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Dynamic accounting of greenhouse gas emissions from cascading utilisation of wood waste

Giorgia Faraca^{*1}, Davide Tonini², Thomas F. Astrup¹

¹ *Department of Environmental Engineering, Technical University of Denmark, Miljøvej. Building 115, 2800 Kgs. Lyngby, Denmark*

² *European Commission, Joint Research Centre, Edificio Expo, Calle Inca Garcilaso 3, 41092 Seville, Spain*

*) Corresponding author: gfar@env.dtu.dk

21 **Abstract**

22 Cascading utilisation of post-consumer wood waste has recently gained increasing attention in the European
23 Union, aiming for a society in which the resource's properties are optimised through sequential uses. To date,
24 material utilisation of wood waste has been limited to particleboard production, with additional niche
25 alternatives being restricted by quality requirements for wood waste. In this consequential life cycle assessment
26 focusing on post-consumer wood collected at Danish recycling centres, Global Warming Potential (GWP)
27 impacts from quality-driven choices for cascading management of wood waste were compared with those from
28 handling mixed wood waste qualities. GWPs were modelled by considering the dynamic profile of greenhouse
29 gas emissions (including biogenic carbon dioxide) for two time horizons (100 and 500 years). The robustness
30 of the results was tested by varying modelling assumptions with respect to electricity system, wood sourcing
31 and associated rotation period, and impacts from indirect land use changes. The results demonstrated that
32 valuing quality over quantity in wood waste management can ensure larger GWP savings, especially if
33 recycling applications have a long lifetime and/or substitute energy-intensive products; such results were
34 confirmed under all scenario analyses. Inclusion of land use changes credited land-intensive products. More
35 cascade steps of the wood waste resource ensured larger savings; however, assumptions on the electricity mix,
36 on the source of the wood alongside the choice of the time horizon for GWP greatly influenced the results on
37 cascading management.

38 **Keywords:** wood waste, time-dependency, Global Warming Potential (GWP), dynamic life cycle
39 assessment (LCA), quality, resource cascading

40

41 **Introduction**

42 Wood waste is a versatile resource that can be used for both material and energy production. In 2007,
43 52% of the 36 million m³ of post-consumer wood waste recovered in the European Union (EU) was redirected
44 to material use (Leek, 2010), with the rest being sent to the energy sector. By 2030, wood waste is expected to

45 contribute with 59-67 million m³ to annual EU wood demand (Mantau et al., 2010). However, increasing
46 demand for this constrained resource and the presence of policy targets simultaneously encouraging recycling
47 (e.g. circular economy) and energy recovery (e.g. Renewable Energy Directives) may generate competition of
48 uses between sectors, potentially incurring suboptimal environmental performances. In this respect
49 environmental assessments, performed to support policies and decisions, have often delivered contradictory
50 results: some life cycle assessment (LCA) studies indicate that wood waste management via recycling is
51 environmentally favourable over incineration (e.g. Bais-Moleman et al., 2018; Borjesson and Gustavsson,
52 2000; Rivela et al., 2006a), although differences are not substantial. By contrast, Werner et al. (2010) conclude
53 that thermal treatment is preferable to recycling because it minimises the risk of dispersing pollutants contained
54 in the waste. Morris (1996) and Dodoo (2009) also indicate energy use as the preferable option.

55 The circular economy adopted by the EU favours a “resource cascading” approach (Sirkin and Houten,
56 1994), aiming for a society in which resource utilisation is optimized through sequential uses. This approach
57 involves two dimensions, namely quality and time: to extend the entire lifetime of the material, its properties
58 are subsequently used in different applications according to its quality grade (Fraanje et al., 1998). Wood waste
59 utilisation can be cascaded in time or in value: when cascaded in time, wood waste is remanufactured to the
60 same type of product in an iterative way, as many times as its quality permits. When cascaded in value, wood
61 waste is recycled into different product applications as its quality declines; therefore, early choices in the
62 resource lifecycle will more or less restrict future recycling applications and increase the overall lifetime of
63 the material. In resource cascading recycling and energy recovery may coexist, as the wood waste that is
64 recycled today will be combusted at the end of its lifetime (when its functions/properties have been fully
65 exploited). “Longer” cascades do not necessarily lead to larger environmental savings but only refer to the
66 potential of extending the overall lifetime of the resource: Dornburg and Faaij (2005), using carbon accounting,
67 showed that longer cascades increase or decrease net CO₂ emissions per hectare of biomass production
68 depending on the net CO₂ emissions of the respective application.

69 Post-consumer wood waste in the Nordic countries is mainly collected at recycling centres, which are
70 manned collection points where private citizens deliver their waste (Krook and Eklund, 2010). Currently,
71 material utilisation of so collected post-consumer wood waste is absorbed almost entirely into the production

of particleboard (European production was 34.8 million m³ in 2015; FAO/UNECE, 2016). This type of wood-based panel is made by chipping wood and pressing it into a mat with the addition of resins. Due to the simplicity of the manufacturing process, particleboard production can accommodate a large amount of post-consumer wood waste – up to 100% of the wood content depending on the country (in Denmark, it represents 80%; Vis et al., 2016). No wood waste inputs are typically expected in the recycling of paper, as it is considered an almost closed-loop sector (Höglemeier et al., 2015), but additional options for wood waste include solid wood applications (e.g. floor boards), cellulosic insulation materials, wood composites and bio-chemicals, albeit representing niche markets linked closely to the quality of the feedstock. According to the German legislation (Altholz V, 2012), post-consumer wood waste can be classified into four quality grades (Q1 to Q4) of which the two upper grades (Q1 and Q2) are suitable for recycling, the third grade (treated wood and wood-based panels, Q3) for incineration and the last (Q4) for disposal of as a hazardous material, due to being impregnated with preservatives (see Table B.1 in Appendix B). In Denmark, particleboard production may also accept input of Q3 wood waste as long as not exceeding thresholds for selected chemical elements. The possibility of unlocking other wood waste recycling applications via a quality distinction of the feedstock has not been explored to date.

