nature sustainability

ANALYSIS

https://doi.org/10.1038/s41893-019-0416-x

Environmental and health co-benefits for advanced phosphorus recovery

Davide Tonini, Hans G. M. Saveyn and Dries Huygens*

Worldwide food production is largely dependent on rock phosphate, a finite raw material used for the production of concentrated phosphorus fertilizers. With the aim to close the biogeochemical phosphorus cycle across regions and urban–rural systems, advanced phosphorus recovery applies thermochemical and precipitation techniques to transform locally available biogenic materials into concentrated phosphorus fertilizers. Due to insufficient insights into the consequential impacts of these circular processes, opportunities to align advanced phosphorus recovery with agricultural sustainability are still widely unknown. Here we show that environmental and health life cycle impacts are often lower for phosphorus fertilizers sourced from secondary raw materials than for rock phosphate-derived products, especially in areas of high livestock and population density. Including externalities from rock phosphate extraction and avoided current-day management of biogenic materials in the comparative product life cycle severely alters the cost assessment relative to an analysis that considers only internal costs from manufacturers' production processes. Societal costs incurred for circular products derived from sewage sludge, manure and meat and bone meal are up to 81%, 50% and 10% lower than for rock-derived superphosphate, respectively. Even without accounting for rock phosphate depletion risks, short-term and local environmental and health co-benefits might underlie the societal cost effectiveness of advanced phosphorus recovery.

A sustained phosphorus (P) supply for agriculture is not assured[1–6]. Among other options, the circular economy may contribute to increased P-use efficiency and a more sustainable food production system[2,3,7,8]. Technologies are being developed that transform biogenic materials into concentrated P fertilizers that are low in contaminants through precipitation processes and thermochemical manufacturing processes under oxidative or reductive conditions[9]. Such advanced P recovery processes offer opportunities to close the P biogeochemical cycle at regional (for example, long-distance transport of excess P) and system (for example, urban–rural interface)[10] levels while simultaneously controlling the possible public health risks associated with the rising amounts of biogenic materials[11]. It is expected that the valorization of recovered P from biogenic materials through those advanced P recovery pathways might substitute 17–31% of the mined rock phosphate P fertilizers by the year 2030 in the European Union (EU), mainly as a result of changing policies on circular economy, waste management and agricultural stewardship[9].

The materialization of such meaningful shifts in manufacturing processes, sourcing strategies and resource consumption might considerably alter the environmental and human health impacts, both 'upstream' and 'downstream' of a P fertilizer manufacturer[12]. While a core aim of advanced technologies that repackage P into concentrated P fertilizers is to decouple end P users from source risk, conceptual frameworks often refer to possible environmental and monetary co-benefits of this circular economy model[5,13–15]. Avoided costly and energy-intensive transport of manure, mitigated eutrophication, economic savings from energy recovery and reduced waste management costs are recurring elements in the framing of P recovery[5,13–15]. Nonetheless, case studies applying life cycle assessments for advanced P recovery from municipal wastewaters[16–18] and food waste[19] in Europe have identified trade-offs between impact categories. Pradel and Aissani[20] and Golroudbary et al.[21] even suggested overall environmental burdens for circular P fertilizers relative to mined rock phosphate pathways. Economic analyses indicate higher internal costs for the manufacturing of most P recovery pathways from municipal wastewaters relative to mined P fertilizers[22,23]. Overall, a need exists to identify implementation opportunities and technology options that maximize the socioenvironmental co-benefits, other than safeguarding rock phosphate reserves, of emerging P recovery pathways.

The circular economy entails a product-oriented approach

Here we evaluate concentrated P fertilizers as functionally equivalent products using a standardized life cycle assessment (LCA) methodology[24,25]. The system is approached from a product perspective, and the production and use on land of 1 kg of bioavailable P in a concentrated P fertilizer (≥4% P) is used as the functional unit for this LCA (Fig. 1). The choice of the functional unit allows us to compare impacts for concentrated P fertilizers produced in the linear and the circular economies because the manufacturing processes share the same type of end product (similar to Pradel and Aissani[20]; Supplementary Methods). Advanced P recovery (AP) involves the production and use of a concentrated P fertilizer from biogenic feedstocks and displaces the combined functions of manufacturing and using a concentrated rock-based P fertilizer (RP) and the current-day management and use of a biogenic feedstock in the business-as-usual life cycle (CF) (Fig. 1a). In other words, to enable a consistent comparison between circular and linear concentrated P fertilizer production systems, the current-day feedstock management and use of the feedstock is considered a displaced activity. The net balance (NB), including the shifted feedstock management from the implementation of advanced P recovery, is thus calculated as NB = AP – CF, and the resulting impacts can be compared with RP (Fig. 1a). To better understand the complex impacts of CF, these results are presented in a disaggregated manner in addition to the

European Commission, Joint Research Centre, Directorate Growth and Innovation, Circular Economy and Industrial Leadership Unit, Seville, Spain.
*e-mail: Dries.HUYGENS@ec.europa.eu

