



# Plant Phosphorus Efficiency from Raw and Composted Agro- and Bio-Waste Anaerobic Digestates

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Received: 10 August 2022 / Accepted: 1 May 2023 / Published online: 11 May 2023  
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## Abstract

The rising generation of organic waste (OW) can be a concern, representing at the same time a valuable opportunity for the phosphorous (P) recycling; however, yet little is known about plant-available P release from this source. In this work, two anaerobic digestates, from agro- ( $AW_D$ ) and bio-waste ( $BW_D$ ), and their respective composts ( $AW_C$  and  $BW_C$ ), were selected to assess their P-release via sequential chemical extraction (SCE) and P species via solution  $^{31}P$ -NMR in NaOH+EDTA extracts. These products were also tested for the relative-P efficiency ( $RPE_{soil}$ ) in a soil incubation ( $30\text{ mg P kg}^{-1}$ ), in comparison with a chemical-P source and a reference compost. The organic products were also compared for the ryegrass relative-P efficiency ( $RPE_{tissue}$ ) at the same P-rate ( $30\text{ mg P kg}^{-1}$ ), in a 112-day pot experiment in an Olsen-P poor soil ( $<3\text{ mg kg}^{-1}$ ), under a non-limiting N environment. The NaOH+EDTA extractions showed that inorganic P prevailed in all samples as proven by solution  $^{31}P$ -NMR. SCE showed very different labile-P ( $H_2O+NaHCO_3$ ) and AW doubling BW products (84 vs. 48%); this was also confirmed by the soil incubation test in which  $AW_D$  and  $AW_C$  attained the best  $RPE_{soil}$ . Pot tests generally confirmed laboratory outcomes showing that  $AW_D$  and  $AW_C$  attained the best  $RPE_{tissue}$ , from 3 to 4-folds of the  $BW_D$  and  $BW_C$  performance. The results showed that the feedstock mainly affects plant P availability from recycled OW and that their SCE are very informative in the description of plant-available P from this type of products to be used in rational fertilization plan.

**Keywords** Sequential chemical extraction · Solution  $^{31}P$ -NMR · Ryegrass · Relative phosphorus efficiency

## 1 Introduction

It is well recognized that phosphorus (P) is the second most important key element for crop nutrition, besides nitrogen (N); in this context, about 80–90 percent of extracted natural phosphate (phosphate rock) is used to maintain high crop productivity (Stamm et al. 2022). This intensive depletion of the natural reservoir reflects in the global incoming P shortage, a pressing issue for the European Union (EU), which has listed P as a critical raw material (EC 2014; Leinweber et al. 2018). In this light, in the latest years, we have assisted to ever increasing re-utilization of organic waste

(OW) as potential substitute of phosphate rock based fertilizers (Bachmann et al. 2016; Grigatti et al. 2019). In this context, the EU's support to anaerobic digestion (AD) for the production of renewable energy strongly fostered the anaerobic processing of different types of organic materials (energy crops, animal slurry, agricultural waste, bio-waste, and sewage sludge). In this framework, the AD process is recognized as a successful approach offering many technical advantages in the OW management (Knoop et al. 2018); AD reduces the OW biological degradability, also limiting possible greenhouse gas and odor emissions during their storing, representing at the same time a renewable energy source (Misselbrook et al. 2016; Zilio et al. 2020; Harrison and Ndegwa 2020). Consequence of these policies is the continuous increasing production of digestates to be properly managed (Banja et al. 2019; Slorach et al. 2019). This represents a management challenge being at the same time a potential resource; as widely reported in the scientific literature, these products can be considered a source of renewable nutrients for crop growth due to their significant amounts of N and P (Möller and Müller 2012; Bachmann

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et al. 2016; Oldfield et al. 2016; Grigatti et al. 2011, 2017a; Nicholson et al. 2017; Slorach et al. 2019). On the other hand, it is generally recognized that raw anaerobic digestates can contain still appreciable amount of readily decomposable OM, thus giving possible important GHG besides to  $\text{NH}_3$  emissions following their utilization in soils (Grigatti et al. 2011; De la Fuente et al. 2013; Nicholson et al. 2017; Grigatti et al. 2020). To reduce these problems, AD is often coupled to composting in order to obtain biologically stable products to be safely used in agricultural soils (Bustamante et al. 2012; Grigatti et al. 2014; Nicholson et al. 2017; Grigatti et al. 2020). As previously said, anaerobic digestates (Ads) can be valuable nutrient sources (N, P); in this framework, there is a wide research about the plant-available N from these products (de Boer, 2008; Grigatti et al., 2011, 2014; Tampio et al. 2016), while still little information is currently available about their potential and actual plant P availability. Crops are widely recognized as primarily utilizing orthophosphate ( $\text{PO}_4^{3-}$ ); this P form is generally poorly available in calcareous soils, where it is rapidly fixed to form poorly soluble Ca-P compounds (Bolan, 1991; Hinsinger, 2001). In fact, in this type of soil, P from chemical fertilizers is poorly efficient, varying between 10 and 25% (Roberts and Johnston 2015). In this context, adding OM to the soil through recycled OW can be beneficial, improving the plant P availability also by increasing competition for P adsorption sites (Guppy et al. 2005), thus promoting the switching from mined P to recycled P sources. Within this framework, many researches have been conducted on P extractability and/or plant P uptake from animal manure/slurry, sewage sludge or their digestate (Bachmann et al. 2011; Waldrip et al. 2011; Pagliari and Laboski 2012, 2013; García-Albacete et al. 2012; Tampio et al. 2016), while only little information is available on the P-fertilizing properties of raw and composted anaerobic digestates from agro- and bio-wastes. The wet co-digestion of agro-waste requires the liquid/solid fraction separation for a proper management of by-products. While the liquid fraction is successfully utilized as N fertilizer, the solid fraction is generally considered as soil amendment with almost unknown fertilizing capacity, especially regarding P. Besides this, there is little information also on the P availability in the products derived from the AD of bio-waste. AD of bio-waste is a rather novel technology, particularly regarding the dry-batch process this giving only a high total solid fraction in the end.

Although little information is available about the P availability from the solid fractions of Ads, it is generally recognized that most of the P is allocated in this component (Möller and Müller 2012). Nevertheless as a result of a general increase in pH, the precipitation of Ca-P and Mg-P compounds may occur during AD (in addition to the precipitation of Fe-bound forms of P), this often limiting the P potentially available to plants (Möller and Müller 2012).

In addition to the changes in the potential P availability following the anaerobic digestion, also composting can modify this trait. These variations were studied by Wei et al. (2015); these authors showed a general reduction of easily available P following the composting of various types of OW (animal manure; municipal solid waste; green waste; food waste), being this reduction confirmed by Grigatti et al. (2017b), during the composting of agro-industrial and urban sewage sludge.

As previously mentioned, there are few studies about agro- and bio-waste derived products, and the researches focused on the variation occurring at P plant availability are even less. In this framework, the utilization of chemical assessment, incubation with soil and pot test with plant, can give a more reliable overview of the potential P substitution of chemical P sources with recycled OW. The potential availability of P from OW can be investigated through sequential chemical extraction (SCE), which provides a reliable insight into theoretical P release in the short-, medium-, and long-term as described by Dou et al. (2000). In this context the use of (NaOH+EDTA) solution  $^{31}\text{P}$ -NMR can be a very helpful technique to assess the different P species in recycled organic waste to be used as fertilizers (Turner and Leytem 2004). Besides P chemical investigation, soil incubation tests are widely utilized to determine the potentially plant-available P (Olsen-P) release from recycled OW under laboratory simulated field conditions. These tests are very informative about the initial P availability and subsequent course over time depending on P fixation by soil components on P release following the OW mineralization (and/or desorption). Additionally, in this test, the comparison with a chemical P source allows the assessment of the relative P efficiency ( $\text{RPE}_{\text{soil}}$ ) from recycled OW (Grigatti et al. 2020). Besides to this, the plant test in pot grown in a non-limiting N environment offers a valuable insight into the apparent P utilization efficiency of the compared products. Working with rapid-growing grass, such as ryegrass under well-controlled conditions of moisture, temperature, and light and constant mineral N supply, allows for a multi-harvest collection offering a better insight to the apparent P utilization kinetics in the time frame of a simulated growing season (Grigatti et al. 2019). The presence of a chemical P source along with an unamend control allows the calculation of the relative plant P efficiency ( $\text{RPE}_{\text{tissue}}$ ).