Cascading utilisation of wood/wood waste has recently received increasing attention. With respect to existing LCA studies on the topic, Dornburg and Faaij (2005) demonstrated the potential for CO₂ mitigation of several cascading chain possibilities for short-rotation poplar. Sikkema et al. (2013) compared cascade chains for forest resources in Canada in terms of climate mitigation, suggesting that harvested wood should be used first for sawn wood, then for wood-based panels and finally energy recovery in order to decrease greenhouse gas (GHG) emissions. Bais-Moleman et al. (2017) demonstrated via environmental optimisation of EU forest sector utilisation that wood product cascading increases the efficiency of the resource use in general, but it can be less favourable in terms of GHG emission reduction in case it would divert the feedstock from the energy sector, that may compensate by using fossil fuels or virgin wood. With respect to wood waste, three studies (Höglemeier et al., 2014; Risse et al., 2017; Suter et al., 2016) assessed the impacts of its cascading utilisation via particleboard recycling in Germany and Switzerland, concluding that wood waste recycling can generate fewer environmental impacts than primary wood systems, although savings in Global

99 Warming Potential (GWP) decrease in line with additional cascade steps. Finally, Mehr et al. (2018) through
100 material flow analysis, LCA and optimisation modelling identified reduced GWP impacts from cascading
101 Swiss wood resources. A review by Thonemann and Schumann (2018) concluded that the environmental
102 soundness of wood cascading has yet to be proved. While comparing cascading of wood products to primary
103 wood production systems, the abovementioned studies have not addressed how quality-driven choices for
104 wood waste management may potentially affect the GWP performance of the system. This implies selective
105 collection/handling and recycling favouring quality in place of quantity. An assessment of potential savings
106 from cascading possibilities of upper qualities wood waste compared to handling mixed wood waste is missing
107 in the literature. Studies on cascading of post-consumer wood waste from recycling centres are also missing.

108 LCA studies of biomass systems especially discuss impacts in terms of the indicator GWP, because
109 wood resources are essentially made of biogenic carbon, therefore potentially providing climate change
110 mitigation by displacing fossil fuels through energy recovery and carbon storage through material substitution,
111 in addition to substituting other non-renewable materials which are usually manufactured through carbon-
112 intensive processes, such as steel and concrete. However, two fundamental aspects related to GWP accounting
113 appear to have been largely overlooked. First, emissions of biogenic CO₂ from wood (waste) combustion are
114 not accounted for, because they are assumed to be re-sequestered as new plantations grow. The consequence
115 is that emissions of carbon dioxide following biomass combustion are implicitly considered neutral. However,
116 recent publications (e.g. Cherubini et al., 2011; EC, 2016a; Guest et al., 2013) have demonstrated that wood
117 combustion should not be considered carbon-neutral when constituted by biomass species with a rotation
118 period of several decades, therefore questioning conclusions obtained under a carbon neutrality assumption.
119 The second limitation is the lack of temporal information: emissions occurring during the life cycle are
120 modelled as an aggregated value irrespective of the time at which they occur. This equates to considering all
121 emissions as if occurring at year zero and then summing them up over the selected time horizon. A dynamic
122 profile of the emissions is rarely considered, despite De Rosa et al. (2017) and Levasseur et al. (2010, 2013)
123 demonstrated that neglecting such emissions occurring late in the time horizon of the scenarios may lead to
124 biased interpretation. A review by Røyne et al. (2016) found that 97% of 101 articles on wood or forest systems
125 LCAs did not consider timing of GHG emissions, and only ten articles clearly included biogenic CO₂

emissions. None of the aforementioned cascading wood studies considered timing of GHG emissions, and only Mehr et al. (2018) included biogenic CO₂ impacts, albeit through a simplified approach.

Acknowledging these shortcomings of existing LCA literature on wood waste management and in the attempt of advancing the knowledge of the associated environmental impacts, the aim of this study is two-fold: i) to systematically assess the GWP of alternative wood waste cascading systems – we do this by considering an emission profile for each scenario and by performing a dynamic accounting where the GWP of biogenic carbon emissions is assessed building on state-of-the-art approaches; ii) to highlight and discuss the role played by the quality of wood waste with respect to the potential climate change mitigation benefits – we do this by comparing recycling options for mixed wood waste qualities versus the case in which the different qualities are handled separately and follow appropriate management routes. We illustrate the results under different scenario assumptions with respect to time horizon, electricity system, wood sourcing and associated rotation periods, and indirect land use change impact. The focus of the study is on Danish wood waste from recycling centres.

2. Materials and methods

2.1 Goal, scope and functional unit

The goal is to assess the GWP (IPCC 2013) of alternative scenarios for post-consumer wood waste management and to compare them against current practice. The functional unit (FU) adopted in the modelling is the management of 1 tonne of post-consumer wood waste collected at Danish recycling centres. Post-consumer wood waste is not a homogenous material but it does consist of a variety of items differing in source, type and quality (Faraca et al., 2017) and can be contaminated by presence of wood impregnated with preservatives and other material fractions. Following the German legislation (Altholz V, 2012), the quality composition of the wood waste in this study is 4% Q1, 56% Q2, 32% Q3 and 6% Q4, the rest being other material fractions (impurities; see Table S1 in SI). Wood waste of grades Q1, Q2 and Q3 is generally collected in a single container at Danish recycling centres, whereas grade Q4 follows other disposal routes (landfill or incineration in special plants) and its presence was accounted for as impurity. The geographical scope is

152 Denmark; technology efficiencies/emissions are assumed to be in line with 2015-2030 expected performances.
153 For the temporal profile of emissions, we assume 2015 as the starting year of the assessment. The study follows
154 a consequential approach, as it intends to model the consequences in terms of GWP of implementing a portfolio
155 of wood waste management schemes. Considering that wood waste represents a minor amount compared to
156 the wood traded for goods and energy (see Annex 5 of Gurria et al., 2017), it is assumed that the changes
157 induced by such waste management schemes do not affect the determining parameters of the overall market
158 situation (direction of the trend in market volume and the constraints on and production costs of the involved
159 products/technologies; see also Weidema et al., 2009).