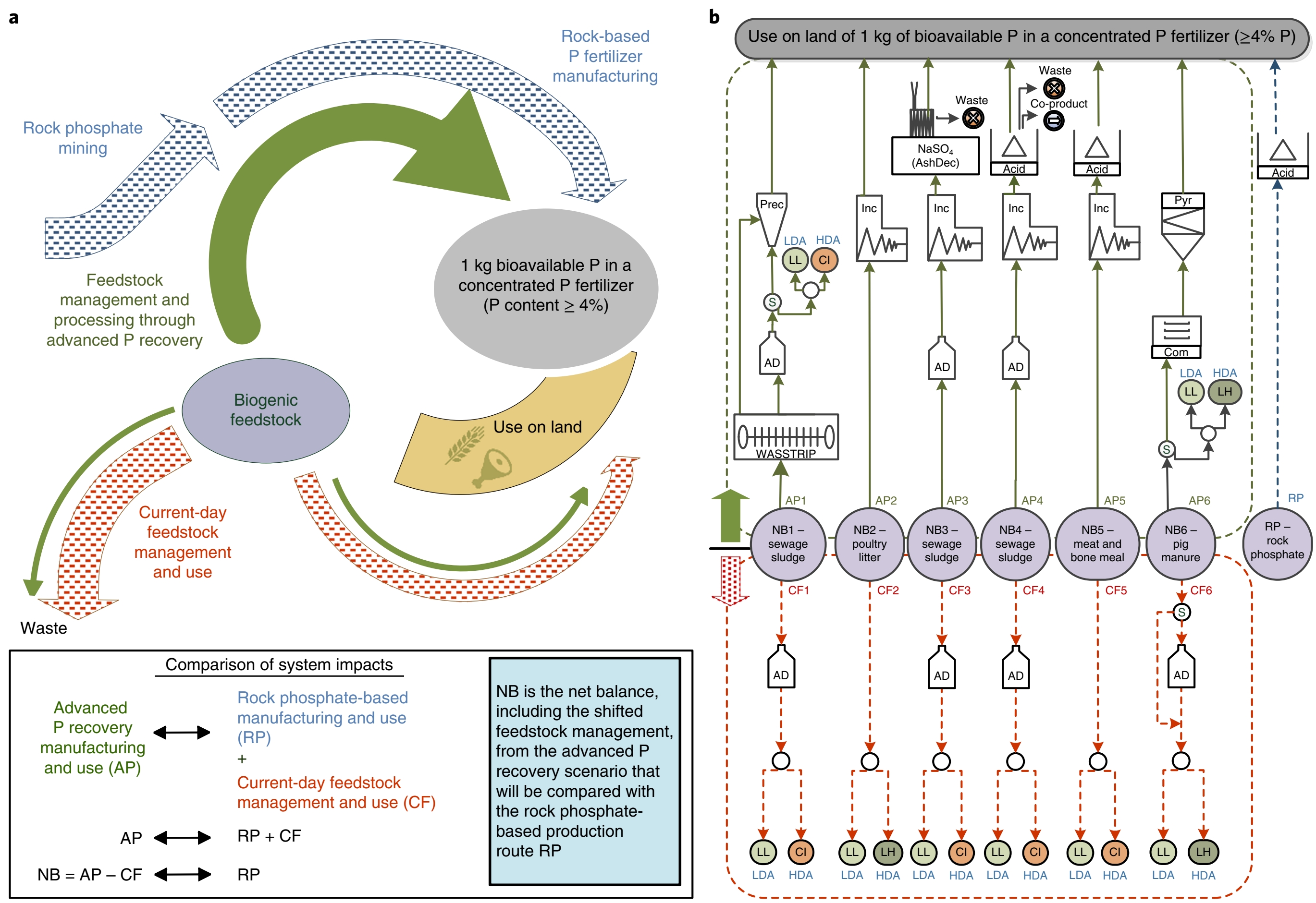

Fig. 1 | Schematic representation of methodological principles and selected pathways of the life cycle assessment that applies the production and use on land of 1 kg of bioavailable P in a concentrated P fertilizer as the functional unit. a, Schematic representation of AP (solid green colours; possibly including waste production and partial landspreading as non-concentrated organic fertilizing materials) and business-as-usual (shaded colours) life cycle systems as two comparable individual systems for the production of concentrated P fertilizers. Advanced P recovery produces a concentrated P fertilizer from biogenic feedstocks and displaces the combined activities of manufacturing a concentrated P fertilizer from rock phosphate and the management of a biogenic feedstock in the business-as-usual life cycle. To be functionally equivalent, life cycle impacts for AP (green arrows) should therefore be compared with the summed impacts from RP (blue-shaded arrows) and CF (red-shaded arrows). **b**, Schematic representation of the seven production options for P fertilizers through advanced P recovery routes (NB1–NB6) and through the acidulation of RP. The net impact balance NB1–NB6 is obtained from the difference between the advanced P recovery manufacturing and use processes (AP1–AP6; upper half) and the current-day feedstock management and use (CF1–CF6; lower half) (**a**; the blue letters indicate the use routes for biogenic materials within high population and livestock density areas (HDA) and low population and livestock density areas (LDA) as follows: LL and LH, land application at low and high application rates, respectively; CI, co-incineration followed by disposal of the ashes; S, solid–liquid separation; AD, anaerobic digestion; prec, precipitation of phosphate salts; inc, mono-incineration; pyr, slow pyrolysis; acid, acidulation process; com, composting; WASSTRIP, Waste Activated Sludge Stripping to Remove Internal Phosphorus).

impacts of AP. The consequential life cycle approach also considers the displacement of conventional market products (for example, mineral nitrogen–phosphorus–potassium (NPK) fertilizers and electricity; Supplementary Methods).

The LCA presented is based on the assessment of selected advanced P recovery manufacturing processes and selected current-day management practices for biogenic materials, commonly applied in the EU ('scenario modelling'; Methods). The selection of advanced P fertilizer pathways was based on the outcome of a participatory policy preparation process that identified examples of feedstock-process technology combinations with high market potential based on technological readiness level (TRL 7–9), feedstock availability, market and consumer confidence, and legislative and policy developments[9]. They centre on precipitation, thermochemical conversion and/or acidulation as core technologies and apply manure, sewage sludge, meat and bone meal or rock phosphate as input materials (Fig. 1b; Supplementary Notes and Supplementary Tables 8 and 9). Concentrated P fertilizers are applied at all times as value-added materials on cultivated soils. We assumed that current-day feedstock management and use practices vary between high and low population and livestock density areas in the EU (HDA and LDA, respectively; Methods) and are in line with applicable EU legislation (Methods). In HDA, sewage sludges and meat and bone meal are assumed to be discarded through co-incineration, whereas (digested) manures are spread on land. In practice, the maximal application rates of such manures in HDA is governed by the legal limit of 170 kg N ha^{-1} yr^{-1} applicable in nitrate vulnerable zones, in accordance with the European Nitrates Directive (91/676/EEC). This may result in low manure P-use efficiencies because the manure P supply that exceeds plant P demands was assumed to be lost to surface and groundwater bodies[3] (Methods). In LDA, biogenic

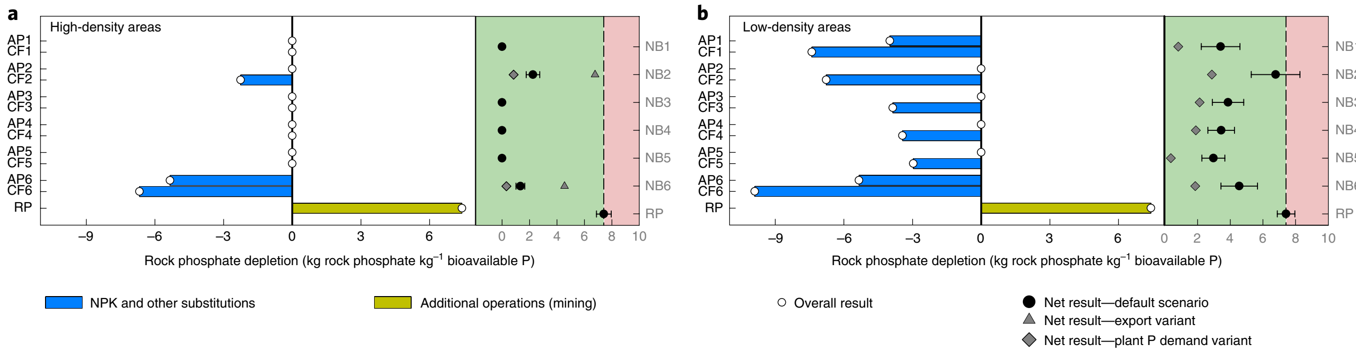

Fig. 2 | Rock phosphate depletion. a,**b**, Impacts on rock phosphate depletion in high population and livestock density areas (**a**) and low population and livestock density areas (**b**) resulting from the production of 1 kg of bioavailable P. The left side of each graph (white background) indicates the impacts for AP and CF, disaggregated for different process and life cycle stages, as well as impacts from RP. The right side of each graph indicates the overall pathway (NB = (AP – CF) and RP) results for the default scenario (mean; error bars indicate ±1 s.d.) and energy, export and plant P demand uncertainty variants when results are located outside the error bars for the default scenario. Pathways NB1–NB6 associated with a net saving relative to the RP pathway are displayed on the green background; pathways NB1–NB6 associated with a net burden relative to the RP pathway are displayed on the red background.

