total operational time). The weather during this time was characterized by sunny and mild days, with average daily measurements that, according to a nearby meteorological station, ranged from 18.8–23.2 °C T and 63–92% RH. These T and RH values were in agreement with those

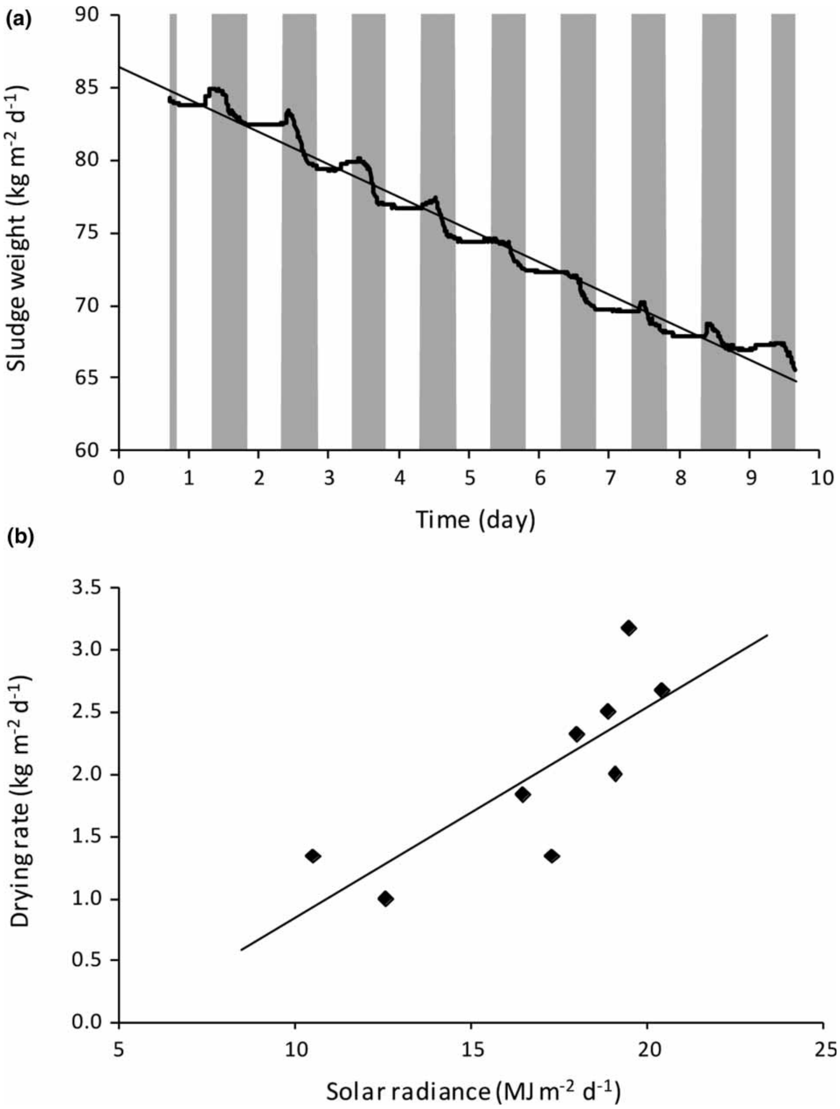


Figure 3 | Time course evolution of the sewage sludge mass sample inside the greenhouse (solid line) and the ventilation intervals (grey bars) during operational Period 1 (a); the correlation line corresponds to an area-specific drying rate of 2.24 kg m^{-2} d^{-1} ($n = 855$; $r^2 = 0.98$). Correlation between the average evaporation rate and the solar irradiance during this same period of operation ($n = 8$; $r^2 = 0.75$), excluding the first and last days of partial operation (b).

obtained from continuous measurements outside the greenhouse, which ranged from 19.7–25.0 °C and 55.5–80.9% (Figure 2). Differences between these two datasets could be explained by microclimatic particularities in the weather station, and the solar drying induced by the presence of constructions and vegetation in the surroundings, proximity to the ground, and so forth. Despite the variability of the meteorological parameters, the amount of water that evaporated every day was fairly correlated with the daily solar radiance, which ranged from 10.5 to 21.1 MJ m^{-2} d^{-1} (Figure 3).