160

161 **2.2 System boundaries and scenarios definition**

162 The system boundary (Figure 1) starts after the collection phase and ends at the point where the waste
163 loses its functional properties and can no longer be recycled, but only disposed of through incineration or
164 landfilling. Two sets of scenarios were developed, based on whether or not the collection (and the management
165 of wood waste) reflected the quality distinction of the material: I) four “Wood Waste, mixed qualities”
166 scenarios (WWmix, A1 to A4; the four scenarios represent the possibility to perform up to four particleboard
167 cascades), where wood waste is collected following current practice (one container for Q1+Q2+Q3 which is
168 sent to particleboard production; Q4 is present as impurity); and II) eleven “Wood Waste, quality distinction”
169 (WWquality, B1 to F1) scenarios, where the distinction in quality classes is in place through separate collection
170 (one container for Q1+Q2, which is sent to recycling, and one for Q3, which is sent to incineration; Q4 is
171 assumed to be collected in a container for impregnated wood and sent to incineration at special plants). In
172 WWquality scenarios alternative recycling options for Q1+Q2 were tested, while Q3 and Q4 were always
173 assumed to be incinerated in municipal solid waste incinerators (MSWI) and special plants, respectively
174 (typical fate in Denmark); thus, their management does not differ across scenarios. All in all, we considered
175 the following possible recycling applications of the collected post-consumer wood waste:
176 - Production of particleboard (PB). Wood waste undergoes pre-treatment (shredding, sieving, milling, and
177 sorting of metals), then it is dried until ~6% moisture content (mc – as wet weight, ww) and sprayed with urea-

178 formaldehyde resin before being hot-pressed into a mat. Wood waste constitutes 80% ww of the feedstock for
179 particleboard in Denmark (Vis et al., 2016), while urea-formaldehyde resin constitutes typically 10% ww of
180 the final product (Wilson, 2010; Rivela et al., 2006b).

181 - Production of floor boards. Wood waste of usable dimensions is manually made impurity-free, sawn to 2.5
182 cm thickness and planed.

183 - Production of wood insulation board (WIB). Wood waste is processed into chips, heated and fed to a
184 defibration process. The fibres are dried in a cyclone and mixed with 1.3% ww bi-component fibres
185 (polyethylene or polypropylene) and 3.6% ww adhesives and fire retardants (González-García et al., 2009;
186 Steico, 2016).

187 - Production of wood plastic composites (WPCs). The wood waste is shredded, dried, and grinded until
188 producing <1mm particles with 10% mc. The wood particles are blended with plastic granules (HDPE assumed
189 from secondary materials) at a 1:1 ratio and a coupling agent and compounded at 170 °C. The last step is
190 compression moulding (Bolin and Smith, 2011; Sommerhuber et al., 2017).

191 - Production of pellets. Although this scenario is typically performed on industrial residues or Q1 wood waste,
192 it was assumed possible also for Q2, as previous studies have confirmed that chemical composition of wood
193 residues of Q1 and Q2 is comparable (Faraca et al., 2017). Wood waste undergoes shredding and grinding
194 before drying until 8-10% mc and pelletising. The product is burnt in a combined heat and power (CHP) plant.

195 Any wooden by-products originating from the wood waste reprocessing stage were assumed to be
196 combusted in small-size biomass boilers provided with heat recovery. Foreign materials found in the collected
197 wood waste were assumed to be separated at collection points and sent for recycling (ferrous and non-ferrous
198 metal impurities), landfilled (glass, stones, composite building materials), and incinerators provided with
199 electricity and heat recovery (plastics, textiles, cardboard, garden waste). Fly and bottom ashes originating
200 from the combustion and incineration processes were assumed to be disposed of in dedicated landfills for fly
201 and bottom ash, respectively.

Figure 1. Schematic system boundaries of the scenarios under assessment. PB = particleboard; MSWI = municipal solid waste incinerator; WIB = wood insulation boards; WPC = wood plastic composites. In WWquality scenarios the management of Q3 and Q4 is maintained constant as depicted exemplarily for scenario B1: Q3 is incinerated in MSWI, Q4 is incinerated in plants treating hazardous waste. Dashed lines represent the system boundaries of the system. Detailed system boundaries for each scenario can be found in Appendix C.

206 **2.3 Identification of marginal processes and technologies**

207 Any additional product generated during the management of the wood waste was credited with the
208 substitution of corresponding market products, applying system expansion as common practice in waste LCAs
209 and recommended by ISO (2006). Following consequential LCA principles, this implies identifying the
210 marginal suppliers (Weidema et al., 2009). For the provision of electricity, the marginal source for Denmark
211 was assumed to consist of 61% wind energy and 39% biomass-energy (in the form of wood pellets) conforming
212 to the figures provided by Munõz et al. (2015) and EC (2016b). Heat was assumed to be produced from natural
213 gas-based boilers, this being the most likely thermal energy source for individual producers (in the absence of
214 a connection to district heating). The marginal product for insulation was identified in glass wool, on the basis
215 of the information reported in Kalt et al. (2016). With respect to the marginal source of the wood, it should be
216 noticed that, along with post-consumer wood waste to be managed, two additional wood biomasses were
217 involved in the assessment: the virgin wood products whose provision and manufacture is avoided through
218 wood waste recycling (scenario A1 to C4 and E1; see Figure 1 and Appendix C), and wood pellets constituting
219 a share of the marginal mix for electricity supply (39% of the total) and also representing one of the wood
220 waste recycling applications (scenario F1; see Figure 1). Currently, wood for particleboard, WPC, and pellets
221 production is typically sourced from forest slash or residues from other wood products manufacture (e.g. Mehr
222 et al., 2018; Rivela et al., 2006b; Wilson, 2010). These residues, however, should not be assumed as the
223 marginal source for wood, as they are not capable of responding to changes in demand (being residues their
224 production is constrained). To reflect this problem, and to illustrate the results under different assumptions, we
225 identified two sourcing alternatives: I) forest wood from surrounding countries with large resource availability
226 (Latvia, Sweden, and Russia), typically having a rotation time between 40 and 100 years; for this, we assume
227 a rotation time of 60 years for use in the baseline calculations. II) short-rotation plantations, exemplified by
228 Brazilian eucalyptus forest, being suggested as a likely future source in previous studies (Schmidt and Brandao,
229 2013; Thrän et al., 2017); for this, we assume a rotation time of 10 years as an alternative to the baseline
230 assumptions (see Section 2.6.3). While the first case reflects the current trend for wood sourcing and imports
231 to Denmark, the second represents a possible future evolution of the market.

