# Sustainability Assessment of Activated Carbon from Residual Biomass used for Micropollutant Removal at a Full-Scale Wastewater Treatment Plant

**Ben Joseph[1], Korbinian Kaetzl[1,2,4], Frank Hensgen[1], Bernhard Schäfer[3] and Michael Wachendorf[1]**

[1] Grassland Science and Renewable Plant Resources, Universität Kassel, Witzenhausen, Germany
[2] Competence Centre for Climate Change Mitigation and Adaptation (CliMA), Universität Kassel, Kassel, Germany
[3] Eigenbetrieb Umwelttechnik Baden-Baden, Germany

[4] Author to whom any correspondence should be addressed.
E-mail: kaetzl@uni-kassel.de

Received xxxxxx
Accepted for publication xxxxxx
Published xxxxxx

**Abstract**

Activated carbon (AC) used for removal of organic micropollutants in European wastewater treatment plants (WWTPs) is usually produced from non-renewable resources that need to be transported over long distances. Utilising local residual biomass as a raw material may be advantageous in terms of sustainability. This study investigated the environmental and energy balances of using biowaste and biomass from landscape management for micropollutant removal at a commercial scale WWTP. Both residual biomasses were processed using the integrated generation of solid fuel and biogas from biomass (IFBB) technique to obtain a press cake that was used as feedstock for AC production. The results showed a lower global warming potential (GWP) and cumulative energy demand in comparison to a fossil-based conventional AC. Differences in GWP between residual and fossil ACs were enhanced when the end-of-life incineration step was considered, and residual AC had a lower social risk associated with its production. Energy efficiency of AC production was substantially increased by utilising waste heat generated in the pyrolysis process of biochar and by using electricity generated in a combined heat and power plant using biogas from the methanation of IFBB press fluids. Converting residual biomass into activated carbon using IFBB and a state-of-the-art pyrolysis and activation unit along with energy recovery would aid WWTPs to become self-sufficient in terms of raw materials and improve the sustainability of WWTPs.

Keywords: Life Cycle Assessment (LCA), Biochar, Social Risk Assessment, Global Warming Potential, Sustainable Resource Management

## 1. Introduction

Due to its adsorbent properties activated carbon (AC) can efficiently remove organic micropollutants (OMPs) from wastewater (Çeçen and Aktaş, 2012; Wang and Wang, 2016). OMPs are found in sewage water treatment systems and one of the main sources are pharmaceuticals originating from households, livestock farming and hospital effluents (Mansour et al., 2018). The detection of OMPs in surface waters indicates that the present treatment at waste water treatment plants (WWTPs) is insufficient to remove OMPs effectively (UBA, 2015), hence legal regulations are already in force in Switzerland for the implementation of a fourth treatment stage at WWTPs (FOEN, 2015). Similarly, European Union water framework directive (2000/60/EC) and the German federal water act (Wasserhaushaltsgesetz, WHG) put forward legal obligations to maintain the quality of water (UBA, 2018).

The usage of powdered activated carbon (PAC) at WWTPs as an additional treatment step is an option to effectively remove micropollutants (Boehler et al., 2012). ACs used in European WWTPs are either manufactured from bituminous coal (Bayer et al., 2005), which is related to substantial environmental and social impacts assosiated with the material extraction, or from cocunut shells, which frequently originate from monocultures with adverse environmental impacts due to land use change and the use of chemical fertilizers and pestisides (Hartemink, 2005). A large share of these materials are produced in Asian countries and need to be transported over long distances.

Using local residual raw materials (e.g. green cut, wood chips, fruit stones) instead, which are plentily available across Europe (Scarlat et al., 2019), may overcome the constraints mentioned above and provide a pathway to convert these residues into a valuable resource. The public enterprise Eigenbetrieb Umwelttechnik in Baden-Baden, Germany, operates a municipal WWTP and biomass recycling center on the same site. Among other things, landscape management residues, hereinafter referred to as Baden-Baden Mixture (BM), and household biowaste (BW) consisting mainly of kitchen and garden waste are regularly delivered there (Table S1.1), which up to now have mainly been used for biogas and compost production. BM and BW were pre-treated using the integrated generation of solid fuel and biogas from biomass (IFBB) technique, which mechanically separates biomasses into a solid and a liquid fraction (Hensgen et al., 2011; Richter et al., 2010; Wachendorf et al., 2009). The solid fraction is released from ash and harmful elements (e.g. N, S, Cl and K) and the liquid fraction produced can be co-digested to produce biogas that can be used as an energy source for the IFBB process. For carbonisation and steam activation a Pyreg A500 plant (Pyreg GmbH, Dörth, Germany) was put into operation which can handle ca. 900 t dry matter (DM) per year.

The aims of this study were to assess the enviromental impact and energy demand resulting (i) from the production of AC based on biomass mixture (BMC) and biowaste (BWC) and (ii) from the usage of these AC as an additional treatment step for micropollutant removal at a WWTP in comparison to conventional activated carbon (CC). A social risk assessment was conducted (iii) aiming at a deeper insight into the social effects during production of ACs. The production of AC from residual materials is well studied, however, most studies on the related environmental impacts were based on laboratory scale data. Therefore, using life cycle assessment (LCA) of AC production and use at a commercial scale WWTP would increase the understanding of environmental and social impacts of this technical approach.

## 2. Methods

### *2.1 Life cycle inventory (LCI), system boundaries and impact categories*

Samples of raw and IFBB pre-treated BM and BW materials used in this study were obtained from the WWTP Baden-Baden. Press cakes from IFBB process were converted into ACs using a Pyreka (Pyreg GmbH, Dörth, Germany) laboratory scale pyrolysis and activation reactor. The NORIT SAE Super (Boston, Massachusetts, USA) was used as a reference CC for benchmarking BMC and BWC. For the LCI, material flows and process data were calculated using data obtained from the WWTP. The datasets used for modelling the processes are elaborated in section S2, and compiling was done using OpenLCA 1.9.0.

A "gate to grave" approach was used in this study considering all processes involved from handling of the incoming raw material at the facility until the use of ACs produced. Incineration of ACs with heat recovery was considered as an end of life option, whereas disposal of ash generated during the incineration was not considered. The maximum annual production capacity of 900 t (dry weight) of BM and BW input material of the pyrolysis and activation unit was used for calculation of mass and energy balances (Figure 1).