In this light, the present study was aimed at investigating the effect of the composting of anaerobic digestates obtained from agro- and bio-waste on the chemical fractionation of P, its speciation via solution  $^{31}\text{P}$ -NMR, along with the release of P into the soil and its plant availability. To do this, we compared the anaerobic digestate from agro- and bio-waste together with their composted homologous via SCE, solution  $^{31}\text{P}$ -NMR, and a soil incubation experiment to assess

the Olsen-P and the  $RPE_{\text{soil}}$  course. These products were compared with a reference compost (from bio-waste, without AD), a soluble chemical P source, and an unamended control. These treatments were also compared in a pot test to determine the P utilization efficiency ( $RPE_{\text{tissue}}$ ) on ryegrass subjected to multiple cuts over 112 days in a non-limiting N environment.

## 2 Material and Methods

### 2.1 Organic Products

Four organic products derived from either agro- or bio-waste were focused in this work: an anaerobic digestate was collected after 62 days of (thermophilic; 50 °C) wet co-digestion of (33/33/33 V/V) energy-crop/agricultural-residues/animal-slurry ( $AW_D$ ); a compost was obtained after 60 days of composting of (screw-pressed)  $AW_D$  without the addition of green waste ( $AW_C$ ) (Fig. 1); another anaerobic digestate

was collected after 28 days of (mesophilic; 37 °C) dry-batch digestion of a mixture of bio-waste and green waste (85/15, w/w) ( $BW_D$ ). The respective compost ( $BW_C$ ) was obtained after 62 days of  $BW_D$  processing: the structural material derived from recirculation of the solid fraction (50% V/V), with the addition of extra green waste (15% V/V) (Fig. 1). In addition to these, a municipal solid waste compost (MSWC) from bio-waste and green waste was used as organic references in the soil and pot test. pH, total solids (TS) and volatile solids (VS), and the oxygen uptake rate (OUR; Grigatti et al. 2007), were measured on fresh products. Then the different products were freeze-dried and ball milled for further analytical determinations. Total organic carbon (TOC) and total nitrogen (TN) were determined via elemental analyzer (EA 1110, Thermo Electron, Germany). The total nutrients and trace elements were determined using ICP-OES (Inductively Coupled Plasma-Optical Emission spectrometry; Spectro Arcos, Ametek, Berwin, PA, USA) on  $\approx 250$  mg samples after microwave-assisted digestion (37% HCl and 65%  $HNO_3$ ). The main product characteristics are summarized in Table 1.

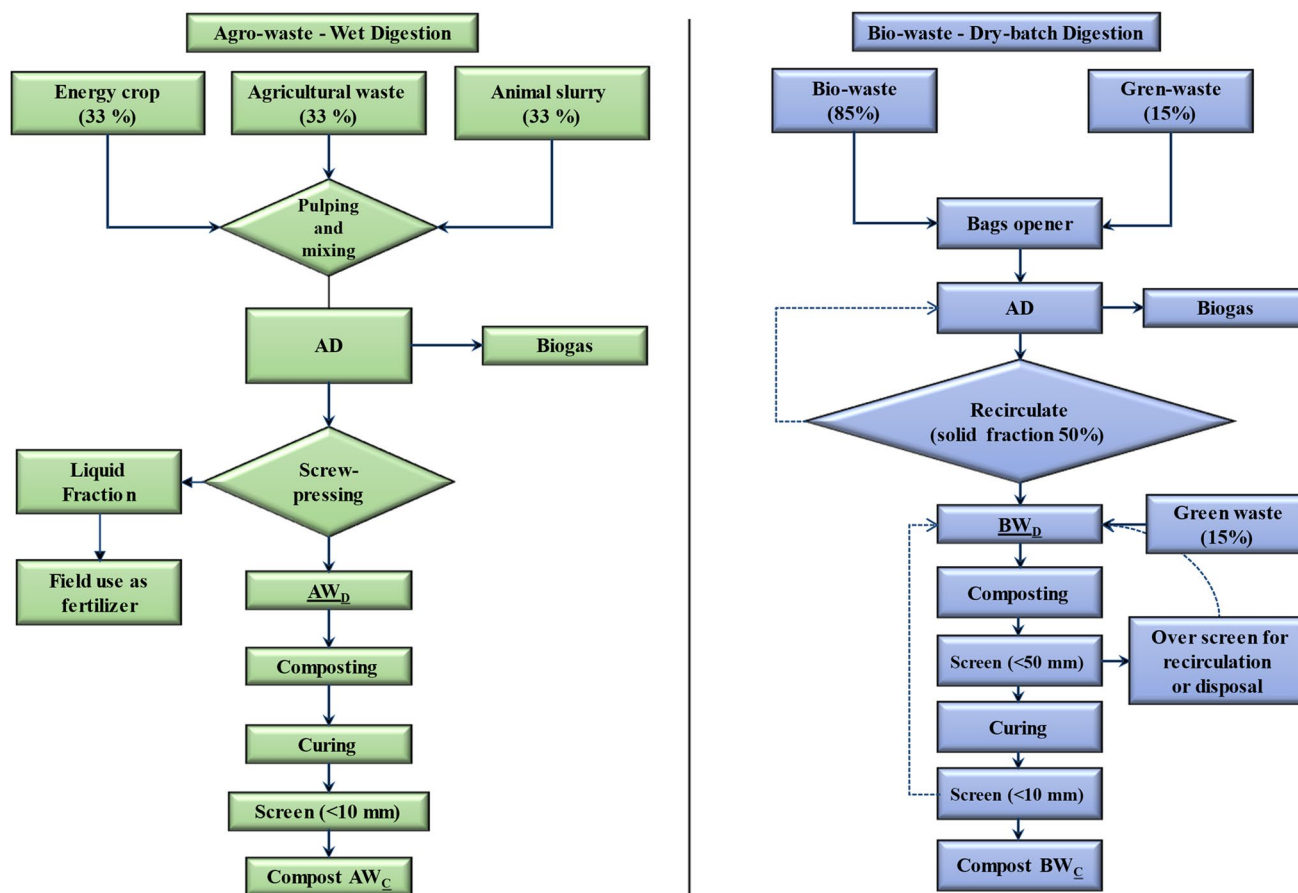


Fig. 1 Schematic representation of the wet and dry-batch anaerobic digestion and composting process of agro- and bio-waste

**Table 1** Main characteristics of the tested products

Product	pH	TS (%)	VS	TOC	TN	C:N	OUR (mmol O <sub>2</sub> kg <sup>-1</sup> VS h <sup>-1</sup> )	P (mg g <sup>-1</sup> )	Ca	Fe	Al	Mg	K	S	Ca:P	Fe:P	Al:P
AW <sub>D</sub>	8.12	81	90	49.0	1.45	34	11	6.1	12.3	1.9	1.0	4.0	11.0	2.9	2	0.3	0.2
BW <sub>D</sub>	8.13	90	56	28.1	2.22	13	162	5.0	85.2	6.6	7.8	4.2	14.2	2.7	17	1.8	2.1
AW <sub>C</sub>	7.64	83	87	48.3	1.94	25	8	6.7	13.8	2.1	1.2	4.4	10.5	3.4	2	0.3	0.2
BW <sub>C</sub>	7.30	92	54	27.7	2.05	14	21	3.9	61.3	8.4	9.7	5.1	7.6	2.2	16	2.7	3.2
MSWC	8.33	89	44	22.2	2.23	10	6	5.0	46.2	14.6	15.5	6.3	12.3	3.4	9	3.4	3.6

Values are means of two replicates; data are expressed on TS; mean deviation <5%

TS total solids, VS volatile solids, TOC total organic carbon, TN total nitrogen, OUR oxygen uptake rate, AW<sub>D</sub> digestate from the wet digestion of agro-waste, BW<sub>D</sub> digestate from the dry-batch digestion of bio-waste, AW<sub>C</sub> compost from AW<sub>D</sub>, BW<sub>C</sub> compost from BW<sub>D</sub>, MSWC reference compost