232

233 **2.4 Assessment method**

234 **2.4.1 Dynamic accounting**

235 The dynamic accounting was facilitated by MATLAB and followed the mathematical approach
236 described in Cherubini et al. (2013, 2015) and Levasseur et al. (2010): each activity in the inventory is assigned
237 the year in which the emission takes place. For example, the activity “production of particleboard” in scenario
238 C2 occurs at year 50. When a GHG is emitted to the atmosphere it degrades according to a GHG-specific
239 atmospheric decay curve (the impulse response function, IRF). Therefore, at each year, the quantity of a GHG
240 emitted by the system is multiplied by the atmospheric decay curve of a unit pulse emission of the GHG under
241 assessment (see Figure 2). IRFs for CO₂ are described in Section 2.2.2; IRFs for other GHGs were retrieved
242 from Levasseur et al. (2010). The yearly curves are summed up to a total curve. The integral over time of the
243 total curve divided by the integral over time of a unit pulse emission of CO₂ emitted at year 0 (often referred
244 to as the absolute global warming potential, AGWP_{CO2}) represents the GWP (kg CO₂-eq/FU) for that system.
245 This methodology could equally be carried out by calculating dynamic characterisation factors (dCF), which
246 are obtained by calculating the integral over time of the atmospheric decay curve of a *unit pulse* emission
247 emitted at year *t* divided by the integral over time of the decay emission of a unit pulse emission of CO₂ emitted
248 at year 0. Yearly dCFs are then multiplied by yearly emissions; the products are summed and represent the
249 GWP (see also Appendix E.5).

250 Since (biogenic and fossil) CO₂ emissions in biomass systems account for the majority of total GHG
251 emissions, other air emissions with a warming effect were modelled separately (Section 2.4.3), also given their
252 temporary behaviour in the atmosphere compared to the persistency of CO₂ (IPCC, 2013). Notice that the
253 dynamic profile of emissions was applied both to the foreground and the background system. However, with
254 respect to the latter, we did not assume a dynamic development of the energy (e.g. for electricity, heat, and
255 transport fuels) supply mix over time, i.e. these were maintained constant over the years. This was done to
256 isolate and highlight the effects related to the management of the wood waste *per se* and to reduce the number

of variables in the system. The overall analysis was then carried out using the LCA-model EASETECH (Clavreul et al. 2014).

2.4.2 Fossil and biogenic carbon accounting

The behaviour of CO₂ in the atmosphere is the same irrespective of the origin of the CO₂ (biogenic or fossil), but the accounting depends on the carbon pools considered for the C cycle model (Cherubini et al., 2011). According to IPCC (2013), the Bern carbon cycle 2.5CC (Forster et al., 2007) was used for the dynamic accounting of fossil and biogenic CO₂, as it considers the interaction of all compartments in the ecosphere. In the case of fossil CO₂ emissions, Eq. 1 was used to calculate the IRF:

$$f(t) = y_{CO_2}(t) = A_0 + \sum_{i=1}^{i=n} A_i e^{(-\frac{t}{\tau_i})} \quad n = 1,2,3 \quad Eq. 1$$

where $f(t)$ is the impulse response function (IRF) after the emission (i.e. the atmospheric decay, IRF of CO₂ fossil) and A_i and τ_i are coefficients (see Appendix D)

In the case of biogenic CO₂, Eq. 2 was used to calculate the IRF (based on Cherubini et al., 2011, 2013, 2016):

$$f(t) = y(t) - \int_0^{TH} g(t)y(TH - t)dt \quad Eq. 2$$

where $f(t)$ (IRF of CO₂ bio) is obtained by the convolution of the CO₂ emissions $y(t)$ released at time t and the CO₂ sequestered by the biomass regrowth $g(t)$ in the period from t to TH . The sequestration function $g(t)$ depends on the rotation period of the wood species in question, as detailed in Cherubini et al. (2011) (see appendix D). Conformingly with the latter, we modelled biomass regrowth in the forest as a normal (Gaussian) distribution.

As the cascading utilisation of wood waste involves preserving the material properties (hence C content) for a certain period, the storage of biogenic carbon in the anthroposphere can be included in the IRF of CO₂ bio as suggested by Guest et al. (2012). Eq. 3 was used to calculate the IRF:

$$f(t) = \begin{cases} f1(t) = \int_0^{\tau} g(t) y(TH - t) dt, & 0 \leq t < \tau \\ f2(t) = y(TH - t) - \int_{\tau}^{TH} g(t) y(TH - t) dt, & t \geq \tau \end{cases} \quad \text{Eq. 3}$$

where τ is the time period during which the wood waste in the FU is kept in the anthroposphere, i.e. the time at which the wood waste can no longer be recycled and is incinerated; it varies across scenarios depending on the lifetime of the recycled product and the number of cascades of the FU.

The IRFs of CO₂ (Eq. 1, 2, 3) are then multiplied by the emission quantity at each year i of the system (B_i) and summed up to a total curve ($F(t)$) of the atmospheric behaviour of CO₂ released by the system under assessment (Eq. 4):

$$F(t) = \sum_{i=0}^n B_i f_i(t) \quad n = 0, 1, 2, \dots, TH \quad \text{Eq. 4}$$

$$GWP_{CO_2}^t = \frac{\int_t^{TH} \alpha_{CO_2} F(t) dt}{C_0 \int_0^{TH} \alpha_{CO_2} y(t) dt} \quad \text{Eq. 5}$$

where α is the radiative efficiency of CO₂ and C_0 is the unit pulse emission. For further details refer to Cherubini et al. (2011), Levasseur et al. (2010), Guest et al. (2012) and Appendix D.

The modelling considered a time horizon (TH) for the integration of the CO₂ emissions equal to 100 years, i.e. for the period 2015-2114 (2114 is the last year) and to 500 years (2015-2514).

293

2.5 Life Cycle Inventory

Inventory data were retrieved from a number of publications and reports available in the literature. All data were taken into account by defining parameter distributions (see Section 2.6.1) which can be found in Section S6.2 of SI. Data for the particleboard production process was retrieved mainly from Merrild and Christensen (2009), Rivela et al. (2006b) and Wilson (2010). The main difference in modelling subsequent iterations of particleboard production was that half of the feedstock in input is lost at every recycling step, due to the dimensions of the chips and the presence of resin that may cause problems during shredding (Vis et al., 2016). Separation of qualities in scenarios B1-B4 led to higher technical yield (100% instead of 94%) and substitution factor (1 instead of 0.6) of the first iteration of particleboard production compared to scenarios A1-A4; following iterations were modelled identically (see Appendix E). The substitution process for

particleboard production from virgin sources was modelled with the same data, the main difference being a higher technical yield and a higher wood feedstock moisture content. The complete list of processes used in the modelling can be found in Appendix E.