### *2.2 Activated carbon production and usage*

Raw BM and BW was pre-treated according to the IFBB technology approach, as described in details by (Joseph et al., 2018). Ensiled BM and fresh BW were mashed in warm water (40 °C) for 15 min. Subsequently, solid (press cake) and liquid fraction was mechanical separated by a screw press. While the liquid fraction was co-digested along with sewage sludge to generate biogas, the press cake was used for AC production (Figure 1). Through the IFBB-process, mineral and ash content of BM and BW could be significantly reduced, which in turn improved the feedstock quality for AC production (Table 1). Energy and material balances for all the processes involved in BM and BW production at the WWTP were calculated (Table S1.2-1.5).

For AC production, BM and BW were pyrolyzed at 900 °C and physically activated by adding water vapour as oxidation agent during pyrolysis, using the Pyreka laboratory-scale reactor. This ensured a very good transferabillity of results to the full-scale Pyreg A500 reactor, which is used at the WWTP Baden-Baden (details in the Supplementary material).

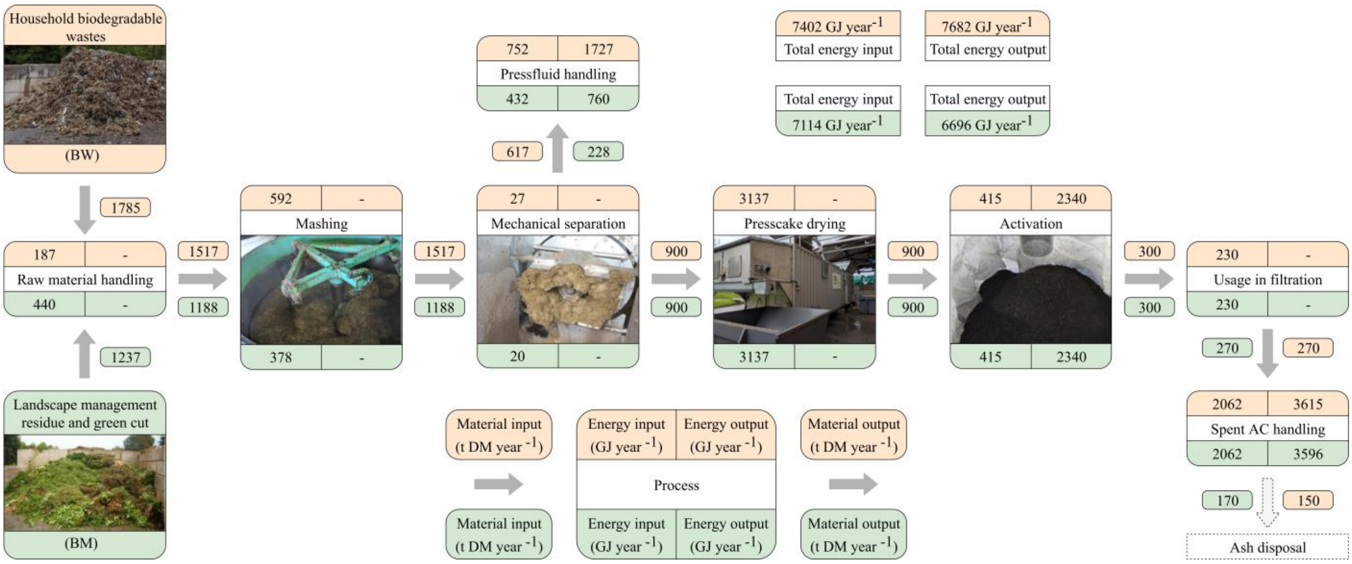


***Figure 1.*** System boundary diagram elaborating the BM and BW AC production at the WWTP. The amount of input and output material and energy required for the production of the annual capacity AC is indicated along with the diagram. Ash disposal was not considered in this study.

At the full-scale PYREG A500 reactor, the thermal energy required for the pyrolysis process was generated by burning the resulting pyrolysis gases. In addition, surplus heat energy from pyrolysis was recovered and used to supply the processes in the pre-treatment steps (Figure S1.3). After its use for the removal of OMPs from wastewater, incineration of used ACs, further referred as spent ACs, was assumed to be the typical end-of-life stage. The resulting heat was assumed to offset local heat requirements and considered in the LCI.

***Table 1.*** Characteristics of Baden-Baden mix (BM) and Biowaste (BW) based materials at different steps in the production process of activated carbon.

| | Unit | BM | BW |
|---|---|---|---|
| **Raw material** | | | |
| DM content | % FM | 36.3 | 34.7 |
| Ash | % DM | 10.8 | 24.2 |
| Volatile solids | % DM | 89.3 | 75.8 |
| C concentration | % DM | 46.3 | 36.5 |
| DM flow into press cake | % | 75.8 | 59.3 |
| DM flow into press fluid | % | 24.2 | 40.7 |
| **Press cake** | | | |
| DM content | % FM | 55.3 | 52.7 |
| Ash | % DM | 6.5 | 10.8 |
| Volatile solids | % DM | 93.6 | 89.2 |
| C concentration | % DM | 47.9 | 43.1 |
| Lower heating value | MJ $DM^{-1}$ | 18.0 | 16.1 |
| DM content after drying | %FM | 85.2 | 85.2 |
| **Activated carbon** | | | |
| Ash | % DM | 70.7 | 62.5 |
| Volatile solids | % DM | 29.3 | 37.5 |
| C concentration | % DM | 27.9 | 37.9 |
| Lower heating value | MJ $DM^{-1}$ | 16.7 | 16.7 |
| **Press fluid** | | | |
| DM content | % FM | 1.8 | 2.9 |
| Ash | % DM | 23.2 | 54.6 |
| Volatile solids | % DM | 76.8 | 45.4 |
| C concentration | % DM | 41.5 | 26.0 |
| Methane yield | $L_n$ $CH_4$ $kg^{-1}$ VS | 275.0 | 395.0 |

*2.3 Functional unit*

OMPs pose an ecotoxicological risk in aquatic environments (Oldenkamp et al., 2019). This risk was calculated according to the USEtox model (Rosenbaum et al., 2008) for selected pollutants, which were measured in our study (Table S2.1). By using the investigated ACs for wastewater treatment, OMPs were removed and, thus, the ecotoxicological risk in receiving water bodies was reduced. The factors defined in the USEtox model for OMPs were multiplied with their concentration in wastewater to calculate the mean ecotoxicity value for all OMPs. Based on these values, the functional unit for the LCA was fixed to the required quantity of AC to reduce the aquatic freshwater exotoxicity potential to a defined level in terms of comparative toxic unit, ecotoxicity ($CTU_e$ $L^{-1}$) per 1000 $m^3$ of wastewater (Figure S2.2) (Rosenbaum et al., 2008).