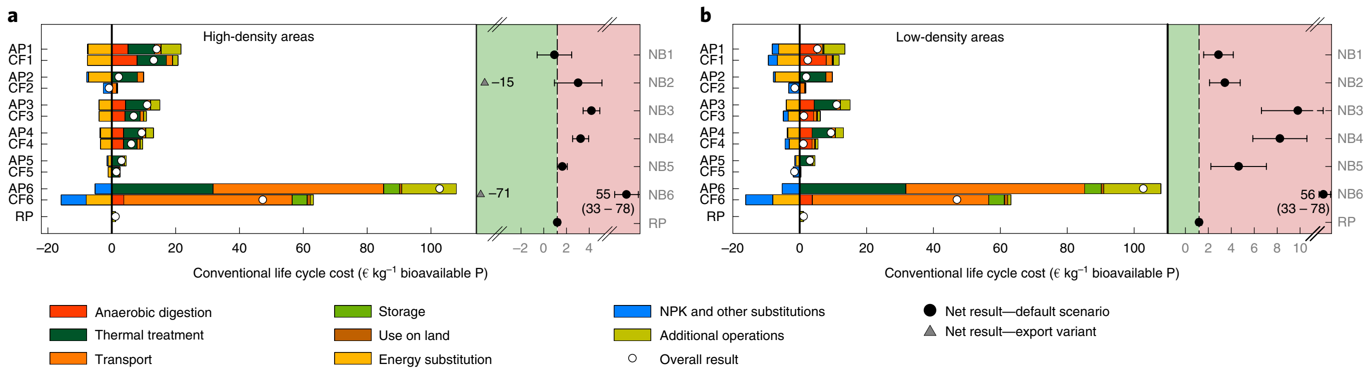

Fig. 3 | Conventional life cycle costing. a,**b**, Conventional life cycle costs in high population and livestock density areas (**a**) and low population and livestock density areas (**b**) resulting from the production of 1 kg of bioavailable P as described in Fig. 2. The left side of each graph (white background) indicates the impacts for AP and CF, disaggregated for different process and life cycle stages, as well as impacts from RP. The right side of each graph indicates the overall pathway (NB = (AP - CF) and RP) results for the default scenario (mean; error bars indicate ±1 s.d.), and energy, export and plant P demand uncertainty variants when results are located outside the error bars for the default scenario. Pathways NB1–NB6 associated with a net saving relative to the RP pathway are displayed on the green background; pathways NB1–NB6 associated with a net burden relative to the RP pathway are displayed on the red background.

materials are assumed to be spread on land at low application rates (Methods). The bioavailable P fraction was assumed to be material specific with default values of 85–100% for the concentrated P fertilizers[26] and 40–85% for the organic fertilizing materials (biogenic feedstocks, or a fraction thereof, that are returned to agricultural land as non-concentrated P materials with a P content <4%)[27,28], respectively (Supplementary Notes). As part of the uncertainty analysis, system variants have been modelled on the basis of (1) a different energy source (natural gas instead of the default energy mix that includes a higher share of renewables; 'energy variant' in Figs. 4 and 5), (2) manure exports from HDA, a common practice for manure excess in the Netherlands, for example (NB2 and NB6; dewatering to reduce manure volumes followed by long-distance transport to nutrient-deficient soils instead of default land application on nearby soils with a nutrient surplus; 'export variant' in Figs. 2–5), and (3) landspreading of (digested) manure fractions on soils cultivated with crops with a low P demand (plant demands of 6.5 kg P ha^{-1} as documented for specific crops such as rice, grain legumes and sweet potato[29] instead of default values of 17.5 kg P ha^{-1}; 'plant P demand variant' in Fig. 2).

Results

Rock phosphate depletion. The RP pathway applied to produce 1 kg of bioavailable P as single superphosphate (SSP) involves the extraction of 7.4 kg of rock phosphate (Fig. 2). The current-day feedstock management and use pathways (CF1–CF6) may involve a P fertilizing function and hence a partial avoidance of rock-derived mineral P fertilizer use (Fig. 2; NPK and other substitutions). As a result, the balances NB1–NB6 may result in a depletion of rock phosphate, albeit always lower than RP (Fig. 2). Balances NB1–NB6 result in a lower depletion in HDA (0.0–2.3 kg of rock phosphate kg^{-1} bioavailable P) than in LDA (3.0–6.7 kg of rock phosphate kg^{-1} bioavailable P) (Fig. 2). In HDA, the greatest net savings are observed when no P is returned to land due to feedstock co-incineration in CF (Fig. 2a). In LDA, greater net savings can be obtained for the NBs that source sewage sludge and meat and bone meal compared with those sourcing manure, because of greater improvements in plant P bioavailability when shifting feedstock management from CF to AP (Fig. 2b). When organic fertilizing materials are applied to plants with a low P demand in CF, the reduced amount of displaced mineral P fertilizer leads to overall greater rock savings (Fig. 2a; plant P demand