According to Section 2.4, we calculated emission profiles for biogenic and fossil CO₂ for each individual scenario investigated. While the detailed emission inventory may be found in Appendix E.5 (along with corresponding dCFs), Figure 2 illustrates the emission profile of biogenic and fossil CO₂ for the case of scenario B1 as an example.

Figure 2. Time-dependent CO₂ decay curves in the atmosphere illustrated exemplarily for scenario B1; (a) and (b) graphs differ in the level of information aggregation. The curves in (a) represent different behavioural curves $f_i(t)$, as described in Section 2.4.2; the black solid curve in (b) represents $F(t)$. FU= functional unit; i= year at which the emission occurs; τ = storage time.

2.6 Uncertainty, sensitivity and scenario analysis

2.6.1 Combined sensitivity and uncertainty analysis

Following the approach suggested by Bisinella et al. (2017), uncertainty on the foreground system (the waste management activities) was tested by thorough uncertainty propagation and sensitivity analysis. For the

uncertainty propagation it was necessary to define LCI data as probability distributions. Due to the diversity of the information available in the literature, triangular distributions of the type (min,max,mode) were assumed. When the mode could not be calculated, the median was used instead (Tables E.18 and E.19 in Appendix E). The mode/median represented the baseline value while the entire distribution was used for the uncertainty propagation, which consisted of propagating the uncertainties associated with each input parameter to calculate the uncertainty associated with the results, covering the possibility of accumulative effects due to the changes in two or more parameters simultaneously. This was achieved by Monte Carlo analysis; the uncertainty in the results was illustrated by means of error bars. For the sensitivity analysis, each parameter was increased by 10% of its value in a “one-at-a-time” manner while keeping all other parameters fixed at their baseline values. Sensitivity coefficients and sensitivity ratios associated with each parameter were calculated (Eq. F.1 and F.2 in Appendix F), identifying how results vary as a consequence of a change in input values (Bisinella et al., 2016).

334

335 **2.6.2 Scenario analysis I – Testing assumptions on electricity system**

For the modelling, some assumptions on the background system (the system that interacts with the foreground system supplying energy and materials) were necessary. These concerned the type of electricity supply mix and the source of wood for material (e.g. particleboard, WPC) and energy (i.e. pellets) production. The robustness of the LCA model to such assumptions was tested by scenario analysis. Regarding the electricity mix, a “fossil” version (26% gas, 4% oil and 70% coal) was used in all parts of the modelling to test the baseline assessment results, including the production of alternative materials.

342

343 **2.6.3 Scenario analysis II – Testing assumptions on wood sourcing and rotation**

Regarding the source of wood, the baseline assumption whereby wood is sourced from Nordic countries (rotation time 60 years, see Section 2.1.2) was supplemented by an import scenario in which Eucalyptus wood is supplied from Brazil (rotation time 10 years). In addition, given the uncertainty associated with the choice of the rotation period of the wood, results were also calculated for ten rotation times between 10 and 100 years (i.e. every ten years), to cover a wider span of possibilities.

349

350 **2.6.4 Scenario analysis III – Inclusion of ILUC and other GHGs**

351 From a consequential modelling perspective, growing wood plantations incurs demand (and further
352 occupation) for land. According to Schmidt et al.(2015), this implies the existence of a market for managed
353 forest land, responsible for a share of the global land use change (LUC) CO₂ emissions from deforestation; the
354 rest being attributed to cropping and pasture, i.e. mainly agriculture. This impact is typically called indirect
355 LUC (iLUC) as it represents the upstream consequences (transformation of land somewhere else, even beyond
356 the assessed geographic system) of demanding the land, regardless of its final use. Avoiding (e.g. with
357 increased recycling and displacement of virgin material) an additional demand for wood products should then
358 be credited with the corresponding iLUC CO₂ impact avoidance. In this study, the approach presented by
359 Schmidt et al. (2015) was followed. The model assumes that the land use impacts from growing a forest depend
360 on the amount of land demanded and the type and share of expansion on virgin nature taking place
361 (transformation), assuming full-elasticity of supply (i.e. short-term effects due to price fluctuations are not
362 included). Unlike crops, when wood residues are removed, no additional mineral fertiliser is assumed to be
363 used to counterbalance the nutrient content of the removed wood residues (Schmidt et al., 2015). To illustrate
364 the potential contribution from iLUC impacts we used a value of 0.32 kg CO₂/kg wood for long-rotation
365 Latvian wood and of 0.05 kg CO₂/kg wood for short-rotation eucalyptus in Brazil (both derived after Schmidt
366 and Brandão, 2012). Note that these figures have also been applied in a recent study by Vadenbo et al. (2017).
367 To single out the importance of these effects, iLUC contribution will be reported and discussed separately.
368 Similarly, the contribution from other GHGs will be kept separate from CO₂ (see Section 2.2.1) and shown as
369 additional scenario analysis.

370

371 **3. Results and discussion**

372 **3.1 Contribution to Global Warming Potential**

373 The characterised GWP results are shown in Figure 3 (rotation time 60 years). Figures below zero
374 represent savings, while above zero burdens; we name the sum of savings and burdens “impact/net savings”.

375 Overall, all scenarios provided savings except for scenario F1 (pellet-CHP); scenario D1 (recycling to WIB)
376 provided the largest savings. Except for the case of F1, the set of scenarios “WWquality” always provided
377 larger savings than the “WWmix” set, highlighting the benefits of a quality-driven wood waste management
378 versus maximising the quantity sent to recycling. Among the “WWmix” set, scenario A1 (recycling to
379 particleboard with incineration of the recycled product at the end of its lifetime) performed worst because the
380 benefits from substituting virgin particleboard did not counterbalance the burdens from reprocessing; this was
381 due to the low substitution factor (0.6) for the recycled particleboard alongside the relatively low impact of
382 particleboard manufacture from virgin wood relative to that from recycled materials. Impacts associated with
383 biogenic CO₂ emissions following incineration of the recycled particleboard at EoL constituted 60% of
384 burdens. Additional recycling iterations of the particleboard at EoL increased the savings. Scenarios B1-B4
385 incurred larger savings than the corresponding scenarios A1-A4 (e.g. -88 kg CO₂eq/FU in B1 compared with
386 -11 kg CO₂eq/FU in A1), owing to a higher substitution factor for the first iteration of recycled particleboard
387 production, despite the burdens deriving from biogenic CO₂ emissions following combustion of the lower
388 quality wood (Q3+Q4; 72% of burdens). Although subsequent iterations of particleboard production in
389 “WWquality” scenarios were modelled under the same assumptions of scenarios A1-A4, scenarios B1-B4
390 showed larger savings, still owing to the credits obtained by the first cascade step.