Performance of AC in terms of OMP removal from wastewater was measured in laboratory experiments. For this, concentrations between 1.75 to 7 mg $L^{-1}$ of AC were added to the wastewater to evaluate the influence of the dosage on the reduction rate of OMPs. The ecotoxicity reduction at a dosage of 5 mg $L^{-1}$ of CC was used as a benchmark. The required amount of BMC and BC to achieve the same ecotoxicity reduction was calculated used for the LCA (Figure S2.2). Based on these data, multiplication factors for the LCA were determined by dividing the amounts of BM and BW AC through the 5 mg $L^{-1}$ of CC (Figure S2.1).

*2.4 Social risk identification*

The social risks involved in the production chain were identified based on the country specific risk assessments adapted from (Saling et al., 2020) considering all main foreground processes. The processes were classified into risk levels according to the intensity of social risk in the country of production, using 11 social impact categories from Verisk Maplecroft™ (Table S4.1). If the country of production was known, the country's social risks for 11 chosen social impact categories were obtained from Verisk Maplecroft™. These categories were weighted according to the severity of the social issues as estimated by INEF (2015) (Table S4.1). Depending on the data availability, social risk scores on country level or mean values for the 10 largest producer countries, weighted by production volume were used for the LCA.

**3. Results**

3.1 *Ecotoxicity*

Removal efficiency of OMPs from wastewater was higher for CC compared to BMC and BWC (Table S2.2 and S2.3). A dosage of 5 mg $L^{-1}$, CC was able to attain a calculated ecotoxicity of approximately 55.000 $CTU_e$ $L^{-1}$. To achieve the same ecotoxicity level, a dosage of 9.1 mg $L^{-1}$ of BWC was needed, which resulted in a 1.82-fold demand of BWC compared to CC. In contrast, performance of BMC was significantly better (6.1 mg $L^{-1}$) resulting in an equivalent of 1.22 times CC (Figure S2.2). Based on an annual amount of wastewater of around 9,510,000 $m^3$ to be treated and a dosage of 5 mg $L^{-1}$ of CC, this resulted in a demand of approximately 48 t of CC. Considering the factors of 1.22 and 1.82 for BMC and BWC respectively, 87 t and 58 t per year was needed to achieve the same ecotoxicity level.

*3.2 Global warming potential*

The balance of GWP for BMC and BWC production (5.3 and 4.5 kg $CO_2$ eq. $FU^{-1}$ respectively) was significantly lower compared to CC (20 kg $CO_2$ eq. $FU^{-1}$) (Figure 2). Considering an incineration of the spent ACs and the recovered energy obtained from it, GWP for BMC and BWC was reduced to 2.7 and 1.3 kg $CO_2$ eq. $FU^{-1}$ respectively, whereas this resulted in an increased GWP of 29 kg $CO_2$ eq. $FU^{-1}$ for CC. Heat usage for press cake drying had the highest GWP for BMC and BWC (7.1 and 4.5 kg $CO_2$ eq. $FU^{-1}$ respectively). For CC, heat usage for pyrolysis and activation was the highest contributor (14 kg $CO_2$ eq. $FU^{-1}$). For the production of the required 58 t of BMC and 87 t of BWC, 345 and 515 t DM of raw material needed to be processed, respectively. Hence, the GWP associated with electricity usage in the facility was higher for BWC as for BMC. Similarly, GWP related to material handling of press fluid digestates depended on the quantity that was transported and applied on agricultural land, leading to higher GWP for BWC, as 6,232 t FM was handled compared to only 4,689 t FM for BMC. Mining and transportation of hard coal for further processing led to substantial material handling related GWP for CC. However, GWP related to infrastructure usage was negligible when compared to the overall GWP for all 3 ACs. The total usage related GWP was highest for CC, closely followed by BWC and much lower for BMC.

In contrast to CC, whose production only generated GWP emissions, the production of BMC and BWC also enabled GWP savings. Heat recovered from the activation unit saved 3.4 kg $CO_2$ eq. $FU^{-1}$ for BMC compared to 5.1 kg $CO_2$ eq. $FU^{-1}$ for BWC. Electricity recovered from combustion of press fluid-derived methane was higher for BWC and saved 5.7 kg $CO_2$ eq. $FU^{-1}$ compared to 1.7 kg $CO_2$ eq. $FU^{-1}$ for BMC. Application of digestate on agricultural fields replaced the need for conventional mineral fertilisers, thereby saving 3.3 (for BMC) and 1.9 (for BWC) kg $CO_2$ eq. $FU^{-1}$, respectively. Energy recovered from incineration of spent AC resulted in GWP savings of 5.8 and 7.1 kg $CO_2$ eq. $FU^{-1}$ for BMC and BWC respectively. But incineration also caused additional emissions of 3.2 (BMC) and 3.9 (BWC) kg $CO_2$ eq. $FU^{-1}$ from heat usage for de-watering and drying of spent AC sludge. For CC there was a saving of 4 kg $CO_2$ eq. $FU^{-1}$ including incineration, however, the benefits of incineration were overshadowed by additional emissions of 12 kg $CO_2$ eq. $FU^{-1}$ resulting from the release of $CO_2$ during incineration and 2 kg $CO_2$ eq. $FU^{-1}$ for de-watering and drying. Overall, GWP savings were highest for BWC, followed by BMC and CC.