The weight of the collected dried sludge after Period 1 was 129 kg on a wet basis (12.9 kg m^{-2}), which corresponded to a 53.6% mass reduction in relation to the loaded sewage sludge (water loss of 65.4% in terms of the initial water content). Such a decrease was equivalent to an average daily drying rate of 1.66 kg m^{-2} d^{-1}, a value that was lower than the rate measured from the sample in the online weight measurements. This difference could be explained by the higher thermal exchange of the sludge sample on the scale with the surrounding hot air than from the total load on the ground of the greenhouse. Furthermore, a strong humidity gradient was observed along the greenhouse longitudinal axis (the sludge had more moisture close to the air outlet than at the inlet), and the crusty nature of the dried sludge prevented effective drying of the inner core of the coarse aggregates. The amount of evaporated water was also calculated through direct flow measurements of the discharged air ($43 \pm 4\ m^3\ h^{-1}$) and from the cumulative differences in absolute humidity between the greenhouse inlet and outlet air during operation. Such estimate yielded an average daily drying rate of $1.17\ kg\ m^{-2}\ d^{-1}$, which accounted for 70% of the weighed total sludge mass loss. This difference might indicate that some degree of evaporation still occurred when the aeration system was turned off.

In order to investigate the process further under less favourable climatic conditions, it was decided to continue with the solar drying of the partly dehydrated sludge from 7 November onwards (operational Period 2). Until then, the material was stored outdoors, covered from the rain, before it was mixed thoroughly and reintroduced into the greenhouse. A sludge sample of 1.5 kg on a dry basis was also placed on the scale for online weight measurements. The meteorological conditions during Period 2 were characteristic of the end of autumn, with a tendency towards cooler and moister days. The average daily T and RH inside the greenhouse during the drying intervals, when the aeration system was on, ranged between 8.7–17.5 °C and 67.8–90.5%, respectively (Figure 4). These indoor T values were about 5 °C higher than those from outside the greenhouse during the first 10 days of operation – conditions