391 When looking at applications other than particleboard, recycling to WIB showed the best performance
392 followed by recycling to flooring: the final net savings were 640% and 446% larger than for recycling to
393 particleboard (scenarios D1 and C1 compared to B1, respectively). Such net savings appeared much larger
394 when compared to the “WWmix” scenarios: net savings in scenarios D1 and C1 were as much as 58 and 35
395 times larger compared to scenario A1. This was mainly due to the larger savings obtained from substituting
396 more energy-intensive products such as virgin wood floorings and insulation glass wool alongside recycling
397 to products characterised by a longer lifetime, thereby increasing GWP100 savings thanks to carbon storage.
398 Indeed, material substitution in scenarios D1 and C1 contributed to 66% and 49% of savings respectively,
399 following a lifetime of the products estimated at 30 and 50 years, respectively (see Figure 1). On the other
400 hand, recycling to WPC composites (scenario E1), despite having a lifetime equal to WIB (30 years), showed
401 a considerably reduced GWP mitigation compared to scenario D1 (-538% of net savings) and in the range of

scenario B2. This was mainly because the production of virgin WPC did not require substantially more energy than the recycling process, thereby indicating that long lifetimes *per se* do not necessarily ensure large savings. Nevertheless, savings in scenario E1 were larger than the “WWmix” scenarios. Recycling of wood waste into pellets (scenario F1) was the only option with a net burden on GWP, mainly due to biogenic emissions from combustion of the wood (407 kg CO₂-eq) that were not offset by substitution of energy (349 kg CO₂-eq). While savings derived from combusting wood appeared significant in previous studies (e.g. Suter et al., 2016), this was however the result of assuming displacement of fossil-based energy and of not considering biogenic CO₂ emissions.

Figure 3. Characterised GWP results for the scenarios under assessment; a) GWP100 and b) GWP500; only CO₂ emissions included; rotation time of the wood: 60y; “Other” includes metal recycling and landfill.

3.2 Cascading effects

The results showed that within each group of scenarios sharing the same recycling application (i.e. within A1-A4, B1-B4 or C1-C4) the more the cascade steps, the larger the savings obtained: net savings brought by four cascade steps could be more than eight times larger than one-cascade systems (e.g. comparing scenario A4 to A1). However, the incremental savings provided by an additional cascade use declined at each step: from 78% (A2 compared to A1), to 43% (A3 to A2), to 15% (A4 to A3), suggesting an asymptotic trend.

421 The reason was that at every cycle half of the mass was lost during particleboard manufacture, owing to a
422 decrease in quality of the feedstock: an indicator of this was that at the third cycle of particleboard recycling
423 (scenarios A3, B3 and C4) 30% of the mass fed into the recycling process was resin. This means that the
424 activity providing the largest savings in a system occurred at the first cascade step, when the quality of the
425 resource is at its highest point, thereby demonstrating that recycling choices at the early stages of a wood
426 resource's cascade chain are the most important from a GWP perspective. Other studies (Höglmeier et al.,
427 2014; Risse et al., 2017; Suter et al., 2016) demonstrated that savings brought by an additional particleboard
428 cascade in the resource's use can be modest and could even worsen total impacts. However, as mentioned
429 earlier, these studies did not include a dynamic evaluation nor considered impacts from biogenic CO₂.

430

431 **3.3 Importance of quality**

432 Comparing “WWmix” against “WWquality” scenarios illustrates that recycling only the upper qualities of
433 wood waste could be beneficial with respect to the GWP performance of the entire system, i.e. the best option
434 in our model was not the one maximising the wood mass sent to recycling but the quality of the resource,
435 despite the burdens from combusting the lower-quality wood (Q3+Q4; Figure 1). For example, in scenarios
436 B1-B4 a combination of incineration and quality-oriented recycling provided larger savings than the
437 corresponding “WWmix” scenarios, despite 651 kg of wood waste are recycled in scenario B1 compared to
438 the 1000 kg in scenario A1. In scenario D1, only 42% of the FU was manufactured into a new product;
439 nevertheless, this scenario showed the best GWP performance.

440 Collecting wood waste according to its quality opens up the possibility of multiple recycling applications
441 that may generate better environmental performances compared to the case of “WWmix” where particleboard
442 production is the only possible option. Results from Fig. 3 suggested that wood waste recycling should be
443 directed toward applications substituting energy-intensive materials or extending the overall lifetime of the
444 wood resource, as floor boards and WIB appeared to be the case. As Dornburg and Faaij (2005) also
445 demonstrated, one cascade recycling to a high-quality application can result in larger savings than many
446 cascades to a lower-quality application, as is the case in our study for scenarios D1 compared with A1-A4 or

447 B1-B4. Additionally, the absence of adhesives in some of the higher-quality recycling applications (in this
448 case floor boards) would in principle allow for larger savings also in other impact categories, since other studies
449 (e.g. Höglemeier et al., 2014; 2015; Suter et al., 2016) demonstrated that the production and use of such
450 chemicals for particleboard making represent the largest contribution in most impact categories.