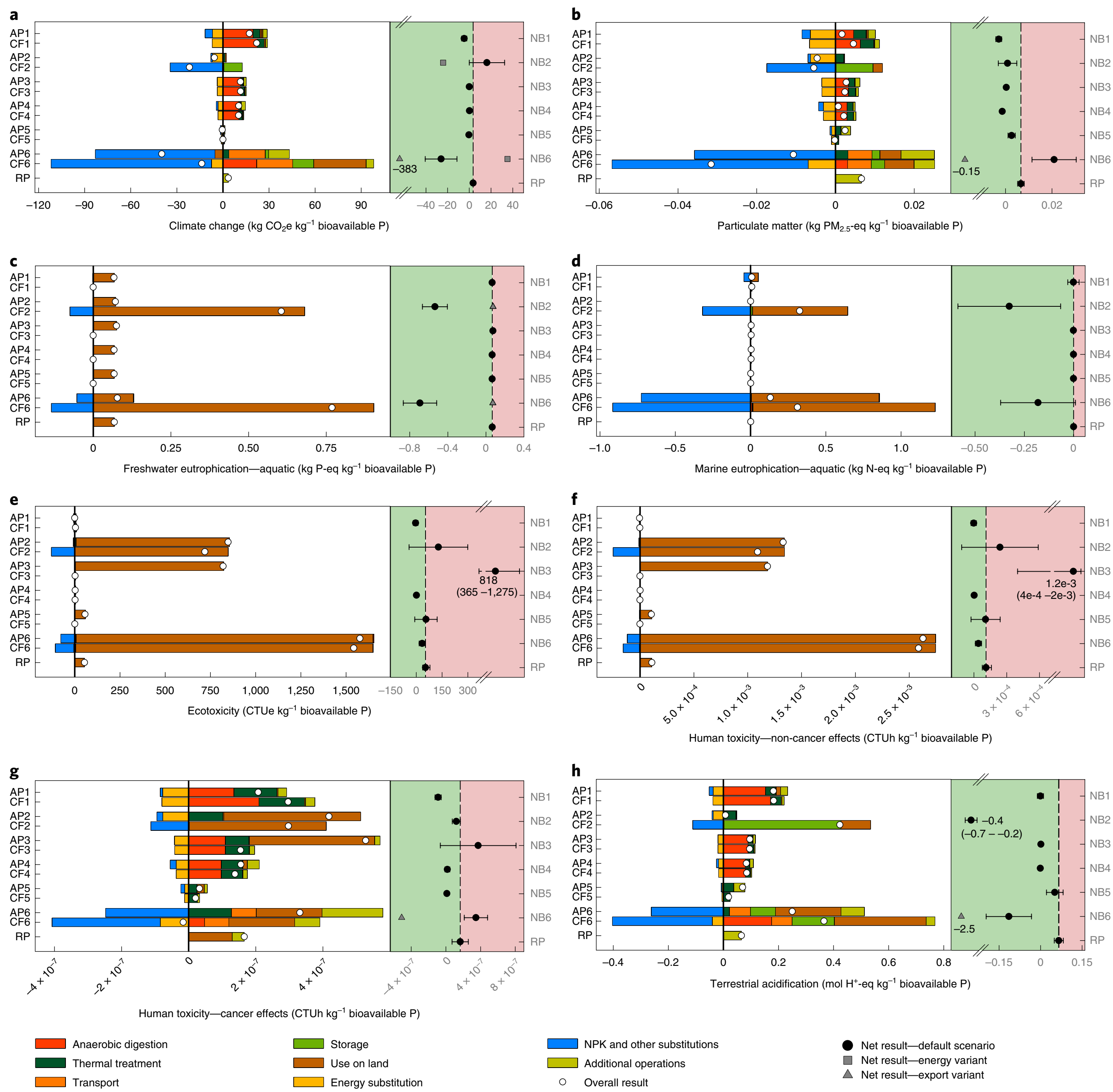

Fig. 4 | Environmental and health impacts. a–h, Environmental and health impacts for climate change (**a**), particulate matter (**b**), freshwater eutrophication—aquatic (**c**), marine eutrophication—aquatic (**d**), ecotoxicity (**e**), human toxicity—non-cancer (**f**), human toxicity—cancer (**g**) and terrestrial acidification (**h**) in high-density areas resulting from the production of 1 kg of bioavailable P as described in Fig. 2. The left side of each graph (white background) indicates the impacts for AP and CF, disaggregated for different process and life cycle stages, as well as impacts from RP. The right side of each graph indicates the overall pathway (NB = (AP − CF) and RP) results for the default scenario (mean; error bars indicate ±1 s.d.), and energy, export and plant P demand uncertainty variants when results are located outside the error bars for the default scenario. Pathways NB1–NB6 associated with a net saving relative to the RP pathway are displayed on the green background; pathways NB1–NB6 associated with a net burden relative to the RP pathway are displayed on the red background. CO_2e, CO_2 equivalent; CTUe, Comparative Toxic Unit, ecotoxicity potential; CTUh, Comparative Toxic Unit, human health potential.

variant). When plant P uptake from the manure is increased in CF relative to the default scenario after manure transport to nutrient-deficient soils, reduced rock phosphate savings are indicated (Fig. 2a; export variant).

Life cycle costing. The conventional life cycle cost for RP equals €1.2 kg^{-1} bioavailable P (Fig. 3). The cost under the default scenario is mostly greater for NB1–NB6 than for RP, excepting NB1 and NB5 in the HDA. Life cycle costs for NB1–NB5 are lower in HDA than in LDA, with values ranging from €−0.1 to €4.2 kg^{-1} bioavailable P and from €2.9 to €9.8 kg^{-1} bioavailable P, respectively (Fig. 3). The difference in CF between HDA and LDA mostly underlies cost results, with thermal treatment and monetary savings from mineral N (NPK and other substitutions) as the main contributors (Fig. 3).

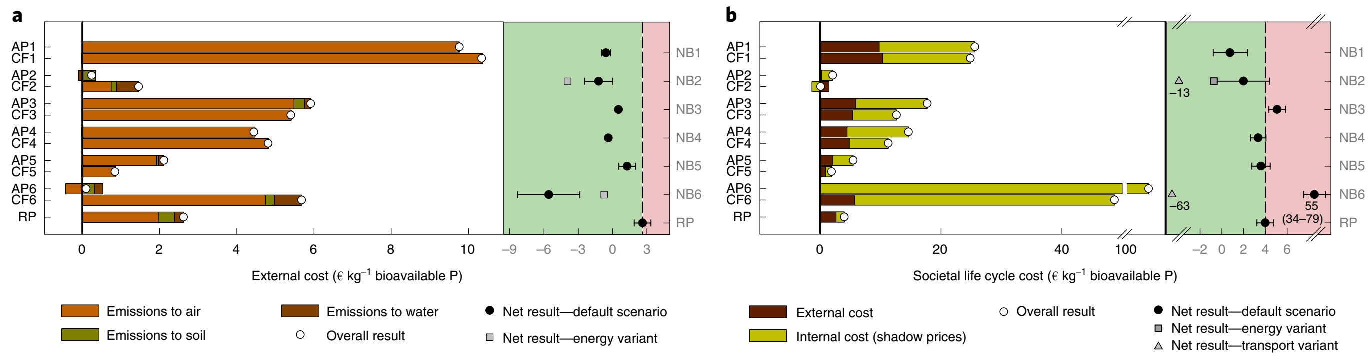

Fig. 5 | External and societal life cycle costs. a,**b**, External costs resulting from the emissions to air, soil and water (**a**) and societal life cycle costs as derived by summing the internal costs (conventional life cycle costs corrected for taxes and based on shadow prices) and external costs (**b**) in high-density areas resulting from the production of 1 kg of bioavailable P as described in Fig. 2. The left side of each graph (white background) indicates the impacts for AP and CF, as well as impacts from RP. The right side of each graph indicates the overall pathway (NB = (AP − CF) and RP) results for the default scenario (mean; error bars indicate ±1 s.d.), and energy, export and plant P demand uncertainty variants when results are located outside the error bars for the default scenario. Pathways NB1–NB6 associated with a net saving relative to the RP pathway are displayed on the green background; pathways NB1–NB6 associated with a net burden relative to the RP pathway are displayed on the red background.

Cost savings for NB1 are observed at the wastewater treatment plant thanks to the implementation of AP, due to reduced sludge volumes for dewatering, digestion and incineration relative to CF. The life cycle cost for NB6 is much higher than for NB1–NB5, mainly due to the high (capital) cost for pyrolysis, the lack of energy recovery and the reduced N substitution for AP relative to CF (Fig. 3). However, when manure exports are assumed as in CF2 and CF6, the life cycle costs for NB2 and NB6 decrease significantly relative to the default scenario of local landspreading thanks to reduced transportation costs (Fig. 3a; export variant).

Environmental and human health impacts. The results for eight relevant impact categories are presented for HDA, jointly covering 97–100% of the person-equivalent normalized impacts across all impact categories initially considered (Supplementary Notes and Supplementary Tables 4–7). Results for LDA and the results of the discernibility analysis are presented in Supplementary Notes (see Supplementary Fig. 1 and Supplementary Table 1).

In HDA, climate change impacts for NB1–NB6 are mostly lower than for RP, with the exception of NB2 (Fig. 4a). Burdens resulting from supplementary manufacturing steps in AP relative to CF are mostly counterbalanced by increased savings from N substitutions (NB1) and co-products (NB4), reduced needs for downstream sludge processing (NB1), reduced chemical demand during thermal treatment (NB3, NB4), reduced nitrous oxide (N_2O) emissions during storage or the use-on-land phase (NB2, NB6) and increased carbon storage in the soil (NB6) for AP compared with CF (Fig. 4a). Because thermal processes remove bioavailable N, substantially higher credits for NPK substitution are observed for CF2 and CF6 (landspreading of manure) relative to AP2 and AP6 (incineration and pyrolysis of manure) (Fig. 4a; NPK and other substitutions). The energy production from poultry litter incineration in AP2 results in substantial savings for NB2, but NB2 still has a slightly higher overall burden relative to RP in the default scenario.

Particulate matter impacts (Fig. 4b) are largely correlated with climate change impacts because energy production is a main contributor to both impact categories. The performance of the pathways that source manure (NB2 and NB6) relative to RP is, however, inverted relative to climate change. NB2 performs well due to reductions in ammonia (NH_3) emissions during storage and use on land for AP2 (manure incineration) relative to CF2 (landspreading). Emissions from mining operations (RP) and sulfuric acid production and use (RP and AP5) contribute to the burdens observed for RP and NB5.