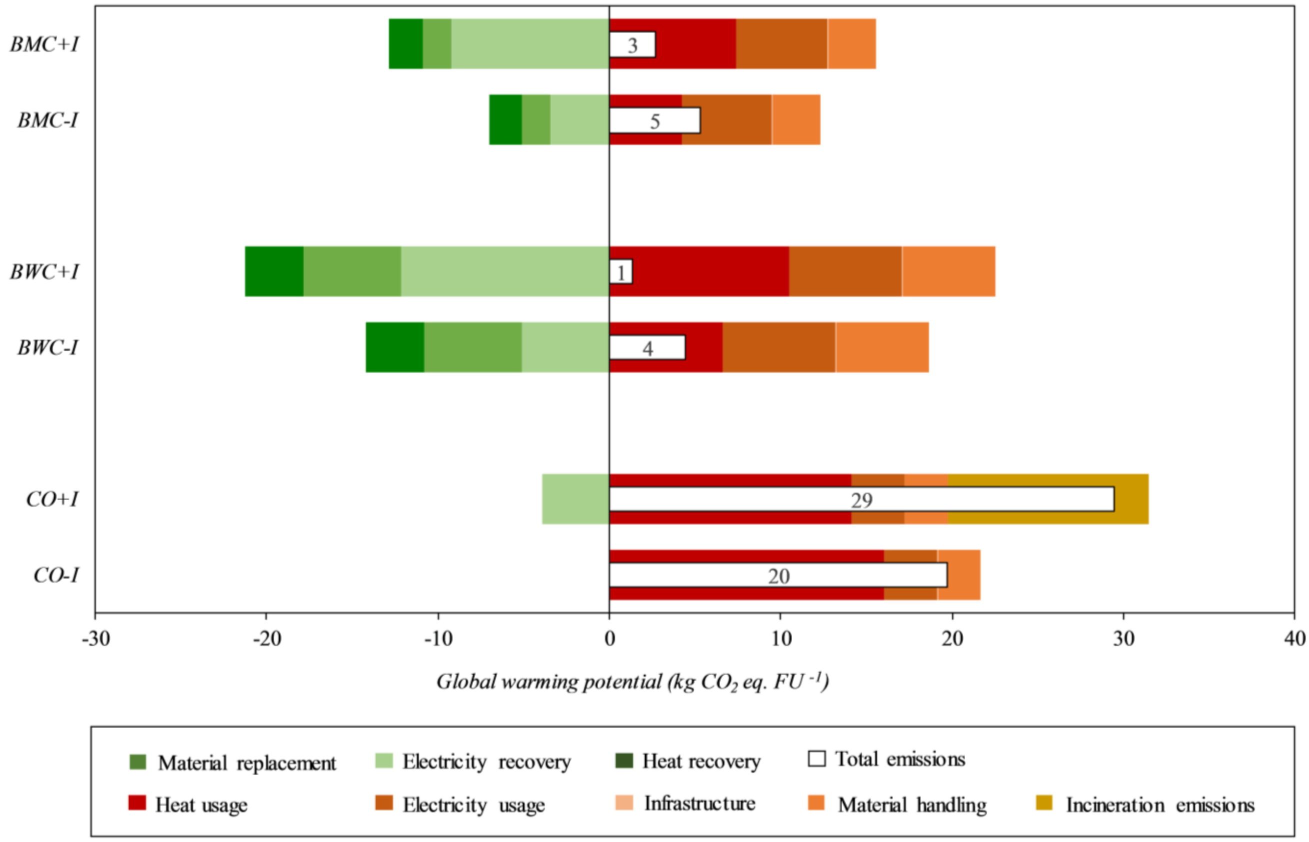


***Figure 2.*** Global warming potential (GWP) generated during the production and usage of activated carbon derived from Baden-Baden mix (BM), Biowaste (BW) and Conventional coal (CC). Each section of the horizontal bars indicates the amount of GWP emitted or saved by the respective parameters as indicated in the legend. (+I) denotes the inclusion of the end of life incineration stage and related effects on the GWP were visualised accordingly, whereas (-I) indicates the case without considering incineration. The negative values indicate GWP savings and the positive values indicate GWP emissions, therefore a lower value would mean the option has a lower GWP.

### *3.3 Cumulative energy demand (CED)*

Energy needed for operating the processes was estimated in terms of CED and classified into renewable and non-renewable CED according to the source of energy. Energy used to produce CC was almost completely based on non-renewable sources, whereas biomass was the primary source of energy for producing BMC and BWC (5% and 11% share of non-renewable CED for BMC and BWC, respectively). Total use of non-renewable energy was substantially higher for CC (269 MJ $FU^{-1}$) than for BMC and BWC (54 and 83 MJ $FU^{-1}$ respectively) (Figure 3).

Due to energy recovery and material replacement, the energy balance for BMC and BWC further improved to 24 and 34 MJ $FU^{-1}$ respectively, and the highest non-renewable CED resulted from fuel usage for transportation of digestate (almost 60% of the total non-renewable CED both for BMC and BWC). While incineration as an end of life option improved the energy balance of all three ACs, the combustion of the loaded BMC and BWC even generated excess energy. Contrarily, the balance for renewable CED was higher for both BMC and BWC (291 and 222 MJ $FU^{-1}$ respectively) compared to 150 MJ $FU^{-1}$ for CC (Table S3.3). Heat usage for press cake drying represented almost 75 % of the renewable CED from both BMC and BWC and it increased to 81 % when the heat required for drying AC sludge was considered. Though overall CED was higher for BMW and BWC, almost 90 % of it was based on renewable biomass feedstock.