## 2.2 Phosphorous Fractionation

The organic products were submitted to P fractionation via sequential chemical extraction (SCE) according to the method of Dou et al. (2000). Freeze-dried and ball-milled products were extracted for 24 h with deionized water (d-H<sub>2</sub>O) in an end-over-end shaker and then centrifuged. The supernatants were then filtered (Whatman #42), and the recovered pellets were extracted with 0.5 M NaHCO<sub>3</sub> (pH 8.5) for 24 h. The same procedure was repeated with 0.1N NaOH and 1N HCl. The phosphorous in the extracts was determined via ICP. The P recovered in each fraction [water extractable P (H<sub>2</sub>O-P), bicarbonate extractable P (NaHCO<sub>3</sub>-P), alkali extractable P (NaOH-P), acid extractable P (HCl-P)] was calculated as follows:

$$P_{fraction\ x}(\%) = \frac{P_{fraction\ x}}{P_{tot\ OW}} \times 100$$

where  $P_{fraction\ x}$  is the P determined (via ICP) in each fraction (H<sub>2</sub>O, NaHCO<sub>3</sub>, NaOH, HCl), and  $P_{tot\ OW}$  is the total P determined in the different organic waste via ICP after microwave-assisted acid digestion. The total recovery was calculated as the sum of all fractions (H<sub>2</sub>O-P + NaHCO<sub>3</sub>-P + NaOH-P + HCl-P) by using the following equation:

$$Tot\ P_{recovery}(\%) = \frac{\sum_{H_2O}^{HCl} P_{fraction\ x}}{P_{tot\ OW}} \times 100$$

where  $\sum_{H_2O}^{HCl} P_{fraction\ x}$  represents the sum of the single P recovery values (P) in each fraction (H<sub>2</sub>O-P + NaHCO<sub>3</sub>-P + NaOH-P + HCl-P) and  $P_{tot\ OW}$  is the total P determined in the different products via ICP after microwave-assisted acid digestion.

## 2.3 NaOH+EDTA Extractable P and Solution <sup>31</sup>P-NMR

The different products were extracted (1:60; 16 h, 25 °C) with NaOH (0.25N) + EDTA (0.05M). After centrifugation (20'; 10000 × g) and filtration (Whatman #42), the extracts were analyzed via ICP to assess the total P (P<sub>ICP</sub>) and via molybdenum blue reaction to assess inorganic PO<sub>4</sub>-P (P<sub>i</sub>). The organic P (P<sub>o</sub>) was determined as the difference between P<sub>ICP</sub> and P<sub>i</sub>. <sup>31</sup>P-NMR spectra were acquired on the same NaOH+EDTA extracts (freeze-dried and re-dissolved in 1M NaOH + 0.1M EDTA), using a Varian Inova spectrometer with operational frequency of 243 MHz. A 10% of deuterium oxide (0.1 mL) was added to samples for signal lock, and the solution was transferred to a 5-mm NMR tube. The inclusion of NaOH ensures consistent chemical shifts at pH>13, while EDTA reduces line broadening by chelating paramagnetic ions (Turner and Leytem, 2004). Spectra were collected at 298 K, using a 45° pulse angle, a relaxation delay (d1) of 5 s, an acquisition time of 1.6 s, and inverse gated decoupling sequence. Between 2000 and 8000 scans were collected to obtain acceptable signals. Chemical shifts of signals were determined in parts per million (ppm) relative to 85% H<sub>3</sub>PO<sub>4</sub> (external reference) and assigned to individual P compounds or functional groups according to Cade-Menun (2005). Because slight shifts in frequencies may happen from one sample to another (due to drifts, matrix effects), AW<sub>D</sub> and BW<sub>D</sub> samples were also subsequently collected adding inorganic orthophosphate (spiked samples), to verify the chemical shift of this reference (Figure S1). Spiked samples were collected on a Varian Mercury Plus NMR spectrometer with operational frequency of 162 MHz at 298 K as qualitative spectra with a 45° pulse angle, a relaxation delay of 1 s, and an acquisition time of 1.6 s. All spectra were plotted using a line broadening ranging from 1 to 3 Hz.



## 2.4 Soil Incubation

The soil used for the incubation was collected from the top layer of a field in the Po Valley (Bologna, Italy), as described by Jimenez et al. (2020). In brief, this showed a sub-alkaline pH (7.90); particle-size distribution (mg kg<sup>-1</sup>), 184 sand, 425 silt, 391 clay; total CaCO<sub>3</sub>, 85 g kg<sup>-1</sup>; total organic carbon (TOC), 10.2 g kg<sup>-1</sup>; total Kjeldahl nitrogen (TKN), 1.60 g kg<sup>-1</sup>; C:N, 8.3; CEC 27.2 cmol kg<sup>-1</sup>. The Olsen-P was detected at 3.00 mg P kg<sup>-1</sup>. Before the incubation, soil was pre-incubated for four weeks at 25 °C (60% water holding capacity in the dark). For the assessment of the Olsen-P in soil, the organic products (AW<sub>D</sub>, AW<sub>C</sub>, BW<sub>D</sub>, and BW<sub>C</sub>), the organic reference (MSWC) and the chemical P source [Ca(H<sub>2</sub>PO<sub>4</sub>)<sub>2</sub>; P-chem] were added to the pre-incubated soil and mixed thoroughly to achieve 30 mg P kg<sup>-1</sup>. A unamended soil was also compared (Control), in a completely randomized design with three replicates. During incubation (25 °C; 112 days), the moisture content was monitored by weighing and addition of deionized water twice a week. The Olsen-P value was determined using the molybdenum blue method at days 0, 14, 28, 56, 84, and 112. The Olsen-P data were used to calculate the relative phosphorus efficiency (RPE<sub>soil</sub>), which was used to normalize the P data obtained in the organic treatments to P-Chem, according to Leytem and Westermann (2005).

## 2.5 Plant Test

The organic products (AW<sub>D</sub>, AW<sub>C</sub>, BW<sub>D</sub>, and BW<sub>C</sub>) and the organic reference (MSWC) were added to the same soil used for incubation at the same rate (30 mg P kg<sup>-1</sup>) and carefully mixed. The chemical P fertilizer (P-Chem) was added by adding 10 mL of a Ca(H<sub>2</sub>PO<sub>4</sub>)<sub>2</sub> solution to achieve a rate of 30 mg P kg<sup>-1</sup>. A unamended soil was also compared (Control) in a completely randomized design with four replicates. The amended soil (1 kg on TS basis) was added to 2 L pots (ø 16 cm), previously half filled (1 L) with expanded agricultural clay (with controlled pH). The ryegrass (*Lolium multiflorum* subsp. *Italicum*) seeds, cv. Sprint were sown (1 g pot<sup>-1</sup>). Pots were placed in a climatic room with a 14/10 h light/dark photoperiod (Master T1d 58 W-840 tubes, Philips, Amsterdam, The Netherlands) and respective temperatures of 23/13 °C. Throughout the experiment ryegrass was fertilized 4 times (day 0, 28, 56, and 84), with NH<sub>4</sub>NO<sub>3</sub> (totally 200 mg N pot<sup>-1</sup>), ensuring non-limiting N environment. The pots were regularly watered with tap water to keep constant moisture level (60% WHC). At 4-week intervals (28, 56, 84, and 112 days after sowing), the plants were harvested by cutting the stems 2 cm above the soil. Harvested tissues were dried in a forced-air oven (60 °C for 72 h). On the last harvest (day 112), the roots were also recovered by watering way the soil, and they were dried as described above. Total P was determined on plant tissue and root via ICP after microwave-assisted acid digestion (65% HNO<sub>3</sub>).

## 2.6 Statistical Analysis

After checking homogeneity of variances by Levene's test, a one-way ANOVA was run on single (days: 28, 56, 84, and 112) and cumulated harvests of shoots (days 28–112) and root at final harvest (day 112), addressing DW, P content, and P uptake. Tukey's HSD test at  $P < 0.05$  was used for means separation (Statistica 7® package; Statsoft, Tulsa, OK, USA).