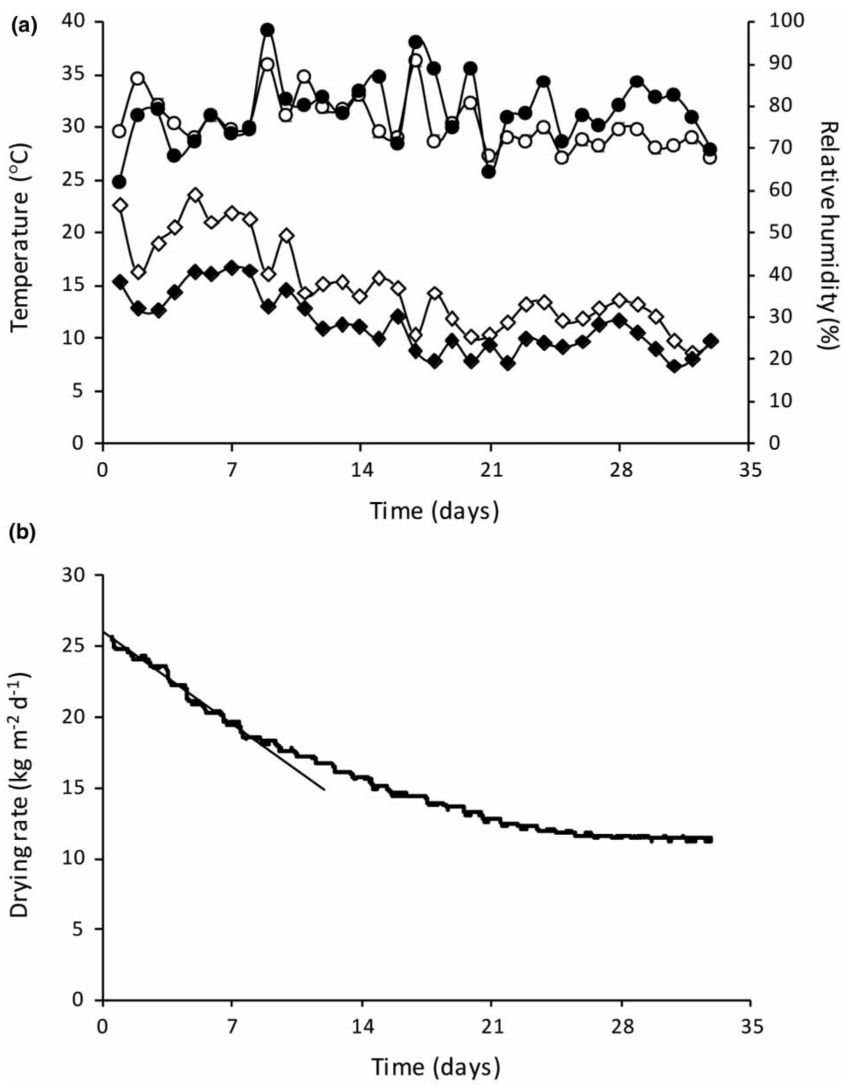


Figure 4 | Average daily temperature (diamonds) and RH (circles) measured inside (empty markers) and outside (solid markers) the greenhouse during the operational Period 2 (a). Time course evolution of the sewage sludge mass sample inside the greenhouse during this same period (b); the linear regression line corresponds to an area-specific drying rate of $0.93\ kg\ m^{-2}\ d^{-1}$ ($n = 701$. $r^2 = 0.98$).

that were sufficient to trigger a specific drying rate of 0.93 $kg\,m^{-2}\,d^{-1}$, as measured by the online scale during the first two weeks of operation. After that, the drying rate tended to decrease along with T values, until the sludge mass stabilized after 25 days of operation (Figure 4). The assay was stopped at day 33 and the collected dried sludge weighed only 29 kg ($2.9\,kg\,m^{-2}$). During operational Period 2 the aeration system worked for 327 hours (41% of the total runtime).

Sludge physicochemical characterization and atmospheric emissions

The physicochemical parameters of the sewage sludge and of the two dried materials obtained after operational Periods 1 and 2 are summarized in Table 1. In general, the content of total solids, organic matter and nutrients in the sewage sludge fell within the range known in the literature for the winery sector (Jin & Kelly 2009; Semitela et al. 2019). During operational Period 1, the moisture content of the sludge was reduced from 89.6% down to 66.8%, which is close to a suitable level for the composting process of below 60% in water content. If sludge co-composting is to be applied and considering that the typical C/N ratio for winery sewage sludge is well below the optimum, supplementation with fibre-rich materials such as stalks and pruning residues will be required (Semitela et al. 2019). Sludge mixed with those structuring materials will also reduce the humidity content within the compostability range.

Table 1 | Physicochemical characteristics of freshly collected sludge from a winery wastewater treatment plant, and of dried materials after 10 days (September; Period 1) and 33 additional days (November; Period 2) of treatment