Net savings on freshwater and marine eutrophication in HDA are observed for the manure-based NB2 and NB6 relative to RP (Fig. 4c,d). Manure incineration (AP2) produces a concentrated N-free P fertilizer that features minor nutrient losses to water bodies, equal to RP (Fig. 4c,d; AP2). Advanced P recovery avoids current-day feedstock management (CF2) that involves manure landspreading on P-saturated soils, resulting in leaching and run-off of P and N (Fig. 4c,d; CF2). While the landspread manure in CF displaces the use of mineral fertilizers and their associated eutrophication impacts (Fig. 4c,d; NPK and other substitutions), an overall net burden for CF2 is nevertheless observed (Fig. 4c,d). A similar reasoning as for NB2 can be applied for NB6, although AP6 and CF6 are associated with supplementary impacts relative to AP2 and CF2 resulting from the use on land of the separated liquid manure fraction. No effects relative to RP have been observed on freshwater and marine eutrophication for NB1, NB3, NB4 and NB5 because the impacts for AP are independent of the chemical composition of the concentrated P fertilizer, and zero eutrophication impacts are observed when the feedstock is incinerated in CF (Fig. 4c,d).

The impacts for ecotoxicity (Fig. 4e) and human toxicity—non-cancer effects (Fig. 4f) in HDA follow parallel trends, almost exclusively determined by the metal content of the products/materials applied on land and the displaced mineral NPK (Fig. 4e,f; use on land and NPK and other substitutions). Net savings are observed for NB1, NB4 and NB6 relative to RP, whereas NB2 and especially NB3 show a higher burden than RP. The concentrated P fertilizers of low metal content produced through AP1 and AP4 (Supplementary Notes and Supplementary Table 10) show negligible impacts, similar to CF1 and CF4 (incineration followed by disposal; Fig. 4e,f). This stands in contrast with balances NB3 and NB5, where metals in sewage sludge and meat and bone meal are largely transferred to the concentrated P fertilizer in AP but not returned to land after co-incineration in CF (Fig. 4e,f). For the manure-based NB2 and NB6, thermal processing results in lower NPK substitution credits in AP compared with CF since the field-applied manure has a higher displacement of mineral N fertilizer and associated ecotoxicological and non-carcinogenic air emissions resulting from its production (Fig. 4e,f).

For human toxicity—cancer effects in HDA, net savings are observed for NB1, NB2, NB4 and NB5 relative to RP (Fig. 4g). The impacts for RP are mostly due to the presence of cadmium in the fertilizer material that is applied on land (Supplementary Notes and Supplementary Table 10). The reduced sludge volumes to be processed in AP1 relative to CF1 result in reduced net emissions from anaerobic digestion and incineration for NB1. For NB2 and NB4, the emissions associated with thermal processing and other manufacturing steps in AP relative to CF are counterbalanced by reduced emissions during the use on land phase (NB2) and credits for co-products (NB4). NB3 and NB6 are associated with a net burden relative to RP due to the higher metal return to land in AP3 than in CF3 and the supplementary impacts of composting and thermal treatment in AP6 relative to CF6.

For terrestrial acidification, net burdens are observed for NB5 and RP, mostly due to the use of sulfuric acid during phosphate ore processing and/or posterior acidulation (Fig. 4h). The greatest savings are observed for the manure-based NB2 and NB6 due to a decrease in NH_3 emissions during manure storage and use on land for AP2 and AP6 relative to CF2 and CF6. Negligible impacts on terrestrial acidification are observed for NB1, NB3 and NB4.

The system variant analysis indicated that when export of pig manure slurries is assumed in CF, savings for NB6 relative to the default scenario are obtained for climate change, particulate matter, human toxicity—cancer and terrestrial acidification (Fig. 4; export variant). The export of poultry litter and pig manure, however, counteracts the savings for freshwater eutrophication in the default scenario, as P in excess of the plant demand accumulates in the nutrient-deficient soil (Fig. 4c). Applying natural gas as an energy source shifts the overall result for the energy-producing NB2 to a net saving, but to an increased burden for the energy-consuming NB6 (Fig. 4a; energy variant). Across impact categories, the largest uncertainties are observed for marine eutrophication when CF involves manure landspreading (NB2, NB6) and for the toxicity categories when AP incurs a significant return of sewage sludge metals to land (NB3). The main parameters contributing to those uncertainties are use on land and feedstock composition (Supplementary Notes and Supplementary Fig. 2c–f).

External and societal costs. The external costs for NB1–NB5 range from €−5.6 to €1.3 kg^{-1} bioavailable P and are thus lower than for RP (€2.6 kg^{-1} bioavailable P) in HDA (Fig. 5a). For RP, emissions to air (for example, particulate matter and sulfur dioxide) are the main contributors to the external cost. For NB1, NB2, NB4 and NB6, the impacts are lower for AP than for CF, mostly due to air emissions reductions resulting from mono-incineration relative to co-incineration. For NB3 and NB5, AP is associated with a higher external cost than CF due to increased metal emissions to the soil and emissions to air and water resulting from acidulation with sulfuric acid. Shifting from the default energy mix to natural gas as an energy source results in external cost reductions for the energy-producing NB2 but increases for the energy-consuming NB6 (Fig. 5a; energy variant).

When summing external and internal costs, the societal life cycle cost is lower (or comparable) for NB1–NB5 (€0.7 to €5.1 kg^{-1} bioavailable P) than for RP (€4.0 kg^{-1} bioavailable P) but much higher for NB6 (Fig. 5b). The robustness of the societal cost effectiveness for NB1–NB5 relative to RP is supported by the discernibility analysis results (Supplementary Notes). When manure export is assumed as CF, a significantly greater net societal cost saving relative to RP is obtained (Fig. 5b; export variant). Also, assuming natural gas as the energy source lowers the cost of NB2 (Fig. 5b; energy variant).

Discussion

In EU regions of high livestock and population density, sewage sludges[30] and meat and bone meal[31] are increasingly being removed from the agrifood chain for the sake of health protection (for example, concerns about the recycling of metals, antibiotics, biological pathogens and so on). Manure generated in HDA is often exported or applied locally in an inefficient manner due to the combined high P application rates and increased levels of P saturation in agricultural soils[32]. Relative to rock-based SSP production (RP pathway), the production of circular P fertilizers thus results in rock phosphate savings that are greater in HDA than in LDA (average savings of 6.8 kg and 3.2 kg rock phosphate kg^{-1} bioavailable P produced relative to rock-based SSP production, respectively; Fig. 2). Advanced P recovery can therefore support P circularity and reduce depletion risks of rock phosphate, a finite material that has no substitute from primary sources[4,6]. Moreover, the present high cost for current handling practices to deal with biogenic materials in HDA also entails opportunity costs, in terms of internal costs for manufacturing, for advanced P recovery pathways that are lower than in LDA (Fig. 3; conventional life cycle costs are on average €3.3 kg^{-1} bioavailable P lower in HDA than in LDA). Opportunities in LDA are smaller because the cheapest and oldest form of circular economy, local landspreading at low application rates, involves practices that are effective from a resource and cost perspective (Figs. 2 and 3). Overall, the implementation potential of advanced P recovery is positively related to the degree of P dissipation and the cost of the current-day feedstock management and use.