The cumulative relative tissue P efficiency (RPE<sub>tissue</sub>) was calculated as follows:

$$\text{RPE}_{\text{tissue}} (\%) = \sum_{28}^{112} \frac{(P_{\text{treatment}(t)} - P_{\text{control}(t)})}{(P_{\text{Chem}(t)} - P_{\text{control}(t)})} \times 100$$

where  $P_{\text{treatment}(t)}$  is the P uptake (mg pot<sup>-1</sup>) from the tissue of treated pots (AW<sub>D</sub>, AW<sub>C</sub>, BW<sub>D</sub>, BW<sub>C</sub>, and MSWC) at time  $t$  (28; 56; 84; 112);  $P_{\text{control}(t)}$  is the P uptake (mg pot<sup>-1</sup>) from the tissue of control pots at time  $t$  (28; 56; 84; 112); and  $P_{\text{Chem}(t)}$  is the P uptake (mg pot<sup>-1</sup>) from the tissue of P-Chem pots at time  $t$  (28; 56; 84; 112). The total P uptake (shoot+ root) was calculated by summing the tissue cumulative P uptake (at day 112) and the root P uptake (at day 112).

## 3 Results

### 3.1 Characteristics of the Organic Products

The organic products compared in this work showed important differences in their main characteristics. In this framework the compared Ads showed wide VS content these ranging between 90 and 56 % (AW<sub>D</sub> and BW<sub>D</sub>), consistent with the organic carbon content (49 vs. 28 %); besides to this, these products showed also wide ranging biological stability (OUR: 11 and 162 mmol O<sub>2</sub> kg<sup>-1</sup> VS h<sup>-1</sup>). In these products, the N content varied between 1.45 and 2.22%, while similar was the P content (6.1 and 5.0 mg g<sup>-1</sup>), displaying at the same time various Ca:P (2 vs. 17), Fe:P (0.3 vs. 1.8), and Al:P (0.2 vs. 2.1). In this context, the composting process weakly affected the biological stability of AW<sub>D</sub>, being on the contrary the stabilization of BW<sub>D</sub> very effective showing a reduction of the OUR from 162 down to 21 mmol O<sub>2</sub> kg<sup>-1</sup> VS h<sup>-1</sup>. Related to the stabilization process, some variations in the N content were registered giving very different C:N ratio in the end, these varying between 24 and 14 (Table 1). Despite some changes in the element content, any appreciable variation in the Ca:P, Fe:P, and Al:P ratio was detectable in the compared products (Table 1). In Table 2, the heavy metal contents are reported clearly showing that AW<sub>D</sub> had notably lower Cd, Cr, Ni, Cu, and Pb contents in comparison to BW<sub>D</sub>; only Zn was detected at similar level. Besides to this, composting AW<sub>D</sub> to AW<sub>C</sub> slightly affected the heavy metal content. Conversely, composting BW<sub>D</sub> to BW<sub>C</sub> reflected in a general heavy metal decrease (Table 2). However, all the

**Table 2** Heavy metal content in the compared products (mg kg<sup>-1</sup>)

Product	Element					
	Cd	Cr*	Ni	Cu	Zn	Pb
AW <sub>D</sub>	0.19	13.8	3.8	17.8	111	nd
BW <sub>D</sub>	0.15	72.3	11.2	80.8	134	18.9
AW <sub>C</sub>	0.19	12.4	3.9	19.5	134	n.d.
BW <sub>C</sub>	0.17	40.2	14.2	64.8	126	22.3
MSWC	0.54	65.5	29.1	149.3	267	377
Limits Reg. EU 1009/2019	1.5	n.a.	50	300	800	120

\*Cr total. Values are means of two replicates of organic samples; data are expressed on TS; mean deviation <5%

AW<sub>D</sub> digestate from the wet-digestion of agro-waste, BW<sub>D</sub> digestate from the dry-batch digestion of bio-waste, AW<sub>C</sub> compost from AW<sub>D</sub>, BW<sub>C</sub> compost from BW<sub>D</sub>, MSWC reference compost, n.d. not detectable, n.a. not applicable

compared organic products showed overall lower heavy metal content in comparison to the reference compost (MSWC).

### 3.2 Phosphorous Fractionation

The organic products compared in this work showed important differences in the P fractionation (Table 3). In this frame, AW<sub>D</sub> showed the highest H<sub>2</sub>O-P, doubling BW<sub>D</sub> (3.8 vs. 1.4 mg g<sup>-1</sup>). Conversely, both AW<sub>D</sub> and BW<sub>D</sub> showed similar NaHCO<sub>3</sub>-P (1.3 vs. 1.0 mg g<sup>-1</sup>), thus performing very different labile-P in the end (84 vs. 48%). These products (AW<sub>D</sub> and BW<sub>D</sub>) showed also important differences in the NaOH-P (0.7 vs. 1.2 mg g<sup>-1</sup>), this representing 12 and 25% of the whole extractable-P. The sparingly soluble P (HCl-P), was detected at 0.9 and 2.4 mg g<sup>-1</sup> in AW<sub>D</sub> and BW<sub>D</sub>, respectively, these corresponding to 14 and 47% of extractable P. Also the composted products (AW<sub>C</sub> and BW<sub>C</sub>) showed very different H<sub>2</sub>O-P (3.2 vs. 0.7 mg g<sup>-1</sup>) and NaHCO<sub>3</sub>-P (2.0 vs. 0.8 mg g<sup>-1</sup>), thus attaining to very different labile-P (79 vs. 40%).

AW<sub>C</sub> and BW<sub>C</sub> showed similar amounts of NaOH-P (1.1 vs. 1.4 mg g<sup>-1</sup>), representing 16 and 37% of extractable P. The amounts of acid-extractable P were similar in AW<sub>C</sub> and BW<sub>C</sub> (1.6 and 1.4 mg g<sup>-1</sup>), even if their recovery was very different (24 and 35%). The reference compost (MSWC) showed 1.2 mg g<sup>-1</sup> of both H<sub>2</sub>O-P and NaHCO<sub>3</sub>-P, thus attaining to 49% of labile-P. The alkali extractable-P was detected at 0.7 mg g<sup>-1</sup> in MSWC, while the acid extractable-P was much higher (2.8 mg g<sup>-1</sup>); these fractions accounted respectively for 13 and 56% of the whole extractable P.

### 3.3 NaOH+EDTA Extractable P and Solution <sup>31</sup>P-NMR

The inorganic/organic P fractionation obtained following the NaOH+EDTA extraction showed the prevalence of P<sub>i</sub> in both agro- and bio-waste derived products, this representing ≈80% of extractable P (Table 3). Composting weakly modified this ratio. In the <sup>31</sup>P-NMR results reported in Fig. 2

**Table 3** Phosphorous removed in the sequential chemical extraction and NaOH+EDTA extraction of the tested products

Extractant/fraction	AW <sub>D</sub>		BW <sub>D</sub>		AW <sub>C</sub>		BW <sub>C</sub>		MSWC	
	(mg g <sup>-1</sup> )	(%)	(mg g <sup>-1</sup> )	(%)	(mg g <sup>-1</sup> )	(%)	(mg g <sup>-1</sup> )	(%)	(mg g <sup>-1</sup> )	(%)
H <sub>2</sub> O	3.8	62	1.4	28	3.2	48	0.7	19	1.2	24
NaHCO <sub>3</sub>	1.3	22	1.0	20	2.0	31	0.8	21	1.2	25
NaOH	0.7	12	1.2	25	1.1	16	1.4	37	0.7	13
HCl	0.9	14	2.4	47	1.6	24	1.4	35	2.8	56
∑ P-fractions (mg g <sup>-1</sup> )	6.7		6.0		7.9		4.3		5.9	
Recovery (% of P <sub>tot</sub> )		110		120		118		111		118
NaOH+EDTA-P <sub>ICP</sub>	7.33	–	4.72	–	5.98	–	2.93	–	4.22	–
NaOH+EDTA-P <sub>i</sub>	5.84	80	3.84	81	4.57	76	2.25	77	2.25	82
NaOH+EDTA-P <sub>o</sub>	1.49	20	0.88	19	1.41	24	0.68	23	0.68	18