Parameter	Units	Fresh sludge	(Period 1) Dried sludge	(Period 2) Dried sludge
Total mass	(kg)	278	129	29
pH	–	7.54 (0.02)	7.24 (0.06)	7.67 (0.04)
EC	(μS)	821 (5)	1,476 (2)	2,760 (52.92)
TS	(%)	10.44 (0.25)	33.23 (2.95)	85.70 (0.34)
VS	(%)	8.13 (0.21)	25.17 (2.26)	62.78 (0.52)
COD	($g\,kg^{-1}$)	140.84 (5.85)	464.45 (21.44)	981.14 (39.18)
TKN	($g\,kg^{-1}$)	6.24 (0.19)	16.00 (0.26)	42.97 (1.95)
TAN	($g\,kg^{-1}$)	574 (20)	1,083 (42)	849 (53)
TP	($mg\,kg^{-1}$)	569 (93)	2,496 (110)	6,229 (525)
SO_4^{2-}	($mg\,kg^{-1}$)	55 (5)	267 (6)	1,181 (36)
TK	($mg\,kg^{-1}$)	152 (34)	426 (103)	1,123 (160)
Cu	($mg\,kg^{-1}$)	20	40	96
Zn	($mg\,kg^{-1}$)	90	360	246

Values correspond to the average and standard deviation (in brackets) of three independent samples.

After operational Period 2 the sludge was dried down to just 14.3% humidity content, but could not be reduced further, possibly due to hygroscopic water. The dehydrated material displayed a stabilized behaviour during storage, with no evidence of biological activity either in terms of microbial colonization or off-odour emissions. The recovered nutrients had a composition equivalent to an NPK index of 4.3:1.4:0.1 (mass percentage equivalences to TN phosphorus pentoxide (P_2O_5), and potassium oxide (K_2O); Table 1). Only 3% of TN was recovered in the form of ammonium, and the remaining 97% as organic nitrogen. Concerning the European regulations for fertilizing products (EC 2016), the dried material obtained in this study must be regarded as a solid organic fertilizer (Product Function Category 1A-I). As for the content of copper and zinc, the concentration of these two metals increased with dewatering, up to 96 $mg\text{-}Cu\,kg^{-1}$ and 246 $mg\text{-}Zn\,kg^{-1}$ on a wet basis in the dried material after Period 2 (Table 1). The reason for the presence of copper and zinc in the winery sewage sludge is due to their generalized use as fungicides in the vineyard to control leaf diseases (Brunetto et al. 2014). Yet, if expressed on a dry matter basis, the amount of these metals in the dried material would correspond to 112 $mg\text{-}Cu\,kgTS^{-1}$ and 287 $mg\text{-}Zn\,kgTS^{-1}$ which, according to regulations in the European Union, is below the thresholds of 200 $mg\text{-}Cu\,kgTS^{-1}$ and 600 $mg\text{-}Zn\,kgTS^{-1}$ for their compulsory declaration on the product label (EC 2016).

Gaseous emissions were also monitored during operational Period 1. Concerning VIC, the concentration of methane and dinitrous oxide in the greenhouse air exhaust was very low, with average values of 7 $mg\text{-}C\,m^{-3}$ and 0.7 $mg\text{-}N\,m^{-3}$ (Table 2). If expressed in terms of the greenhouse warming potential and considering the volume of vented air, these emissions would account for 57 g CO_2-eq $m^{-2}\,d^{-1}$. As for carbon dioxide, the average concentration of 540 $mg\text{-}C\,m^{-3}$ was about 2.5-fold higher than that from the ambient air; given its biogenic origin, it should not counted as a net contribution to the greenhouse effect. These measurements indicate that both aerobic (heterotrophs and nitrifying) and anaerobic (methanogenic and denitrifying) microbial populations from the sludge were still active to some extent during the drying process. Ammonia emissions were also observed at an average concentration of 7 $mg\text{-}N\,m^{-3}$ (Table 2), but these nitrogen

Table 2 | Averages and standard deviations of four measurements taken at the operational Period 1 (days 1, 4 and 9) of the concentration of selected contaminants in the air biofilter inlet and outlet, and of the removal efficiency

Parameter	Outdoor air (mg m^{-3})	Biofilter inlet (mg m^{-3})	Biofilter outlet (mg m^{-3})	Removal efficiency (%)
C–CO_2	217	540 (171)	505 (146)	11
C–CH_4	0.90	7.03 (2.37)	6.43 (2.99)	10
N–N_2O	[b]–	0.67 (0.01)	0.68 (0.01)	–1
N–NH_3	–	6.97 (5.01)	2.03 (1.98)	71
Total VOC[a]	–	71 (27)	58 (16)	35