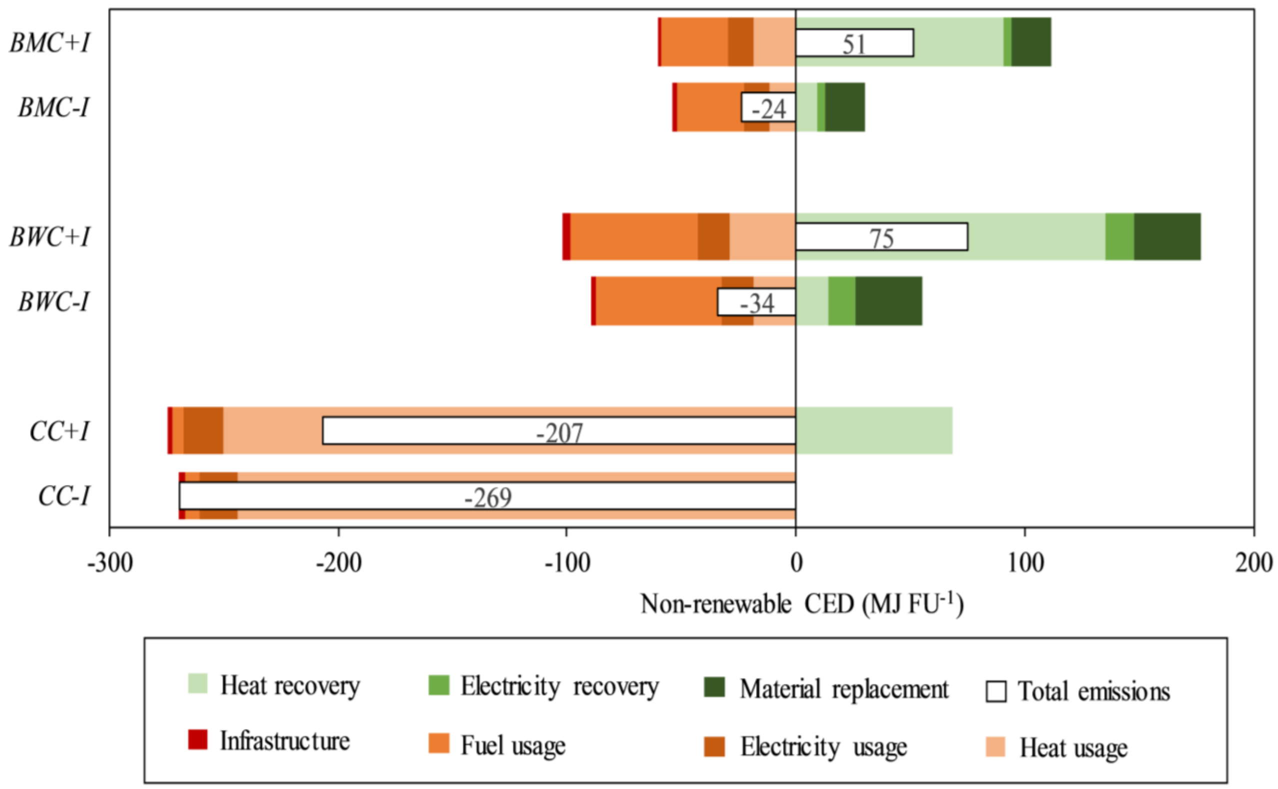


***Figure 3.*** Cumulative energy demand (CED) of non-renewable sources generated during the production and usage of activated carbon derived from Baden-Baden mix (BM), Biowaste (BW) and Conventional coal (CC). Each section of the horizontal bars indicates the amount of non- renewable CED used or saved by the respective parameters as indicated in the legend. (+I) denotes the inclusion of the end of life incineration stage and related effects on the GWP were visualised accordingly, whereas (-I) indicates the case without considering incineration. The negative values indicate the usage of energy and the positive values indicate the savings therefore a higher positive value would mean the option saves more energy than it uses.

### *3.4 Additional environmental impacts*

The acidification potential associated with the production and usage of bio-based AC was significantly higher than for CC (Table S3.1). This was mainly caused by the $SO_x$ and $NO_x$ emissions from the activation process, which accounted for more than 60% of the total acidification potential. While $NH_3$ emissions from the press fluid digestate application on the field had a substantial contribution of 26 and 33 % for BMC and BWC, respectively. Almost 42 % of the acidification potential of CC was related to electricity and fuel demands during coal mining. Further emissions associated with heat and electricity usage during drying, pyrolysis and activation amounted to 53 % for CC.

BWC had the highest eutrophication potential followed by BMC and CC. $NH_3$ emissions from digestate application accounted for the largest share of total emissions both for BMC and BWC (38 % and 50 % respectively). For CC, nitrogen (N)-based emissions for heat and electricity generation were the highest contributors. Including the incineration of loaded ACs reduced the acidification and eutrophication potential of all ACs (Table S3.1 and S3.2).

Environmental and energy balances were also examined at a dosage of 20 mg AC $L^{-1}$, which is the highest standard dosage used at WWTPs, and it was found that this increased GWP balances of BMC, BWC and CC by a factor of 3.2, 3.0 and 3.55, respectively. Considering incineration resulted in a lower GWP for BMC and BWC (6 and -4 kg $CO_2$ $FU^{-1}$) but CC had a higher GWP (107 kg $CO_2$ $FU^{-1}$) (Figure S3.1).

### *3.5 Social life cycle assessment*

Producing bio-based ACs at the WWTP was found to reduce social risks substantially compared to the conventionally produced AC, as 83 % of all processes at Baden-Baden were categorised as “low risk” in contrast to only 14 % for the production of CC (Figure S4.2). This was mainly due to the fact that processes involved in the extraction of bituminous coal take place in countries which usually perform poorly for almost all social impact categories (Table S4.2). While processing, production and usage of CC took place at locations with somewhat lower social risks, the associated use of diesel and natural gas resulted in very high to high social risks, respectively.

## 4. Discussion

*4.1 Ecotoxicity*

Performance of BMC and BWC in terms of OMP removal from wastewater was significantly lower compared to CC (Table S2.2 and S2.3). This may be explained by the significantly lower specific surface area and adsorption capacity of bio-based ACs. Hence, higher dosages of BMC and BWC were needed to achieve the same level of ecotoxicity reduction. Removal of OMPs from effluent water mainly depends on the adsorption characteristics of ACs used, such as specific surface area (Mailler et al., 2016). These characteristics are highly affected by the properties of raw materials (Erdoğan et al., 2017) and the activation process (Bergna et al., 2019). In particular the influence of the raw materials on adsorption of OMP were not clear, although the process conditions for BMC and BWC production were comparable (Mailler et al., 2016). However, it is remarkable that the removal of OMP through BMC was higher than for BWC though the specific surface area for BMC was lower. The high ash and low carbon content for BMC and BWC may have affected the adsorption capacity, as these properties have an influence on the pore structure (Anisuzzaman et al., 2015). The ash content of the raw biomass was significantly reduced by the IFBB pre-treatment, which improved the quality of the ACs (Hensgen and Wachendorf, 2018; László et al., 1997). Further improvements in biomass processing, pyrolysis and activation can be expected to increase the adsorption capacity of activated carbons. This, in turn, would result in a further reduction of GWP, CED and other environmental impacts through BMC and BWC production and usage.