Values are means of two replicates of organic samples; data are expressed on TS; mean deviation <5%

AW<sub>D</sub> digestate from the wet digestion of agro-waste, BW<sub>D</sub> digestate from the dry-batch digestion of bio-waste, AW<sub>C</sub> compost from AW<sub>D</sub>, BW<sub>C</sub> compost from BW<sub>D</sub>, MSWC reference compost

(Figure S2a and S2b for a whole picture), all the analyzed samples showed limited range of signals comprised between 5.97 and  $-4.64$  ppm. More in depth, the compared samples ( $AW_D$ ;  $BW_D$ ;  $AW_C$ ;  $BW_C$ ; Fig. 2a–d) exhibited a narrow resonance between 5.80 and 6.05 ppm, which is attributed to orthophosphate, also detectable in the reference compost (MSWC; Fig. 2e). Slight variations in phosphorus species frequencies in organic waste extracts were already reported; this may be due to a matrix effect (Turner et al. 2003; Cade-Menun, 2005); however, the samples spiked with orthophosphate confirmed that the main peak around 5.80–6.05 ppm is correctly attributed to this species (Fig. S1a and S1b). The signals between 4.0 and 6.0 ppm were assigned to phosphate monoesters, with those at approximately 5.75, 5.01, 4.70, and 4.56 ppm assigned to phytic acid (Cade-Menun, 2005). All samples, with the exception of MSWC, showed a small signal close to  $-4.56$  and  $-4.59$  ppm assigned to pyrophosphate; a slight increase of the pyrophosphate signal is noted in composted samples (Preston et al., 1998).

### 3.4 Soil Olsen-P

As reported in Fig. 3a, the Olsen-P in control soil (Ctrl) was detected at the lower level throughout the incubation ( $\approx 3$  mg  $kg^{-1}$ ). Among the organic products  $AW_D$  and  $AW_C$  attained the best since the beginning ( $\approx 30$  mg  $kg^{-1}$ ), thus showing a common reduction at the third sampling (day 28), performing fairly constant Olsen-P to the end.  $BW_D$  and  $BW_C$  performed the worst Olsen-P from the beginning (5–6 mg  $kg^{-1}$ ), maintaining fairly constant value to the end (day 112), in the same range of the reference compost (MSWC). The chemical P source (P-Chem) showed  $\approx 25$  mg  $kg^{-1}$  Olsen-P since the beginning, then a decrease within two weeks, and finally a constant value to the end. As shown in Fig. 3b, the  $RPE_{soil}$  (%) at day 0 was generally consistent with Olsen-P's results:  $AW_D$  (163) >  $AW_C$  (121) >  $BW_D$  (16)  $\geq$  MSWC (11)  $\geq$   $BW_C$  (8), also being consistent throughout the incubation test.

### 3.5 Plant Test

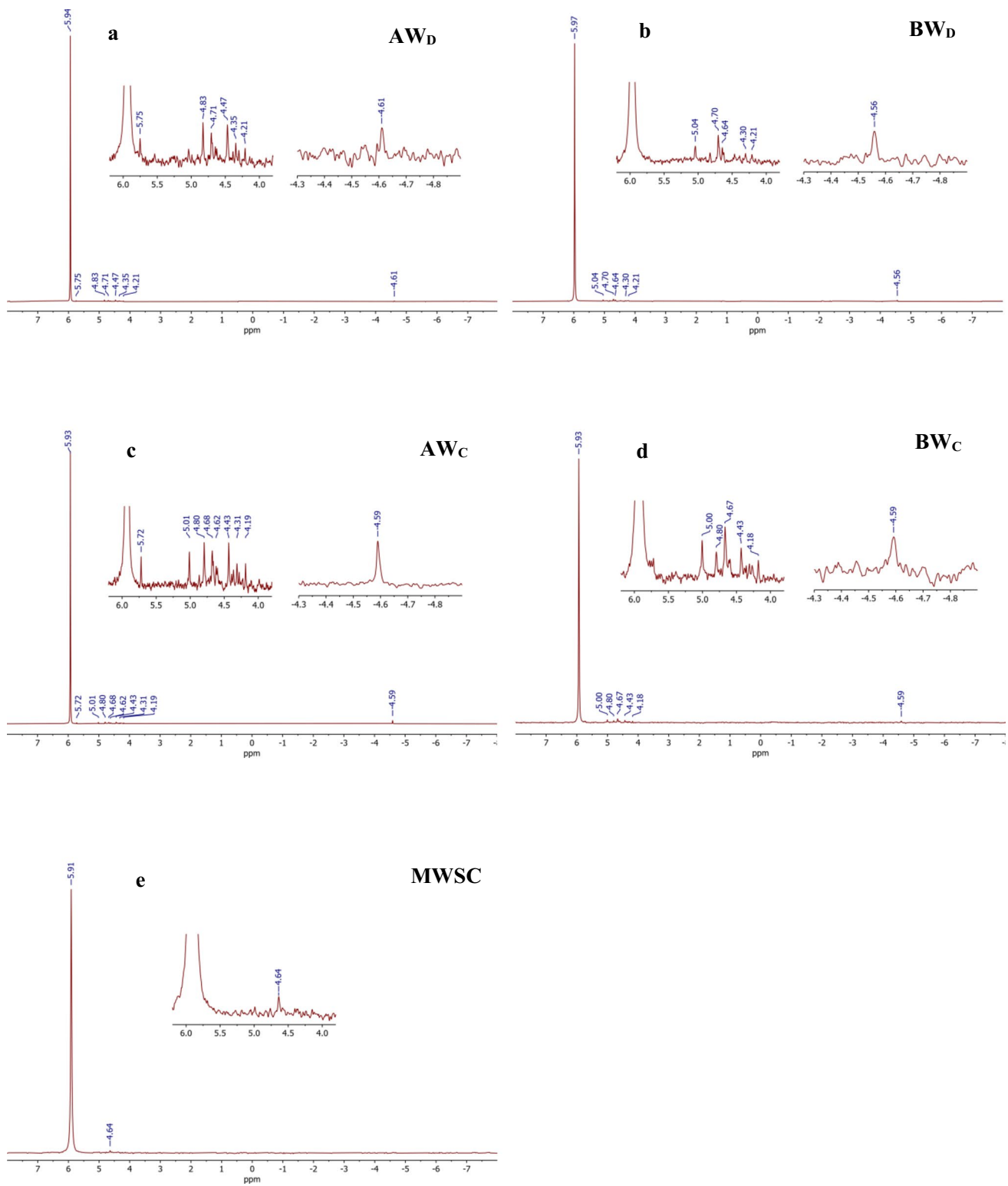
Pot test showed the ryegrass dry biomass was affected by both treatment ( $p < 0.05$ ) and harvest ( $p < 0.05$ ). As reported in Table 4, the worst shoot DW was registered at the first harvest at day 28 (0.84 g  $pot^{-1}$ , on average); in the following harvests (days 56, 84, and 112), a shoot DW increase was registered this attaining to 1.20 g  $pot^{-1}$  (on average). To a deeper insight unamended control soil (Ctrl) performed the worst shoot biomass over 112 days (0.97 g  $pot^{-1}$ , on average), while  $AW_C$  performed the best (1.34 g  $pot^{-1}$ , on average). The other treatments ( $AW_D$ ,  $BW_D$ , and  $BW_C$ ) were intermediate, in the same range of MSWC and P-Chem (1.09 g  $pot^{-1}$ , on average). Similar was the cumulated (28–112 days) shoot DW pattern, showing  $AW_C$  the best (5.35 g