The composition of the outdoor air has also been considered. Standard deviations are shown in brackets.
Notes: [a]Volatile organic compounds. [b]Not detected.

volatilization losses accounted for 1.8% of TN present in the fresh sludge (Table 1). The biofiltration efficiency for methane was very low, and even null for dinitrous oxide, but ammonia was reduced on average by 71%.

The emission of VOC (Period 1) from the greenhouse exhaust was, in average concentration terms, 71 mg m^{-3}. The identified chemical compounds belonged to the ketones (acetone, methylacetone and methylethylacetone), hydrocarbons (p-cymene, toluene, cyclohexane, n-pentane, n-hexane), organohalogens (trichloroethylene) and mercaptans (ethanethiol and 1-propanethiol), and the effectivity of the biofilter in reducing VOCs was, on average, 35%. These VOCs are commonly found in composting off-gases treated by biofiltration, but the relatively low removal efficiency might be explained by the empty bed contact times of the treated air of just 9.4 seconds, which is below the range at above 60 seconds found in similar biofilters packed with pine bark (Prenafeta-Boldú *et al.* 2012).

Treatment efficiency

The mass balance differences between the fresh and dried sludge compounds, as derived by multiplying their concentrations and the total mass of sludge (Table 1), were very difficult to compare. While the amount of bulk components such as TS, TKN and COD were down to 28% lower in the final dried material (after Period 2) compared to the fresh sludge, they were instead up to 54% higher in the intermediate dried material (after Period 1). The main explanation for these deviations might be because of the heterogeneity of the water content, particularly after Period 1, both because of the uneven drying of the sludge core/crust aggregates, and because of the effect of the humidity gradients along the greenhouse. The implementation of a mechanical mixing strategy might therefore significantly improve the process efficiency; in fact, it has been deployed in full-scale solar driers for the treatment of urban sewage sludge (Table 3).

The energy balance between the incident solar radiation and the absorbed enthalpy for the vaporization of water (2.44 MJ kg^{-1} under normal conditions) was also considered. During Period 1 about 14.9 kg m^{-2} of water was effectively evaporated and the cumulative incident solar radiation was 135.5 MJ m^{-2}, so that the energy balance was 27%. The water vaporization enthalpy to radiative energy ratio calculated for Period 1 (September) and Period 2 (November) combined was 19%. The remaining incident solar energy might have been initially reflected by the greenhouse cover, re-emitted to the environment as thermal radiation, and lost through conduction into the external ground/air and via convection through the extracted hot air. The lower apparent solar energy efficiency values from this study in relation to other pilot and large-scale installations (Table 3), could be attributed to the used construction materials (use of thin polyethylene foils versus thick polycarbonate plates), to operational aspects (differences in the aeration regime) and to the climatic conditions (operation under relatively high irradiative conditions).

Concerning the non-solar energy inputs, taking into account the nominal electrical power of the air pump of 66 W and a working time of 106 h during operational Period 1, it is estimated that about 7.0 kWh of electricity was consumed in the ventilation system. This energy consumption is equivalent to 47.2 kWh t^{-1} of evaporated water, which is about half of the energy requirements reported in other full-scale sludge solar drying systems (Table 3). It must be noted that exhaust ventilators at these plants worked under aeration regimes considerably higher than those applied in the present study, and that additional energy must have been spent in mechanical sludge transport and mixing systems that were absent in our pilot-scale plant. Energy consumption during the overall process, including Period 1 and the less favourable Period 2 (433 h of operation and 249 kg of weight loss), amounted to 114.8 kWh t^{-1} of evaporated water.