*4.2 Global warming potential*

The combination of raw biomass collection, processing into ACs and their subsequent utilization at the investigated WWTP is unique. To facilitate the comparison with other studies, all life cycle stages including production, usage and end of life were examined individually and compared with similar studies.

Emissions during the production stage exhibited maximum contributions to GWP in the life cycle of all ACs. To allow a comparison with other studies at this stage, the GWP generated by producing 1 kg of the respective AC was calculated and used instead of the functional unit. The production of 1 kg BMC and BWC resulted in emissions of 0.43 and 0.20 kg $CO_2$ eq. respectively, which was significantly lower than those reported for AC from olive waste cake (11.1 kg $CO_2$ eq. $kg^{-1}$) (Hjaila et al., 2013). GWP for ACs produced from conventional raw materials was 1.15 kg $CO_2$ eq. $kg^{-1}$ for coconut shell (Kim et al., 2019), 8.4 to 11.1 kg $CO_2$ eq. $kg^{-1}$ for bituminous coal (Bayer et al., 2005; Gabarrell et al., 2012) and 2.45 kg $CO_2$ eq. $kg^{-1}$ for regenerated AC (ROFA CARBON, 2018), which is notably higher compared to BMC and BWC. In our study, GWP for CC production from bituminous coal (3.41 kg $CO_2$ eq. $kg^{-1}$) was significantly lower than the range reported by (Bayer et al., 2005) and (Gabarrell et al., 2012), which can be explained by the fact, that $CO_2$ emissions associated with removal of fossil carbon was not considered in the production stage for this study, but rather included in the end of life stage as $CO_2$ emitted during incineration.

The low GWP in our study can be attributed to the efficient pyrolysis and activation process, where the combustion of pyrolysis gas provided more heat than needed. While pyrolysis was reported to account for 30-47 % of the total GWP (Gu et al., 2018; Hjaila et al., 2013), it was only 7% in the present study. Further, electricity generation from the IFBB press fluid reduced the energy demand for production and, consequently, the GWP.

The delivery of residual raw materials to the facility prevented $CO_2$ emissions related to transportation and extraction. Conversion factors from raw material to AC were slightly different for BM (11 %) and BW (7 %) which was due to different physico-chemical properties, such as dry matter and carbon content, as well as differing mass flows during mechanical separation. With approximately 24 % the conversion efficiency from press cake to AC was equal for BM and BW. However, the transfer of BM dry matter into the press cake (76 %) was considerably higher than for BW (59 %), which resulted in a lower contribution of AC production to the entire GWP for BM, as less raw material had to be processed. On the other side, the higher mass flow of BW dry matter into the press fluid increased GWP savings from electricity recovery. Though the type of raw material affected the overall GWP, the impact of the production process was much greater. Since the complete production and usage of BMC and BWC took place at the WWTP, material transport did not contribute to the GWP. However, emissions related to the transport of the reference CC were also neglectable (0.003 kg $CO_2$ eq. $FU^{-1}$). Nevertheless, the GWP associated with the production did not consider the contrasting performances of ACs in terms of OMP removal. To this end an appropriate functional unit (i.e. the quantity of AC to achieve a defined ecotoxicity level in wastewater) was introduced which allowed determining the amounts of ACs required and, thus, a true comparison between conventional and bio-based ACs.

Energy recovery through incineration of the spent BMC and BWC reduced the overall GWP for these ACs, whereas the combustion of CC resulted in an increased GWP, as the resulting $CO_2$ emissions originated from fossil sources. Hence, using AC produced from local residual biomasses has an advantage during incineration which is a commonly used end of life step for ACs. OMP removal using ozonation treatment usually has a lower GWP (18 kg $CO_2$ eq. 1000 $m^{-3}$) (Kounina and Wencki,

2015) compared to CC treatment (20 kg $CO_2$ eq. 1000 $m^{-3}$) (Baresel et al., 2019), however, the figures for BMC and BWC of 4 and 5 kg $CO_2$ eq. 1000 $m^{-3}$ respectively in our study were much lower in comparison to ozonation treatment.

### *4.3 Cumulative energy demand (CED)*

The energy balance of AC production and use is decisive for the profitability of the concept (Blumenstein et al., 2012). The CED for the production of BMC and BWC (30 and 15 MJ $kg^{-1}$respectively) was notably lower than for CC (53 MJ $kg^{-1}$). Our data differ widely from results of other studies, which found CED values between 158 and 241 MJ $kg^{-1}$ for woodchip and coal-based AC respectively (Bayer et al., 2005; Gu et al., 2018). The lower CED for BMC and BWC production in our study was mainly caused by the efficient pyrolysis and activation processes as well as by recovered electricity from anaerobic digestion of press liquids.

Methane yield from the press fluid was considerably higher for BWC (395 $L_n$ $CH_4$ $kg^{-1}$ VS) compared to BMC (264 $L_n$ $CH_4$ $kg^{-1}$ VS). However, methane content and mass flow of volatile solids into the press fluid were highly influenced by material properties, such as dry matter content and IFBB process parameters (Hensgen et al., 2011; Richter et al., 2009). Even if the overall CED was lower for production of BMC and BWC, the share of non-renewable CED was higher than for CC. As these energy sources are based on depletable resources (Arvidsson and Svanström, 2016), additional research is needed to improve the sustainability of alternative AC production. Among others, this may include the substitution of diesel for transportation and processing through biofuels.