The impact hotspot analysis identified that the main elements contributing to impact savings relate to recycling P in an efficient manner in agriculture, reducing nutrient losses to water bodies, decreasing metal return to land and avoiding long-distance transport of high-volume materials. There are also risks of adverse and unintended negative effects, however, in advanced P recovery processes involving the removal of other valuable materials from the biogenic input materials and—to a minor extent—effects related to additional manufacturing steps with a high chemical or energy demand. Nitrogen removal and compensation by mineral N fertilizers during advanced P recovery processes have an adverse impact on global warming, human health and costs in the life cycle system[33], as the Haber-Bosch process for the production of mineral N fertilizers is an N_2O-emission-prone, costly and energy-demanding process[34,35]. The metal content of the P fertilizer and the system co-products that are returned to land is also of critical importance for human health and ecotoxicity effects. An engineering and design challenge for the industry sector is the development of cost- and energy-efficient technologies that preserve material value and contemplate the recycling potential of other valuable components, thus selectively isolating or transforming compounds along material transformation cascades for their subsequent disposal or further use as system co-products.

Reducing external and societal costs. Rock phosphate depletion is of great concern, but the decade-to-century horizon for the incidence of 'peak phosphorus'[36] and the lack of global governance on P scarcity[37] may limit dedicated efforts in the very short-term. Therefore, co-benefits from P recovery to society are particularly important because they are mainly short-term and local. Possibly, safeguarding rock phosphate reserves by means of advanced P recovery can be part of a strategy that moderates prevailing externalities resulting from mining and from food production and consumption systems. On the one hand, rock phosphate extraction and processing in dryers, calciners, grinders and posterior acidulation involves emissions to air, mostly in the form of fine rock dust and sulfur dioxide[38,39], that contribute to the high external costs for the rock phosphate-sourced SSP[38,39]. On the other hand, signs exist that agricultural systems may be reaching their limits[40] and negative externalities resulting from current agricultural practices have in particular been reported in regions with a high population and livestock density in the EU[41–43]. Factual examples in the EU that represent a risk of degrading water, air and soil quality include fertilizer run-offs and percolations contributing to reduced oxygen levels in water bodies[44,45], greenhouse gas emissions from agriculture that exacerbate climate change[46] and above-average metal accumulations in EU agricultural soils[47]. Externalities in linear and nutrient-leaky systems resulting from pollution are largely borne by society, rather than by nutrient users and polluters[48]. Notwithstanding the methodological challenges and uncertainties, external costs are real costs related to non-market items (lives, health, biodiversity, and so on)[49]. Hence, it is important to estimate these costs in monetary terms with the best methodologies and data available. In most cases, a greater external cost is indicated for current-day feedstock management than for the transformation of the same material into a concentrated P fertilizer (Fig. 5a). Therefore, the external costs of the balance resulting from the manufacturing and use of advanced P fertilizers are significantly lower than for the rock phosphate-based pathway (€−5.6 to €1.2 kg^{-1} bioavailable P for NB1–NB6 versus €2.6 kg^{-1} bioavailable P for RP). Short-term and local co-benefits of advanced P recovery in terms of reducing negative externalities are thus indicated. Internalizing external costs caused by pollution would enable value to be created out of dissipated nutrients and can support the economic case for P fertilizer manufacturing processes from secondary raw materials in a circular economy (Fig. 5b). The risks of rock phosphate depletion[50] and the benefits from the sanitation of manure[9] have not been monetarized in the present societal cost assessment but would further modify the balance in favour of these advanced technologies. Hence, advanced P fertilizer manufacturing processes can shift from a focus on the exclusive supply of 'products' to 'services' that combine the production of high-quality fertilizers with decreased waste generation and reduced emissions to air, soil and water bodies.

Dissenting views on sustainability. Our conclusions stand in contrast to other works[20,21,33] that challenge the overall sustainability of advanced P recovery. In our view, the dissenting views are mostly due to three issues. First, this study departs from the assumption that, at the moment of their production, biogenic materials are intermediate materials that continue to produce impacts along the rest of their life cycle (Fig. 1a). It is thus acknowledged that biogenic materials cannot cease to exist without further handling and resulting impacts. Our approach takes into consideration that the circular economy model not only produces new products but also fulfils a material or waste management function in the relevant European context (Fig. 1a). Most of the environmental and cost savings of the life cycle balances result from the avoided impacts of current-day feedstock handling and use. Second, this assessment is based on examples of advanced P recovery pathways that have been identified as showing a high market potential by relevant stakeholders through a participatory process. Third, this work, on top of carrying out a standardized LCA, also presents an overall societal cost, including the external costs of all relevant emissions (for example, dissipated phosphate, greenhouse gases, metals and particulate matter). Such an integrated approach avoids incurring market failures by overlooking impacts or other threats to the planet. On the basis of our approach, we conclude that the implementation of advanced P recovery is one element that may contribute to a new paradigm for P management[8] by providing a societal cost-effective avenue for the local production and supply of high-quality P fertilizers in geographic areas of high P dissipation.

Methods

Scope and functional unit. The functional unit of the study is the production and use on land of 1 kg of bioavailable P in a concentrated P fertilizer. Bioavailable P is defined as the sum of P that is immediately available to plants and P that can be transformed into an available form by naturally occurring processes in the short-term. The focus is on concentrated P fertilizers obtained from biogenic feedstocks through advanced P recovery, which are compared with SSP derived from mined rock phosphate. The assessment of the environmental aspects and potential impacts associated with a product was performed following International Organization for Standardization standards for LCA[24,25]. A consequential approach[51–53] was applied to capture the environmental consequences arising from the avoided current-day feedstock management and use (Fig. 1a) as well as the displacement of conventional market products (for example, NPK fertilizers and electricity). Hence, system expansion has been performed to ensure equivalence of functions across scenarios (Fig. 1a). The geographic scope of the study is the territory of the EU. In line with the reporting guidelines of the Intergovernmental Panel on Climate Change, the temporal scale boundary corresponds to a period of 100 yr, assuming continuous production of biogenic feedstocks and use on land. The impacts resulting from the transformation of carbon, nutrients (N, P and K) and metals (arsenic, cadmium, chromium, copper, mercury, nickel, lead, zinc) were considered in the foreground system of this LCA whereas biological pathogens, micronutrients (for example, magnesium) and other contaminants (for example, uranium) were not considered. The impact assessment was performed for the following impact categories, following the International Reference Life Cycle Data recommendations[54]: climate change[55]; terrestrial acidification[56]; photochemical ozone formation and particulate matter[57]; marine eutrophication—nitrogen[58] and freshwater eutrophication—phosphorus[58]; human toxicity cancer, non-cancer and ecotoxicity[59]; ozone depletion; infrared radiation; fossil resource depletion[60]. Water and abiotic resource depletion were not included due to insufficient inventory data. The assessment was carried out with the LCA tool EASETECH[61].

Life cycle costing. Alongside the impact assessment, conventional and societal life cycle costing (CLCC and SLCC, following naming principles introduced earlier[14,15]) were performed to quantify, respectively, financial and socioeconomic costs following the approach suggested in recent studies[62–64]. CLCC represents the traditional financial assessment accounting for marketed goods and services (internal costs). This is obtained as the sum of budget costs, accounted for in factor prices (market prices excluding transfers), and transfers (for example, taxes and subsidies)[65]. The internal costs of production may differ from the societal cost when specific actions have an impact on unrelated parties. As a result, there are differences between private returns or costs and the returns or costs to society as a whole. Those indirect costs could, for example, include healthcare costs resulting from particulate matter intake, or forgone production opportunities when pollution harms activities such as tourism. SLCC further internalizes these costs to inform on the true cost of the action to society[66] and thus represents a socioeconomic or welfare-economic assessment. The societal cost is obtained as the sum of budget costs, accounted for in shadow prices, and external costs[64]. To translate factor prices into shadow prices, the net transfer factor proposed by the Danish Ministry of Environment was used[65]. Compound-specific shadow prices to calculate external costs resulting from emissions to air, water and soil were taken from the Environmental Prices Handbook[49] reporting values for the Netherlands, assumed to be representative for HDA. Shadow prices for particulate matter emissions from rock phosphate mining were set to worldwide average values[67], being significantly lower than the Dutch values. In the absence of relevant and robust values, no societal cost assessment was made for LDA. The assessment was carried out with the LCA tool EASETECH[61], with LCA and LCC sharing the same system boundary in terms of treatment processes included and displaced products/activities.