CONCLUSIONS

We have demonstrated that the direct exploitation of solar energy for drying sewage sludge from the winery industry after harvest is feasible. Assays were performed at the pilot

Table 3 | Comparative assessment of the operational parameters and performance obtained with the solar drier in the present study, with literature accounts on different aerated greenhouse solar driers for treating sewage sludge and related organic waste

Dried substrate	Region/ country	Initial total solids (%)	Final total solids (%)	Drying area (m^2)	Plant type and scale[a]	Superficial airflow[b] ($m\ h^{-1}$)	Solar irradiance ($MJ\ m^{-2}\ d^{-1}$)	Electrical consumption ($kWh\ tH_2O^{-1}$)	Drying rate ($kg\ m^{-2}\ d^{-1}$)	Energy balance[c] (%)	Reference
Winery sewage sludge	Catalonia (Spain)	10.4	33.2	10	PET, TL, PS	4.3	16.9	47.2	2.2	27	This work (Period 1)
Winery sewage sludge	Catalonia (Spain)	10..4	85.7	10	PET, TL, PS	4.3	9.7	114.8	0.7	19	This work (Periods 1 and 2)
Pig slurries	Catalonia (Spain)	11.7	89.6	10	PET, TL, PS	6.0	19.0	30.2	2.0	26	Prenafeta-Boldú *et al.* (2020))
Olive mill sewage	Crete (Greece)	4.9	52.0	4.2	UP, TL, LS	n/a[d]	12.9–29.2	n/a	5.2	n/d[d]	Galliou *et al.* (2018))
Urban sewage sludge	Rhodope (Greece)	15.0	94.0	< 1	PC, MM, LS	n/a	8.6–19.4	n/a	4.0–12.0[e]	n/d	Mathioudakis *et al.* (2009))
Urban sewage sludge	Warsaw (Poland)	20.0	48.0	90	PC, TL, DP	133.3	16.9	n/a	2.3	34	Krawczyk & Badyda (2012)
Urban sewage sludge	Tassos (Greece)	9.7–16.4	82.3–94.3	66	PC, MM, DP	75.8	11.8–25.9	83.0	4.0–11.4	n/d	Mathioudakis *et al.* (2013)
Urban sewage sludge	Paphos (Cyprus)	14.9–23.7	55.7–91.3	3,853	PC, MM, FS	101.3	18.8	77.3	3.1	41	Oikonomidis & Marinos (2014)

Notes: [a]PET, polyethylene film greenhouse cover; PC, polycarbonate plates greenhouse cover; UP, unspecified plastic cover; TL, static thin layer of sludge; MM, mechanical mixing of sludge; LS, laboratory-scale plant; PS, pilot-scale plant; DP, demonstration plant; FS, full-scale plant. [b]Ratio between the ventilation air flow ($m^3\ h^{-1}$) and the drying area (m^2). [c]Ratio between the water enthalpy of vaporization (at 25 °C) and solar irradiance during operation. [d]n/a: not available; n/d: not determined. [e]Maximum measured values.

scale using a relatively simple greenhouse setup coupled with a biofiltration unit, which was demonstrated to be effective for minimizing residual ammonia emissions and VOC contaminants. The obtained dried material retained and concentrated the nutrients that were originally present in the sludge and, therefore, it can be valorized as an organic fertilizer. The dewatering degree of the treated sludge could easily be adjusted for co-composting or lowered to the levels of biological stability for direct storage and utilization. Our results also highlight that the use of copper and zinc-derived fungicides in the field did not compromise the value of the obtained organic fertilizers. Further research on the solar drying of winery organic waste is currently focusing on the mathematical modelling of the process in order to maximize the process efficiency through optimized design parameters and operational conditions. The co-composting of partly dried sludge with other winery organic waste, as well as agronomical assays with the obtained fertilizers, are currently being performed to further validate and improve this innovative treatment technology.

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DATA AVAILABILITY STATEMENT

All relevant data are included in the paper or its Supplementary Information.

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