451

452 **3.4 Dynamic accounting and biogenic CO₂**

453 The time dependency of CO₂ impacts made results heavily depend on the TH chosen for the assessment.
454 When a TH of 500 years was selected, the magnitude of the results differed considerably from the GWP100
455 case: in general, savings decreased by 31-135% depending on the scenario, except for scenarios A1, B1, E1
456 and F1, where savings were 1.5-5 times larger (Figure 3b). “WWquality” scenarios provided savings 2-50 times
457 larger than “WWmix”, depending on the scenario. The ranking of scenarios was also affected: for example,
458 additional cascade steps worsened the results (scenarios A1-A4, B1-B4, C1-C4). This reflects the fact that,
459 when considering a GWP assessment over 100 years, most of the savings in the system are related to the
460 biogenic CO₂ because of avoiding sourcing of wood for material production and the associated biomass-energy
461 consumption. However, such savings become minor with long THs (e.g. 500 years), as the effect of biogenic
462 carbon storage and emissions tend to fade away and have minor importance (biogenic CO₂ savings decreased
463 by 52-89% depending on the scenario). On the other hand, impacts from fossil CO₂ (mainly burdens) are much
464 more prominent. This confirms that biogenic CO₂ long-term effects are negligible in comparison to fossil CO₂,
465 and emissions from biomass (harvested and re-planted) may be considered carbon-neutral in a very long-time
466 perspective. Results for TH=500 years are indeed in accordance with what found by previous studies that did
467 not consider biogenic CO₂ (Höglmeier et al., 2014; Risse et al., 2017; Suter et al., 2016). Scenario F1 changed
468 from being a load onto the environment to being a saving even larger than scenarios C1-C4 (F1 savings
469 increased by 4 times compared to the TH=100-years case). This shows that combustion of biogenic CO₂ is
470 indeed beneficial with respect to the indicator GWP when very long timeframes (e.g. 500y) are considered.
471 However, recent publications questioned the usefulness of metrics for TH above 100 years, due to the strong
472 assumption of constant background conditions (Levasseur et al., 2016), thereby challenging the climate

mitigation potential of biomass-energy. These results indicate that the TH of the assessment should be carefully selected according to the scope of the study. In LCAs a TH of 100 years is commonly assumed, although the IPCC (2013) states that “there is no scientific argument for selecting 100 years compared to other choices”. Røyne et al. (2016) found that less than 10% of forest LCAs published in 1997-2013 considered time perspectives other than 100 years. This is supported by the fact that policies and strategies for climate change mitigation generally reflect shorter time perspectives, and by IPCC (2014) that discarded the 500 years case from the list of THs for which providing parameters values. Additional arguments questioning the robustness of very long THs are brought by Reisinger et al. (2010) and Joos et al. (2013) that showed how uncertainties on the IRF for CO₂ increase with increasing time horizon. All in all, our choice of assessing also a TH of 500y follows the recommendations from Cherubini et al. (2016b) and Levasseur et al. (2016) to provide results for various time horizons. As in our study a 20-year-TH was too low due to the typically longer lifetime of the wood products investigated, a TH of 500 years was instead chosen to illustrate the time dependency on the GWP results. .

3.5 Uncertainty, sensitivity and scenario analysis

3.5.1 Combined uncertainty and sensitivity analysis

The propagation of input uncertainty widened the results for all scenarios, and the obtained score range overlapped for scenarios A1-A4, B1-B4 and E1 (Figure 4). Therefore, it should be considered for decision-making that the ranking of scenarios could vary and increased effort should be put into obtaining data tailored to the scope. However, conclusions on scenarios C1-C4, D1 and F1 (i.e. those providing the largest and smallest savings) appeared robust. The uncertainties of the results did not increase with increasing number of cascades, but rather appeared constant throughout the cascading system. This was because, despite another “layer” of uncertainty is added by including an additional cascade step, the mass of the material involved in the extra steps decreases (or even halves). Table H.1 in Appendix H shows the results of the perturbation analysis. For all scenarios covering the material use of wood waste the most sensitive parameter was the technical substitution factor of the recycled product manufactured in the first cascade system, irrespective of

499 the number of cascades considered in the system. Energy conversion efficiencies at power plants also played
500 an important role, and were clearly the most important parameters for scenario F1. The model appeared less
501 sensitive to parameters related to energy consumption in the material manufacture process.

502

503 **3.5.2 Scenario analysis I – Testing assumptions on electricity system**

504 Figure 4 illustrates how characterised results changed as a consequence of different scenario analyses.
505 When a fossil fuel-based electricity mix was assumed significantly larger savings were observed for all
506 scenarios compared to the electricity mix (wind and wood) assumed in the baseline, regardless of the time
507 horizon. This was a consequence of the very large savings provided by incineration when the recovered
508 electricity replaced coal, oil and gas. The “WWquality” scenarios still performed better than the “WWmix” in
509 all cases. The ranking of scenarios did alter: scenario F1 (pellets-CHP) changed from being a net load onto the
510 environment to providing larger savings than scenarios A1-A4, B1-B4 and E1. Additional cascade steps
511 resulted in lower total savings (e.g. scenarios A2-A4 compared to A1); the decrease in incremental savings
512 given by additional cascades was confirmed and with a more or less steep trend depending on the TH. Under
513 this assumption for electricity mix, the timing of emissions had a negligible effect on the results, which
514 appeared to be driven by savings from avoided electricity production only. This indicates that the energy use
515 of wood may be favourable when displacing fossil fuel-based energy. Yet, this is unlikely to be the future
516 marginal electricity source in Denmark or the EU, given the commitment to ambitious renewable energy
517 targets.

518

Fig 4. Global Warming Potential results for TH=100 years (a) and TH=500 years (b) under baseline conditions (only CO₂ emissions included; grey vertical bars, error bars represent the min and max value obtained by uncertainty propagation; rotation time of the wood 60y), fossil-based electricity provision (red-cross mark), sourcing of wood from Brazil (dark-red triangular mark), inclusion of other GHGs (cyan-triangular mark), inclusion of iLUC (crossed-lilac diamond mark; assuming a rotation time of the wood of 60y). Refer to Figure 1 for definition of scenarios.

3.5.4 Scenario analysis II – Inclusion of iLUC and other GHGs

The inclusion of iLUC greatly affected the results for most of the assessed scenarios (Figure 4), incurring considerably larger savings. The contribution of iLUC generally accounted for 38-48% of savings for TH=100 years and 52-64% for TH=500 years. This is in line with the figures reported in Hoeglemer et al. (2014) for land transformation, thereby suggesting that the substitution of energy-intensive products does not ensure the highest savings when compared to substitution of land-intensive products (scenario D1 ranked second-to-last when iLUC effects are included). These results should be used cautiously as only illustrative of the potential iLUC contribution under the assumption of sourcing wood from long-rotation forests (baseline used in this study; rotation 60 years); the iLUC factor for wood derived from short-rotation plantations would be significantly lower, e.g. 0.05 kg CO₂/kg wood from Brazilian eucalyptus (Schmidt and Brandao, 2012; Tonini et al., 2017), thereby decreasing the overall iLUC contribution by 70% compared to the value shown in Figure 4 for the baseline wood source(see Figure G.2 in Appendix G).