Scenario modelling. Biogenic feedstocks are either processed through an advanced P recovery pathway ('advanced P recovery', AP1–AP6) or under business-as-usual practices in Europe ('current-day management and use', CF1–CF6) (Fig. 1b). The NB of the advanced circular systems for the production and use of 1 kg bioavailable P in a concentrated P fertilizer derived from secondary raw materials is then obtained as NB = AP − CF (Fig. 1a). These impacts from NB can be compared with the linear system for the production and use of 1 kg bioavailable P in a concentrated P fertilizer derived from RP (Fig. 1a). The approach assumes that the production of concentrated P fertilizers from secondary raw materials will not impact the generation of the biogenic material (the production of sewage, manure and meat and bone meal is not affected by the demand for P fertilizers). Advanced P recovery is modelled to cause no effects on upstream (for example, water quality after treatment) or downstream (for example, food production) systems. In line with recent studies[19,68–71], the applied consequential LCA approach intrinsically takes into consideration that advanced process implementation results in the displacement of current-day management and use of the feedstock. Therefore, it is unnecessary to value or apply allocation factors to the upstream production system of the biogenic material as upstream impacts are equal in AP and CF, and thus offset in NB. This approach circumvents possible discussions on whether a zero-burden assumption for waste-based materials is correct[20,72,73]. The LCA rests on closed mass and energy system balances (Supplementary Notes and Supplementary Tables 11–13).

Six advanced P recovery pathways of TRL 7–9 were assessed starting from sewage sludge (AP1, AP3, AP4), manure (AP2, AP6) and meat and bone meal (AP5) (Fig. 1b). The main technologies underlying the material transformation and fertilizer production processes include precipitation (AP1), incineration (AP2, AP3), incineration followed by acidulation (AP4, AP5) and slow pyrolysis (AP6) (Fig. 1b). Chemicals and reactants (for example, H_2SO_4, HCl) applied in the manufacturing processes were consistently assumed to be primary raw materials, instead of industrial by-products as often applied in fertilizer production. AP1 departs from biological sludge in biological wastewater treatment plants where phosphate, magnesium and potassium are separated before digestion and thickening due to the action of volatile fatty acids in a process called Waste Activated Sludge Stripping to Remove Internal Phosphorus (WASSTRIP)[74]. The separated nutrients are then sent together with the reject water from the sludge thickening after digestion to a precipitation reactor (Ostara Pearl), where the P fertilizer struvite is produced[74] (Fig. 1b). AP2 involves the incineration of poultry litter and the subsequent use of the resulting poultry litter ashes as a PK fertilizer on agricultural land. AP3 (AshDec) involves the treatment of digested sewage sludge mono-incineration ashes with a sodium sulfate to partially remove heavy metals and to increase the plant availability of P contained in the sewage sludge; a rhenanite-like P fertilizer is produced[75]. AP4 is the Ecophos process in which digested sewage sludge mono-incineration ashes are acidulated with HCl, after which the metals/metalloids are separated from the P (H_3PO_4) and co-products ($CaCl_2$, $FeCl_3$). Phosphoric acid has a similar agronomic efficiency to its solid concentrated P fertilizer counterparts (for example, SSP)[76]. AP5 is the process envisaged by the mineral fertilizer industry for the production of SSP from biogenic materials; it involves the acidulation of meat and bone meal mono-incineration ashes with sulfuric acid to produce SSP-like substances. AP6 involves the composting of the solid pig manure fraction obtained after solid–liquid separation, followed by the slow pyrolysis of the compost to produce a P-rich biochar. RP is the production of SSP through the acidulation of rock phosphate with sulfuric acid (Fig. 1b). Under the assumption that no spills occur, all pathways recover between 46% and 100% of the P from the feedstock applied (Supplementary Notes and Supplementary Table 8). In HDA, the digested sewage sludge (AP1) is co-incinerated, and the liquid manure fraction (AP6) is spread on land at a rate of 170 kg N ha^{-1}. In LDA, the residual P-rich biogenic fractions that are not turned into concentrated P fertilizers (digested sewage sludge for AP1; liquid manure fraction for AP6) are spread on land at a rate of 50 kg N ha^{-1}.

For the current-day feedstock management and use in HDA, it was assumed that digested sewage sludge and meat and bone meal are co-incinerated, whereas land application at the maximal limits allowed by EU legislation for nitrate vulnerable zones (170 kg N ha^{-1} yr^{-1}; default values of the Nitrates Directive (91/676/EEC)) was assumed for manure (Fig. 1b). It was presumed that sewage sludge was digested at the wastewater treatment plant before co-incineration[77]. Anaerobic digestion is the most common route applied to fulfil the stabilization requirements for sewage sludge spread on land laid down in the corresponding EU legislation (86/278/EEC)[78,79]. In many EU countries with a high population and livestock density, increasing volumes of biogenic materials are being incinerated and removed from the biogeochemical P cycle for the sake of environmental and human health protection and public acceptance (for example, incineration was the dominant fate for sewage sludge in the Netherlands (99%), Belgium (82%) and Germany (55%) according to data for the year 2012)[30,78,80]. It was assumed that advanced P recovery relies on manure inputs originating from larger industrial facilities to benefit from central collection and that manure processing (drying, anaerobic digestion) was already put in place in CF to address issues of odour nuisance and zoonosis and to facilitate good practices for manure storage and possible export. Specifically, it was assumed that manure slurries were anaerobically digested, whereas poultry litter was dried in a tunnel and spread on land without prior anaerobic digestion and solid–liquid separation[79]. The assumed manure application rates are in line with representative values for livestock-dense areas (for example, country-wide averages of 172 and 198 kg N ha^{-1} used agricultural lands yr^{-1} for Belgium and the Netherlands, respectively, for the year 2014)[81]. Livestock density is also closely related to nutrient surplus in the EU[30]; soils from seven out of eight EU countries with the highest livestock density of >1 livestock unit per hectare have a gross P input–output balance of >3 kg P ha^{-1} (ref. [81]). Raw and digested manure in nutrient-dense areas is often applied at high rates in excess of crop P demands[82], resulting in significant levels of P accumulation in soils[32] and elevated P concentrations in water bodies[83,84] in the main dairy-, pig- and poultry-producing regions of France, Belgium, the Netherlands, Denmark, Germany, Italy and Spain.

The current-day feedstock management and use in LDA was assumed to be landspreading at a rate of 50 kg N ha^{-1}, regardless of the type of biogenic feedstock (Fig. 1b). Landspreading is the dominant fate of manure and sewage sludge in many EU countries with a low population and livestock density (for example, >80% of stabilized sewage sludge spread on land in Slovakia, the Czech Republic, Lithuania and so on)[81], whereas meat and bone meal are used as an NP fertilizer in more extensive and organic agriculture[80,85]. For consistency and in line with legal requirements for sewage sludge stabilization, similar stabilization methods for the biogenic materials before land application were assumed as for HDA (Fig. 1b). Despite being representative situations in the EU, we are aware that the assumptions on current-day feedstock management and use are a major simplification of practices applied within the complex geographical EU landscape. The interpretation of the results is therefore only valid if advanced P recovery effectively substitutes the assumed current-day feedstock management and use in HDA and LDA.