$pot^{-1}$ ),  $BW_D$ ,  $AW_C$  and Ctrl performed the worst (4.04 g  $pot^{-1}$ , on average), in the same range of MSWC (4.36 g  $pot^{-1}$ ).  $AW_D$  was intermediate, similarly to P-Chem (4.58 g  $pot^{-1}$ , on average). Root biomass showed a different pattern; in this frame, Ctrl and MSWC showed the best and the worst DW, respectively (2.49 and 1.83 g  $pot^{-1}$ ), while the other treatments ( $AW_D$ ,  $AW_C$ ,  $BW_D$ , and  $BW_C$ ) were intermediate (2.28 g  $pot^{-1}$ , on average), in the same range of P-Chem (1.96 g  $pot^{-1}$ ). Overall the whole biomass of shoot + root (g  $pot^{-1}$ ) was the best in  $AW_C$  (7.64)  $\geq$   $AW_D$  (7.06) =  $BW_C$  (6.50) = P-Chem (6.49) = Ctrl (6.38) = MSWC (6.19). The P content measured in shoot was affected by treatment ( $p < 0.05$ ) and harvest ( $p < 0.05$ ). As reported in Table 3, this was the best at the first harvest at day 28 (1.67 mg  $g^{-1}$ , on average), showing a progressive averaged decline down to 1.34, 1.13, and 0.98 mg  $g^{-1}$  at the following harvests (days 56, 84, and 112). The shoot P content was the worst in Ctrl (1.05 mg  $g^{-1}$ , on average).  $AW_D$  attained the best, in the same range of  $AW_C$  (1.53 and 1.45 mg  $g^{-1}$ , on average), close to P-Chem (1.33 mg  $g^{-1}$  on average).  $BW_D$  and  $BW_C$  performed lower shoot P (1.18 mg  $g^{-1}$ , on average), close to MSWC (1.22 mg  $g^{-1}$ ). Root P showed a similar pattern: Ctrl was the worst (0.86 mg  $g^{-1}$ ), in the same range of  $BW_D$  (0.95 mg  $g^{-1}$ );  $AW_D$  and  $AW_C$  performed the best (1.52 mg  $g^{-1}$ , on average), higher than  $BW_C$  (1.30 mg  $g^{-1}$ ), MSWC (1.15 mg  $g^{-1}$ ), and P-Chem (1.06 mg  $g^{-1}$ ). The plant P uptake (Table 3) was affected by treatment ( $p < 0.05$ ) and time ( $p < 0.05$ ). The total (shoot + root) P uptake (mg  $pot^{-1}$ ) was the best in  $AW_C$  (11.14)  $\geq$   $AW_D$  (10.58) >  $BW_C$  (7.75) = P-Chem (7.76)  $\geq$  MSWC (7.37) =  $BW_D$  (6.69) = Ctrl (6.15).

The combination of these data gives the ryegrass tissue cumulated (days 28–112) and the total (shoot+ root) net uptake at day 112. As shown in Fig. 4a, b,  $AW_D$  and  $AW_C$  best performed in comparison to  $BW_D$  and  $BW_C$  and also of the reference compost (MSWC), while Fig. 4c shows the tissue relative P efficiency ( $RPE_{tissue}$ ; % of P-Chem) thus showing low ( $BW_D$  and  $BW_C$ ) and high performers ( $AW_D$  and  $AW_C$ ).

## 4 Discussion

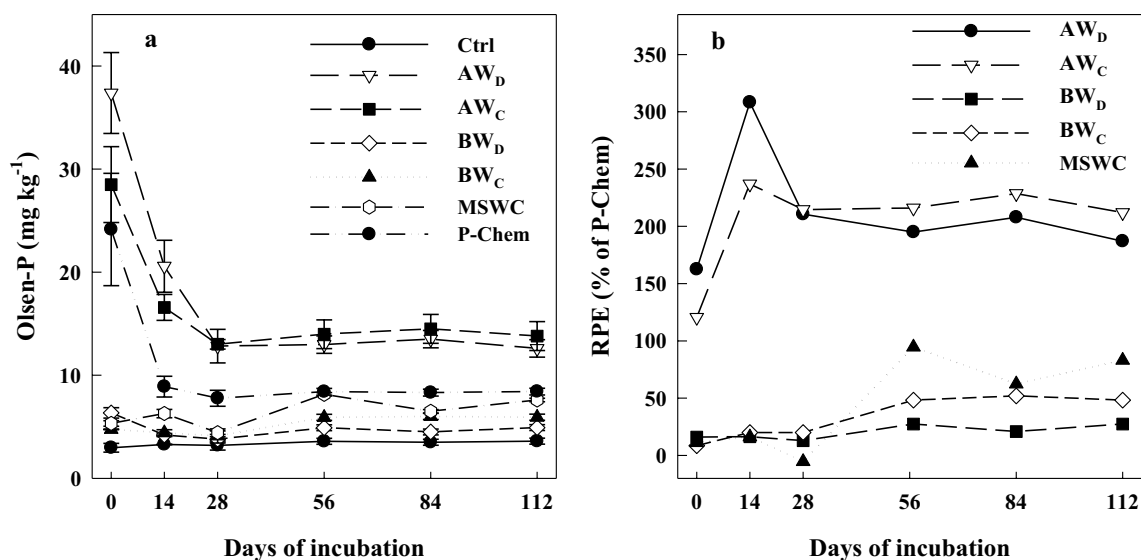
The organic products compared in this work exhibited markedly varying properties derived from the different feedstock and processes. In this framework, the agro-waste-derived products averaged  $\approx 50\%$  higher VS and C content in comparison to the bio-waste ones, these performing oppositely  $\approx 20\%$  lower N. These data are in agreement to what reported in literature for similar products (Bachmann et al. 2016; Peng and Pivato 2019). In this frame, it is recognized that agro-waste and animal slurry mixture to be co-digested are mainly formed by carbon-rich source (starch, cellulose, hemicellulose, and lignin); oppositely bio-waste are,



**Fig. 2**  $^{31}\text{P}$ -nuclear magnetic resonance (NMR) spectra of tested samples extracted with NaOH+EDTA. Insets for detailed view. **a** Digestate from the wet digestion of agro-waste ( $\text{AW}_D$ ); **b** digestate from

the dry-batch digestions of bio-waste ( $\text{BW}_D$ ); **c** compost from  $\text{AW}_D$  ( $\text{AW}_C$ ); **d** compost from  $\text{BW}_D$  ( $\text{BW}_C$ ); **e** reference compost (MSWC)





**Fig. 3** Olsen-P in soil (a) and relative P efficiency ( $RPE_{soil}$ ) (b) during the soil incubation of the compared products.  $AW_D$ , digestate from the wet digestion of agro-waste;  $BW_D$ , digestate from the dry-

batch digestions of bio-waste;  $AW_C$ , compost from  $AW_D$ ;  $BW_C$ , compost from  $BW_D$ ; MSWC, reference compost; P-Chem,  $Ca(H_2PO_4)_2$

generally, composed of a mixture formed of organic materials (carbohydrates, proteins, and fats), along with important amount of ash-rich materials such as egg shells and bones (Campuzano and González-Martínez 2016). Besides these features, the compared products showed also very different biological stability performing  $AW_D$  very low OUR in comparison to  $BW_D$  (11 vs. 162  $mmol O_2 kg^{-1} VS h^{-1}$ ). It is recognized that wet digestion performs higher OM degradation, giving more stable products in the end in comparison to the dry-batch process (Peng and Pivato 2019). In addition, it is recognized that anaerobically non-degradable compounds (such as lignin) are collected in the solid fraction, being the most easily degradable molecules generally collected in the liquid fraction (Bachmann et al. 2016). In this framework, composting poorly affected the OUR during the stabilization process of  $AW_D$  (from 11 to 7  $mmol O_2 kg^{-1} VS h^{-1}$ ); conversely, the composting process efficiently reduced the respiration rate of  $BW_D$  (from 162 down to 21  $mmol O_2 kg^{-1} VS h^{-1}$ ). In this context, composting has proven to play a key role in stabilizing dry-batch digestate from bio-waste in order to be safely used it in agricultural soils, being 25  $mmol O_2 kg^{-1} VS h^{-1}$  the OUR threshold indicated in the recently proposed EU Fertilizer Regulation (EU, 2019). The OM mineralization occurred during the composting of  $AW_D$  and slightly increased the elements and the heavy metals contents in its compost ( $AW_C$ ); conversely, the addition of a bulking agent for the composting of  $BW_D$  produced small elements and heavy metals dilution in  $BW_C$ . However, the elemental content of all the organic products compared in this work were within the normal range for this type of product (Jin and Chang 2011; Kupper et al. 2014), also meeting

the EU Fertilizer Regulation regarding the heavy metal content (EU, 2019). In any case, the composting process did not significantly modify the Ca:P and Fe:P (and Al:P) ratios of raw digestates, which are recognized to control P extractability.