539 The inclusion of other GHGs changed neither the ranking nor the order of magnitude of the results.
540 However, their contribution to the total scenario impact differed among scenarios: while in “WWquality”
541 scenarios their contribution was relatively small (1-18% of net savings with TH=100 years), in “WWmix”
542 scenarios it accounted for up to 55% (scenario A1), mainly originating from a larger production of resin in
543 particleboard manufacturing for a larger fraction of the FU (100%) compared to scenario B1 (60%, since only
544 Q1+Q2 are recycled). The contribution from other GHGs decreased to about half when selecting TH=500
545 years, and appeared to decrease with increasing number of cascade steps in the system.

546

547 **3.5.5 Scenario analysis II – Testing assumptions on wood sourcing and rotation**

548 In the scenario where wood was imported from Brazilian short-plantation eucalyptus (rotation time: 10
549 years), the benefits on GWP100 increased compared to the baseline results (where wood had a rotation time
550 of 60y), especially for short-life cascades (i.e. A1-A2, B1-B2, D1, E1, F1) while these were less prominent for
551 longer-life cascades (i.e. A3-A4, B3-B4, C1) and finally decreased for scenarios C2-C4 (see also Figure G.1
552 in Appendix G). The reason behind is that shorter rotation plantations sequester carbon more rapidly than
553 temperate forests, with visible savings in the short term; however, the sequestration process terminates earlier
554 than for temperate forests, which continue sequestering carbon and provide larger overall savings if the wooden
555 product has a longer lifetime. The results for GWP500 were very similar to the baseline, underlining once
556 again that the GWP effects of biogenic emissions are minor over long timeframes. The additional analysis
557 performed by varying the biomass rotation period affected the results differently depending on the number of
558 cascades. In Figure 5 activity emissions were aggregated according to the use cycle at which they occurred to
559 highlight the role of each cascade step in the assessed scenarios. In general, the shorter the rotation of the
560 wood, the larger the net savings, because the new plantation can re-sequester emitted CO₂ faster. However,
561 while net savings for one-cascade systems decreased (and became burdens) with longer biomass rotations, net
562 savings for four-cascade systems remained stable irrespective of the biomass rotation.

563

the emission year (0, 10, 20, etc.), which conversely depends on the scenario under assessment (please refer to Figure 1, also for the definition of scenarios).

3.6 Implications and limitations of the study

The study focused on the evaluation of wood management systems from a climate change point of view only. However, recycling and incineration of wood waste may provide environmental savings/impacts in other impact categories. For example, the type of wood species may influence not only the GWP results, but also the biodiversity loss when fast-growing species are grown in place of longer rotation forests, as it is anticipated to occur in Brazil (see Thrän et al., 2017). Also, it is reported that incineration of particleboard releases higher levels of dioxins than solid wood (Gao et al., 2017), thereby potentially contributing to ecotoxicity and human toxicity impacts. This may be an incipient issue to tackle, as the more cascades to particleboard, the higher the concentration of resin in the recycled products, and potentially the higher the emissions when the product is incinerated at the end of its lifetime.

The consequential approach adopted in this study was based on two fundamental assumptions: the first is the long-term full-elasticity of supply, i.e. possible short-term effects due to fluctuations in prices of goods are not included (Weidema et al., 2009). This is especially the case for the iLUC factors used in our analysis (Schmidt et al., 2015; see section 2.6.4). Other studies applying economic modelling and considering short-term effects following price fluctuations argue that decreasing demand for virgin wood for material/energy production may decrease the price of forest timber which could potentially increase demand for other wood products, thus ultimately leaving the amount of forest-land unchanged (Plevin, 2016; Valin et al., 2015). Yet, this is unlikely for long-rotation forests where forest owners may decide to delay harvest if prices for timber are too low (Valin et al., 2015). This once again stresses the importance of both including iLUC effects and also finding a consensus on the methodology, although this goes far beyond the aim of this study. The second assumption regards the scale of the effects induced by the change in waste management, assumed here to be small (or negligible), i.e. not affecting the demand/market volume and the constraints on and production costs of the involved products/technologies (Weidema et al., 2009). We infer this assumption to be fair considering

594 that the current wood waste potential is minor compared to the wood supply market (Section 2.1); nevertheless,
595 complementary analyses could be performed in support of policy proposals.

596 The focus of the study was on Danish wood. Although the results from the scenario analysis on the
597 electricity system (Section 3.5.1) showed that country-specific background conditions may alter the magnitude
598 of the results, the environmentally sound conclusions on quality separation of wood waste at the collection
599 phase were robust even under different electricity mixes or considering iLUC, all in all suggesting that results
600 can be extended to other contexts. Management systems should be configured as to separately collect wood
601 waste qualities; this may, in addition to saving CO₂, reduce competition of uses between energy and material
602 sector. This would hold true especially in the case of countries with a growing economy, where resource
603 cascading is an arising subject. Finally, we would like to stress that a dynamic assessment of GWP impacts
604 (including biogenic CO₂) should be implemented whenever addressing long-time systems of bio-based
605 products, for example when quantifying the carbon footprint of wood-based buildings or in the use of bio-
606 plastics.

607

608 **4. Conclusions**

609 Drawing upon the comparison of 15 alternative wood waste cascading management scenarios, we showed –
610 from a climate change mitigation perspective – that quality-oriented recycling is preferred to maximizing the
611 quantity sent to recycling. Assuming a time horizon for the Global Warming Potential assessment of 100 years,
612 the benefits appeared larger when: i) increasing the number of cascade steps in the system, ii) the recycling
613 application had a long life-time, and iii) the recycling application substituted energy-intensive materials. The
614 latter, though, is only true when potential indirect land use change effects are excluded: their inclusion makes
615 substitution of land-intensive products preferable. Other sensitive assumptions were the choice of the time
616 horizon, the electricity system and the rotation time of the wood biomass: for longer time horizons, fossil fuel-
617 based electricity systems or short-rotation plantations (as marginal source of wood), shorter cascade systems
618 appeared preferable to longer cascade chains. Including other GHGs negligibly affected the results. Performing
619 a dynamic GHG emissions accounting is key to capturing the effects of biogenic CO₂ and storage, e.g. for

620 long-lifetime products; however, the results heavily depend on the time horizon chosen for the Global
621 Warming Potential assessment.

622

623 **Disclaimer**

624 The views expressed in the article are the sole responsibility of the authors and in no way represent the view
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626

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631

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634

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