### *4.4 Additional environmental impacts*

Emissions of $NO_x$, $SO_x$ and $NH_3$ damages land ecosystems by affecting plant health and change in soil pH (Rosenbaum et al., 2018). Though the emission of $NO_x$ and $SO_x$ from the activation process were the largest contributors towards acidification, they were still below the German limit of 350 mg $m^{-3}$ (Bundesministerium für Umwelt, Naturschutz und Reaktorsicherheit, 2002). Mass flow of N during mechanical separation into press cake and fluid caused N emissions in the following stages, i.e. $NO_x$ emissions during pyrolysis of the press cake and activation of the resulting biochar (Zhan et al., 2018) as well as $NO_x$ and $NH_3$ emissions from the field application of digestates originating from the anaerobic digestion of press fluids (Eggleston, 2006). Recent research showed that the flow of elements during mechanical separation was influenced by the raw material properties, process parameters (Richter et al., 2011) and scale of the conversion process (Joseph et al., 2018). The modification of these factors would affect the flow of elements and, thus, reduce the acidification and eutrophication potential. Further reduction is possible through the use of precision application techniques for digestates (Nicholson et al., 2018) and advanced flue gas cleaning systems (Turconi et al., 2013).

### *4.5 Social risk assessment*

For estimating the social risks associated with AC production, available average values on country level were used, as data from producers were not existent or not representative (Saling et al., 2020). Our results indicate that risks could be reduced when all processes take place in countries with a low risk level or when materials from countries with a lower associated social risk were used. To this end, materials need to be labelled appropriately.

The local production and use of AC from residual biomass in industrialized countries like Germany, would affect economic activities in the current production countries. Possible losses of employment could, however, be offset by investments in alternative employment opportunities for people working in fossil-based sectors (Sparkes, 2008). Due to a lack of revenue, this may lead to a further deterioration of the social situation in the developing countries. Regardless of this, socially responsible investments would help to improve the social conditions and certification of socially compatible production processes may contribute to a higher demand of products with lower social impact (Goedkoop et al., 2018). Nevertheless, the conducted social risk assessment for AC production contained global and, therefore, inaccurate data. Hence, only social hotspots in the production process on country level could be identified (Saling et al., 2020). However, our findings may serve as a starting point for further investigations on the exact sources of social risk at a sector and company level.

## **5. Conclusion**

Using ACs produced from local residual biomasses for the removal of OMPs at a large-scale WWTP was found to have lower environmental impacts in terms of GWP and CED when compared with conventional ACs. The novel concept was found to be flexible in terms of raw material, though both BM and BW used in this study had distinctive properties. The IFBB step significantly reduced the ash content of feedstocks for AC production and the generated press fluids further improved energy generation and material replacement. Thus, IFBB was a crucial step in utilising residual biomass for AC production. On-site production and use of the AC enabled a significant reduction of monetary and environmental costs compared to systems based on conventional ACs. The lower OMP adsorption performance of BMC and BWC could be overcome by using higher dosages of AC without hazarding GWP and CED advantages. Performance of BMC and BWC is expected to be increased by further process optimization, which would render the novel system attractive for WWTPs that currently use conventionally produced AC.

**Acknowledgement**

The authors would like to acknowledge the European Union for financing the Re-Direct project (NWE 294) through the INTERREG European regional development fund. We would also like to thank O. Hermann from Eigenbetrieb Umwelttechnik Baden-Baden for providing the material and information needed for the study. We are grateful to the support of K. Stenchly, T. Astor, J. Wijesingha, T. Fricke, R. Graß and A. Gerke.

# References

Anisuzzaman, S.M., Joseph, C.G., Daud, W.M.A.B.W., Krishnaiah, D., Yee, H.S., 2015. Preparation and characterization of activated carbon from Typha orientalis leaves. Int J Ind Chem 6, 9–21.

Arvidsson, R., Svanström, M., 2016. A framework for energy use indicators and their reporting in life cycle assessment. Integrated environmental assessment and management 12, 429–436.

Baresel, C., Ek, M., Ejhed, H., Allard, A.-S., Magnér, J., Dahlgren, L., Westling, K., Wahlberg, C., Fortkamp, U., Söhr, S., Harding, M., Fång, J., Karlsson, J., 2019. Sustainable treatment systems for removal of pharmaceutical residues and other priority persistent substances. Water science and technology : a journal of the International Association on Water Pollution Research 79, 537–543.

Bayer, P., Heuer, E., Karl, U., Finkel, M., 2005. Economical and ecological comparison of granular activated carbon (GAC) adsorber refill strategies. Water research 39, 1719–1728.

Blumenstein, B., Bühle, L., Wachendorf, M., Möller, D., 2012. Economic assessment of the integrated generation of solid fuel and biogas from biomass (IFBB) in comparison to different energy recovery, animal-based and non-refining management systems. Bioresource technology 119, 312–323.

Boehler, M., Zwickenpflug, B., Hollender, J., Ternes, T., Joss, A., Siegrist, H., 2012. Removal of micropollutants in municipal wastewater treatment plants by powder-activated carbon. Water science and technology : a journal of the International Association on Water Pollution Research 66, 2115–2121.

Bundesministerium für Umwelt, Naturschutz und Reaktorsicherheit, 2002. Erste Allgemeine Verwaltungsvorschrift zum Bundes–Immissionsschutzgesetz (Technische Anleitung zur Reinhaltung der Luft – TA Luft). http://dgengineering.de/download/open/TA-Luft.pdf. Accessed March 22, 2019.

Çeçen, F., Aktaş, Ö., 2012. Activated carbon for water and wastewater treatment. Integration of adsorption and biological treatment. Wiley-VCH, Weinheim, Germany, Chichester.

Eggleston, S., 2006. IPCC guidelines for national greenhouse gas inventories. Institute for Global Environmental Strategies (IGES), Hayama.

FOEN, 2015. Gewässerqualität: Revision der Gewässerschutzverordnung. https://www.bafu.admin.ch/bafu/de/home/themen/.

Gabarrell, X., Font, M., Vicent, T., Caminal, G., Sarrà, M., Blánquez, P., 2012. A comparative life cycle assessment of two treatment technologies for the Grey Lanaset G textile dye: biodegradation by Trametes versicolor and granular activated carbon adsorption. Int J Life Cycle Assess 17, 613–624.