There are few researches focusing on P extractability via SCE and its variation following the composting of agro- and bio-waste digestates, especially in the light of their P behavior in soil and on plant. In this study, we used the protocol suggested by Dou et al. (2000). While this procedure is highly informative, a certain extent of noise related to the whole P recovery is often reported in the literature (Turner and Leytem 2004). Pagliari and Laboski (2012) have reported a 20% over-estimation in the P recovery from dairy and chicken manure fractionation. In our study, we found some P recovery overestimation; however, it was in a similar range to previously reported recovery of recycled organic fertilizer (Pagliari and Laboski 2012). The data from this work showed  $AW_D$  was dominated by labile-P, almost doubling  $BW_D$  ( $\approx 85$  vs.  $\approx 45\%$ ). As suggested by Bachmann et al. (2016), labile-P is considered easily available P source for plants; it is recognized to be formed by water soluble free-P (DNA, simple phosphate monoesters), along with by P forms weakly sorbed on Ca and Fe (and Al) compounds extractable in mild solution (sodium bicarbonate). The Ca:P ratio along with Fe:P (and Al:P) of the compared products was in general agreement with these outcomes. The compared digestates showed also very different alkali extractable P (NaOH-P) and  $BW_D$  doubling  $AW_D$  (25 vs. 12%). This fraction is generally ascribed to inorganic/organic metal-bound (Fe-, Al-) P forms, generally considered long-term

**Table 4** Dry biomass (DW), phosphorous (P) content, and uptake in ryegrass shoots at four successive harvests (days 28–112) and root at final harvest (day 112)

Treatment	Days after sowing							
	Shoot				Mean	Sum	Root	Total (shoot + root)
	28	56	84	112	0–112	0–112	112	0–112
DW (g)								
Ctrl	0.76	1.08	1.13	0.92	0.97 c	3.89 b	2.49 a	6.38 ab
AW <sub>D</sub>	0.80	1.08	1.15	1.59	1.15 b	4.62 ab	2.45 ab	7.06 ab
BW <sub>D</sub>	0.78	1.01	1.19	1.07	1.01 bc	4.05 b	2.04 ab	6.09 b
AW <sub>C</sub>	1.05	1.55	1.34	1.41	1.34 a	5.35 a	2.30 ab	7.64 a
BW <sub>C</sub>	0.76	1.20	0.98	1.23	1.04 bc	4.17 b	2.33 ab	6.50 ab
MSWC	0.90	1.20	1.14	1.13	1.09 bc	4.36 b	1.83 b	6.19 ab
P-Chem	0.81	1.18	1.15	1.38	1.13 bc	4.53 ab	1.96 ab	6.49 ab
Mean	0.84 b	1.19 a	1.15 a	1.25 a				
P (mg g <sup>-1</sup> )								
Ctrl	1.35	1.09	0.89	0.88	1.05 d	–	0.86 c	–
AW <sub>D</sub>	2.03	1.48	1.45	1.15	1.53 a	–	1.57 a	–
BW <sub>D</sub>	1.54	1.23	1.14	0.87	1.19 cd	–	0.95 c	–
AW <sub>C</sub>	1.74	1.81	1.26	1.01	1.45 ab	–	1.47 a	–
BW <sub>C</sub>	1.52	1.18	0.94	1.01	1.16 cd	–	1.30 ab	–
MSWC	1.39	1.13	1.21	1.14	1.22 bd	–	1.15 bc	–
P-Chem	2.09	1.44	1.01	0.80	1.33 ac	–	1.06 bc	–
Mean	1.67 a	1.34 b	1.13 c	0.98 c				
P uptake (mg pot <sup>-1</sup> )								
Ctrl	1.03	1.16	1.00	0.81	1.00 d	4.00 c	2.15 bc	6.15 c
AW <sub>D</sub>	1.62	1.57	1.67	1.87	1.68 ab	6.73 ab	3.85 a	10.58 ab
BW <sub>D</sub>	1.19	1.23	1.38	0.94	1.18 cd	4.74 bc	1.95 c	6.69 c
AW <sub>C</sub>	1.82	2.85	1.67	1.42	1.94 a	7.76 a	3.38 a	11.14 a
BW <sub>C</sub>	1.16	1.41	0.93	1.24	1.19 cd	4.74 bc	3.01 ab	7.75 bc
MSWC	1.25	1.35	1.38	1.28	1.32 bd	5.27 bc	2.10 bc	7.37 bc
P-Chem	1.69	1.73	1.17	1.09	1.42 bc	5.69 bc	2.08 c	7.76 bc
Mean	1.39 ab	1.62 a	1.31 b	1.24 b				

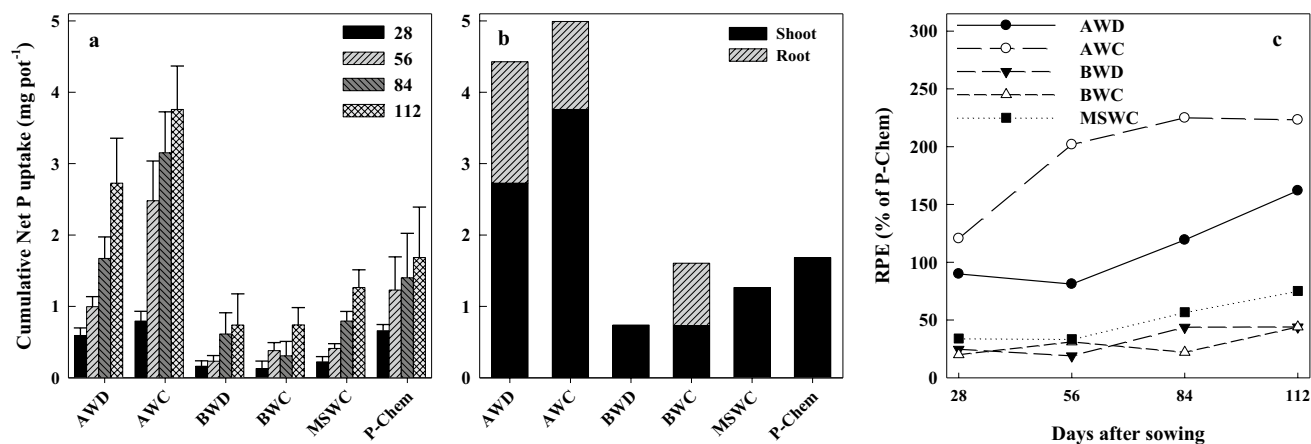
A one-way ANOVA was applied to harvest, cumulated harvests, and root data; in each column and for each trait, different letter intervals indicate statistically different mean data according to Tukey test ( $p < 0.05$ )

Ctrl unamended soil, AW<sub>D</sub> digestate from the wet digestion of agro-waste, BW<sub>D</sub> digestate from the dry-batch digestion of bio-waste, AW<sub>C</sub> compost from AW<sub>D</sub>, BW<sub>C</sub> compost from BW<sub>D</sub>, MSWC reference compost, P-Chem chemical reference (Ca(H<sub>2</sub>PO<sub>4</sub>)<sub>2</sub>)

plant available (Bachmann et al. 2016). These data are in agreement with the different Fe:P (and Al:P) ratio from the different tested products, thus showing BW<sub>D</sub> the highest Fe (and Al) content, this promoting the formation of metal-P compounds, very likely. At last, the compared digestates showed also variation in the acid extractable P, this representing the most important fraction in BW<sub>D</sub>, with five-fold HCl-P in comparison to AW<sub>D</sub> ( $\approx 50$  vs.  $\approx 10\%$ ). HCl-P is considered sparingly soluble in calcareous soil and not available for plant nutrition (Bachmann et al. 2016; Grigatti et al. 2017a). The formation of sparingly soluble Ca-P compounds detectable in the HCl-P fraction is recognized to be mainly regulated by the Ca:P ratio of samples, being the Ca and P content (and their ratio) from the two digestates in agreement with this assumption.