It is noted that technology, policy and legislation develop and change over time, and that, for example, further developments in pollution prevention and control may induce changes in the inventory data and the associated outcome of this analysis. Even though considerable effort has been dedicated to the uncertainty analysis, the LCA results are based on a set of parameter values and assumptions that may be unable to represent the numerous site-specific settings across the EU (for example, heat recovery, use on land phase, feedstock management). Our entire assessment is thus based on present-day 'scenario analysis modelling', and the results are not timeless, exhaustive or specific to particular manufacturers, industry sectors or situation-specific settings. Rather, this work intends to assess the possible impacts of new advanced circular economy products in general and to provide numerical data that may help to better conceptualize and understand circular economy business models.

System boundaries and displaced activities. The life cycle stages included in the impact and cost assessment for advanced P recovery and current-day feedstock management include transport, material transformation and manufacturing stages as depicted in Fig. 1b, granulation, storage at the farm and use on land. For the RP pathway, the rock phosphate extraction (mining) and long-distance transport from Morocco was included. All other transport distances for NB1–NB6 were set to 25 km in the default scenario. Electricity generated during P fertilizer manufacturing or feedstock management was credited by substituting corresponding market electricity production, including fuel extraction. This was identified in the marginal electricity mix for the Netherlands[86], a representative country for HDA. For LDA, the same mix was used to restrict the difference between HDA and LDA to nutrient management practices. Heat recovery was not considered due to the low overall district heating share in the EU. The acidulation process applied in the AP4 pathway also generates $FeCl_3$ and $CaCl_2$ as co-products that were credited by displacing a corresponding amount of the conventional market products. Handling of residual materials and waste generated during feedstock processing as depicted in Fig. 1b was also included. For organic fertilizing materials applied on land (P content <4%), the potential benefits of use on land of nutrients and long-term carbon sequestration and the benefits of organic matter on agricultural yields were evaluated on the basis of whether the benefits are proven and quantifiable. In line with Martínez-Blanco et al.[87], it was concluded that nutrient supply and carbon sequestration are proven and quantifiable, whereas the benefits with regards to organic matter on yields were indicated not to be proven according to meta-analyses studies for non-carbonized[88] and carbonized[89] organic materials in a relevant European context. The uptake/release of biogenic CO_2 associated with the feedstock was assigned a characterization factor equal to zero. The sequestered biogenic CO_2 in soils within the 100 yr time horizon considered was assigned a factor equal to −1 for climate change, following common practice. It was considered that 11% of the input carbon content was sequestered (not emitted) over the 100 yr time horizon considered for organic fertilizing materials returned to soil[90], whereas for biochar a value of 90% was assumed[91]. The full inventory data for the use-on-land phase are presented in Supplementary Notes. The 'effective' NPK supply to plants in organic fertilizing materials (P content <4%, for example, liquid pig manure fraction, residual digested sewage sludge) and the 'effective' NK supply in concentrated P fertilizers were calculated and assumed to displace mineral fertilizer production and use. The average EU mix for N fertilizers (urea 24.5%, ammonium nitrate 27%, calcium ammonium nitrate 33% and urea-ammonium nitrate 15.5%[26]), SSP for P fertilizers and potassium chloride for K fertilizers[25,27] was therefore assumed. Some fertilizing materials may be characterized by imbalanced stoichiometric ratios, especially N/P ratios. As a result, the application of high amounts of fertilizing materials can result in the application of nutrients in excess of plant needs[92]. In such cases, bioavailable nutrients present in organic fertilizing materials do not substitute their mineral fertilizer counterparts in a 1:1 ratio. This work relies on the 'maintenance substitution principle[92]', where potential plant nutrient uptake from organic fertilizing materials is compared with general crop nutrient requirements for P and K, and any excess nutrients are assumed not to substitute mineral fertilizers (Supplementary Notes). Plant nutrient requirements were valued at 17.5 kg P ha^{-1} and 91 kg K ha^{-1} on the basis of FAO data for crop P and K requirements[29] and offtake values[93,94] for productive cropland ecosystems. For HDA, soil P saturation was assumed, and bioavailable P applied in amounts in excess of plant P demands was assumed to be lost through surface run-off and groundwater leaching in the default scenario[95]. P-depleted soils were assumed in LDA, and bioavailable P in excess of plant demands was assumed to be stored in the soil matrix. The system boundaries and crediting of displaced activities were consistently applied to both advanced P recovery and current-day feedstock management scenarios. An example is provided for production option no. 3 in Supplementary Methods and Supplementary Fig. 7.

Inventory data. The technology data used to model the foreground LCA systems were mostly derived from plant operators. When necessary, eventual data gaps were filled with appropriate literature sources[18,96,97]. Detailed input–output tables and parameter values for all the technologies and processes included in the analysis are presented as outlined in Supplementary Notes (Supplementary Tables 8–10 and 14–19, respectively). Storage of fertilizers and organic fertilizing materials before use on land was modelled on the basis of literature data[98–103], taking into account the best available storage techniques[100]. Background life cycle inventory data to model energy and chemicals used as input to the technologies and processes modelled in the foreground system were mostly taken from the Ecoinvent v3.5 database[86] or from the European reference Life Cycle Database (ELCD)[104], while the associated market unit cost was derived from available sources (Supplementary Notes and Supplementary Table 20). The inventory on capital goods required for trucks, incinerators (also used as a proxy for pyrolysis), anaerobic digestion, composting and landfill was based on Brogaard and Christensen[105]. The initial investments in terms of capital and expenditure costs (CAPEX) were mostly based on primary data from operators (see Supplementary Methods for calculation details), except for co-incineration and composting that were based on the literature[64,106]. All inventory parameter values applied for LCA modelling are listed in Supplementary Notes and Supplementary Tables 2 and 14–20.

Sensitivity analyses. Sensitivity analysis was addressed at two levels: (1) testing key scenario assumptions (variants) one at a time and (2) propagating parameter uncertainty. We tested the following key scenario variants: (1) natural gas instead of the marginal electricity mix for the Netherlands chosen as the default electricity source; (2) manure dewatering to at least 25% dry matter content (pig manure) followed by export to nutrient-deficient areas instead of local application in HDA; (3) plant P demand equal to 6.5 kg P ha^{-1} instead of the default value of 17.5 kg P ha^{-1}. Parameter uncertainty was addressed using the propagation method described in Bisinella et al.[107] (Supplementary Methods).

Data availability
The data supporting the findings of this study are available within the paper and its supplementary information files.

Received: 11 April 2019; Accepted: 24 September 2019;
Published online: 4 November 2019

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Acknowledgements

We are grateful to the manufacturers that provided primary data for the life cycle inventory. We thank P. Eder and E. Garbarino for guidance and revising previous drafts of this manuscript and A. Atkinson for language editing. The views expressed are purely those of the authors and may not in any circumstances be regarded as stating an official position of the European Commission.

Author contributions

D.T. performed the environmental life cycle and cost analyses; H.G.M.S. and D.H. conceived the research and supervised the collection of primary data from manufacturers; D.T. and D.H. conceptualized the life cycle approach applied; D.H. wrote the paper with substantial contributions from D.T. and H.G.M.S. All authors interpreted the results, elaborated the structure for data presentation and developed the research conclusions.

Competing interests
The authors declare no competing interests.

Additional information
Supplementary information is available for this paper at https://doi.org/10.1038/s41893-019-0416-x.

Correspondence and requests for materials should be addressed to D.H.