Goedkoop, M.J., Indrane, D., Beer, I.M. de, 2018. Product Social Impact Assessment Handbook - 2018, Amersfoort.

Gu, H., Bergman, R., Anderson, N., Alanya-Rosenbaum, S., 2018. Life Cycle Assessment of Activated Carbon from Woody biomass. Wood and Fiber Science 50, 229–243.

Hensgen, F., Richter, F., Wachendorf, M., 2011. Integrated generation of solid fuel and biogas from green cut material from landscape conservation and private households. Bioresource technology 102, 10441–10450.

Joseph, B., Hensgen, F., Bühle, L., Wachendorf, M., 2018. Solid Fuel Production from Semi-Natural Grassland Biomass—Results from a Commercial-Scale IFBB Plant. Energies 11, 3011.

Kim, M.H., Jeong, I.T., Park, S.B., Kim, J.W., 2019. Analysis of environmental impact of activated carbon production from wood waste. Environmental Engineering Research 24, 117–126.

Kounina, A., Wencki, K., 2015. LCA and LCC of the ozonation system in Neugut.

Mailler, R., Gasperi, J., Coquet, Y., Derome, C., Buleté, A., Vulliet, E., Bressy, A., Varrault, G., Chebbo, G., Rocher, V., 2016. Removal of emerging micropollutants from wastewater by activated carbon adsorption: Experimental study of different activated carbons and factors influencing the adsorption of micropollutants in wastewater. Journal of Environmental Chemical Engineering 4, 1102–1109.

Mansour, F., Al-Hindi, M., Yahfoufi, R., Ayoub, G.M., Ahmad, M.N., 2018. The use of activated carbon for the removal of pharmaceuticals from aqueous solutions: a review. Rev Environ Sci Biotechnol 17, 109–145.

Nicholson, F.A., Bhogal, A., Rollett, A., Taylor, M., Williams, J.R., 2018. Precision application techniques reduce ammonia emissions following food-based digestate applications to grassland. Nutr Cycl Agroecosyst 110, 151–159.

Oldenkamp, R., Beusen, A.H.W., Huijbregts, M.A.J., 2019. Aquatic risks from human pharmaceuticals—modelling temporal trends of carbamazepine and ciprofloxacin at the global scale. Environ. Res. Lett. 14, 34003.

Richter, F., Fricke, T., Wachendorf, M., 2010. Utilization of semi-natural grassland through integrated generation of solid fuel and biogas from biomass. III. Effects of hydrothermal conditioning and mechanical dehydration on solid fuel properties and on energy and greenhouse gas balances. Grass and Forage Science 65, 185–199.

Richter, F., Fricke, T., Wachendorf, M., 2011. Influence of sward maturity and pre-conditioning temperature on the energy production from grass silage through the integrated generation of solid fuel and biogas from biomass (IFBB): 2. Properties of energy carriers and energy yield. Bioresource technology 102, 4866–4875.

Richter, F., Graß, R., Fricke, T., Zerr, W., Wachendorf, M., 2009. Utilization of semi-natural grassland through integrated generation of solid fuel and biogas from biomass. II. Effects of hydrothermal conditioning and mechanical dehydration on anaerobic digestion of press fluids. Grass and Forage Science 64, 354–363.

ROFA CARBON, 2018. LCA & LCCA of Acquiring Virgin AC and of Thermal Regeneration. https://www.spire2030.eu/sites/default/files/users/user735/Deliverables/WP6/D6%201_Virgin%20activated%20carbon%20%20Thermal%20regeneration_v4_FINAL.pdf. Accessed September 23, 2019.

Rosenbaum, R.K., Bachmann, T.M., Gold, L.S., Huijbregts, M.A.J., Jolliet, O., Juraske, R., Koehler, A., Larsen, H.F., MacLeod, M., Margni, M., others, 2008. USEtox—the UNEP-SETAC toxicity model: recommended characterisation factors for human toxicity and freshwater ecotoxicity in life cycle impact assessment. Int J Life Cycle Assess 13, 532.

Rosenbaum, R.K., Hauschild, M.Z., Boulay, A.-M., Fantke, P., Laurent, A., Núñez, M., Vieira, M., 2018. Life Cycle Impact Assessment, in: Hauschild, M.Z., Rosenbaum, R.K., Olsen, S.I. (Eds.), Life Cycle Assessment. Springer International Publishing, Cham, pp. 167–270.

Saling, P., Perez, A.A., Kölsch, P., Grünenwald, T., 2020. Generation, Calculation and Interpretation of Social Impacts with the Social Analysis of SEEbalance®, in: Traverso, M., Petti, L., Zamagni, A. (Eds.), Perspectives on social LCA. Contributions from the 6th International Conference. Springer International Publishing, Cham, pp. 75–94.

Sparkes, R., 2008. Socially Responsible Investment, in: Fabozzi, F.J. (Ed.), Handbook of finance. Wiley, Hoboken, New Jersey, p. 73.

Turconi, R., Boldrin, A., Astrup, T., 2013. Life cycle assessment (LCA) of electricity generation technologies: Overview, comparability and limitations. Renewable and Sustainable Energy Reviews 28, 555–565.

UBA, 2015. Pharmaceuticals in the environment- avoidance, reduction and monitoring. http://www.umweltbundesamt.de/background-pharmaceuticals-in-the-environment. Accessed August 15, 2019.

UBA, 2018. Recommendations for reducing micropollutants in waters. www.umweltbundesamt.de/publikationen.

Wachendorf, M., Richter, F., Fricke, T., Graß, R., Neff, R., 2009. Utilization of semi-natural grassland through integrated generation of solid fuel and biogas from biomass. I. Effects of hydrothermal conditioning and mechanical dehydration on mass flows of organic and mineral plant compounds, and nutrient balances. Grass and Forage Science 64, 132–143.

Wang, J., Wang, S., 2016. Removal of pharmaceuticals and personal care products (PPCPs) from wastewater: A review. Journal of environmental management 182, 620–640.

Zhan, H., Zhuang, X., Song, Y., Yin, X., Wu, C., 2018. Insights into the evolution of fuel-N to NO precursors during pyrolysis of N-rich nonlignocellulosic biomass. Applied Energy 219, 20–33.