The composting process produced an overall reduction of H<sub>2</sub>O-P in the final products compared to raw anaerobic digestates (-25%, on average); this is in agreement with the findings of Wei et al. (2015).

Besides SCE, the NaOH+EDTA extraction can be helpful to assess the inorganic/organic P from recycled OM (Stutter 2015; Huang and Tang 2015); despite the great amount of organic matter in the samples tested in this work the NaOH+EDTA extraction showed the inorganic P was the dominant form in both Ads, regardless of feedstock. These data are in agreement to what described for agro-waste digestates by Bachmann et al. (2016) and about bio-waste digestate by Grigatti et al. (2020). A general prevalence of inorganic P was showed also by Stutter (2015) and by Möller (2018) on a wide range of OW and by Mazzini et al.



**Fig. 4** Cumulative phosphorous apparent net-uptake of ryegrass shoots in four subsequent harvests during pot trial (a), cumulated apparent net-uptake in plant biomass (shoots and roots) at final harvest (b), and RPE<sub>tissue</sub> (c). AW<sub>D</sub>, digestate from the wet digestion of

agro-waste; BW<sub>D</sub>, digestate from the dry-batch digestions of bio-waste; AW<sub>C</sub>, compost from AW<sub>D</sub>; BW<sub>C</sub>, compost from BW<sub>D</sub>; MSWC, reference compost; P-Chem, Ca(H<sub>2</sub>PO<sub>4</sub>)<sub>2</sub>. Error bars, SE ( $n = 4$ )

(2020), on many different types of Ads. Besides this, the data showed that composting poorly modified the whole  $P_i:P_o$  ratio. In this framework, solution <sup>31</sup>P-NMR can be very helpful to assess the different P forms from the studied organic samples. Although there is no general agreement about P forms in these products, solution <sup>31</sup>P-NMR has been widely utilized for the characterization of different types of recycled organic waste (Turner and Leytem 2004; Hansen et al. 2004; Toor et al. 2006). In our work, solution <sup>31</sup>P-NMR showed that all the compared samples had a limited ranging of signals being these comprise between 5.97 and -4.64 ppm, to a deeper insight all the analyzed organic products (AW<sub>D</sub>, AW<sub>C</sub>, BW<sub>D</sub>, and BW<sub>C</sub>), and exhibited an intense contribution from orthophosphate (5.80; -6.05 ppm), being very similar also to the reference compost (MSWC). These findings are in agreement with the literature on these topics, in which there are many reports of different organic samples analyzed via <sup>31</sup>P-NMR (Turner and Leytem 2004; Cade-Menun, 2005; Toor et al. 2006; Stutter 2015). It is also in agreement with Preston et al. (1998), who reported the relatively high proportion of orthophosphate P and the lack of detectable polyphosphate or phosphonate, ascribing this phenomenon the high levels of both microbial activity and availability of P. The pyrophosphate signal (-4.56; -4.64 ppm) was observed in all spectra, with the exception of the reference compost (MSWC). In addition, samples derived from agro-waste (AW<sub>D</sub> and AW<sub>C</sub>) showed some peaks which can be assigned to phytic acid according to Cade-Menun (2005). Especially, the peak detected at about 5.70 ppm is considered discriminant for phytic acid (Hansen et al. 2004). This peak was clearly detectable in products based on energy-crop/agricultural-residues/animal-slurry; as reported in literature (Hansen et al. 2004), phytic acid is the most important P form in cereals used to feed animals.

All these different P forms and their different extractability turned in a different potential and actual plant available P.

In this frame, the soil test showed agro- and bio-waste behaved oppositely being the Olsen-P the best in the former case. Later on throughout the soil incubation, a rapid P fixation in all the compared treatments was detectable. These outcomes resulted in agreement with the P fractionation previously discussed; in fact, both AW<sub>D</sub> and AW<sub>C</sub> were characterized by high labile-P, which is considered short term available. The soil incubation results were also in agreement with the NaOH-P outcomes from the SCE of the tested organic products. NaOH-P is recognized to become available in the middle-long term, being this the reason of the BW<sub>D</sub> and BW<sub>C</sub> middle-term and long lasting P release in comparison to P-Chem, very likely. Besides this, data were in general agreement also to the acid extractable P (HCl-P), being this fraction considered unavailable for plant. In this context, the agronomical test conducted in pot in a controlled environment (temperature; moisture; light), in a N non-limiting environment deeply stresses the P plant uptake capacity. In this framework, the utilization of poor available-P soil (Olsen-P  $\leq 3$  mg kg<sup>-1</sup>) resulted helpful to reduce the background giving a favorable environment for this study. Along with this, the comparison with a (soluble-) chemical and an organic P source (P-Chem and MSWC) can represent valuable references to the P release from the different products compared in this works (AW<sub>D</sub>, BW<sub>D</sub>, AW<sub>C</sub>, and BW<sub>C</sub>). In this study, the apparent relative efficiency of the chemical P source was lower than generally reported about chemical P fertilizers this ranging between 10 and 25% (Roberts and Johnston 2015). These results are in agreement with the strong fixing capacity of the selected soil: this showing 60% reduction of available P within two weeks from P-Chem as previously discussed. In this context, the analysis of the relative P efficiency, from the

different treatments can give a valuable insight to the plant P utilization from the compared products as potential substitutes for mined P. The results from the pot test clearly showed highly- ( $AW_D$  and  $AW_C$ ) and poorly- ( $BW_D$  and  $BW_C$ ) P efficient products, with the reference compost (MSWC) intermediate. These results are in a general agreement with the SCE outcomes that proved very different labile-P from agro- and bio-waste paths (84 vs. 48%).

It is widely recognized for crop cultivation that a coupled rapid- and long-lasting P release can satisfy the initial “starter” plant P requirement and the whole requirements during the different growth stages, showing also a reduction of P availability in the latest stage of plant maturity (Steele et al. 1981; Clarke et al. 1990). In this light composting, the agro-waste digestate appears a winning strategy to obtain a final product with low  $H_2O$ -P available for possible leaching, resulting also in a more plant-synchronized P release. Oppositely, the bio-waste derived products performed constantly poor  $RPE_{tissue}$ , being lower than the reference compost. Both bio-waste anaerobic digestate and compost hardly reached 40% relative efficiency output in comparison to the chemical P source. These results are in agreement to what exposed about the P fractionation and soil test of these products because they show moderate plant available P from the bio-waste path mainly due to the high Ca:P ratio together with the high Fe:P and Al:P ratios.

## 5 Conclusions

The raw digestates compared in this work showed very different characteristics (biological stability; total phosphorus), along with very different phosphorus fractionation, these differences mainly derived from the feedstock characteristics (calcium to phosphorus and iron to phosphorus ratios). The agro-waste digestate showed the highest labile phosphorus, almost doubling the bio-waste one. In this framework composting reduced the water-soluble phosphorus (potentially leachable phosphorus), also align the carbon to nitrogen ratio and biological stability to the standards for the agronomically safe utilization of this type of recycled organic waste. Oppositely composting poorly affected the inorganic/organic phosphorous ratio being the orthophosphate prevailing in all the tested products, as proven by solution  $^{31}P$ -NMR. The soil and pot test generally confirmed the phosphorus fractionation, showing compost from the agro-waste digestate as the best P releaser. The anaerobic digestion followed by the composting appeared to be strategic for the management, of both agro- and bio-waste thus reducing environmental risks. Their agronomical utilization can be successful improved by taking in account for sequential chemical extraction outcomes, fitting their application on the basis of their potential phosphorus release, according to the soil phosphorus status and to the expected plant phosphorus requests.

**Supplementary Information** The online version contains supplementary material available at <https://doi.org/10.1007/s42729-023-01274-8>.

**Funding** Open access funding provided by Alma Mater Studiorum - Università di Bologna within the CRUI-CARE Agreement.

## Declarations

**Competing Interests** The authors declare no competing interests.

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