**A Carbon Footprint Analysis of Railway Sleepers in the United Kingdom**

Georgios Rempelos a,\*, John Preston b, Simon Blainey c

*a \* Transportation Research Group. University of Southampton. Burgess Road SO16 7QF Southampton – United Kingdom. E-mail address:* *gr5g11@soton.ac.uk*

*b Transportation Research Group. University of Southampton. Burgess Road SO16 7QF Southampton – United Kingdom. E-mail address:* *J.M.Preston@soton.ac.uk*

*c Transportation Research Group. University of Southampton. Burgess Road SO16 7QF Southampton – United Kingdom. E-mail address:* *S.P.Blainey@soton.ac.uk*

**Abstract**

This paper provides an assessment of the lifecycle Greenhouse Gas (GHG) emissions associated with the four most common sleeper (railroad tie or cross-tie in North America) types present in the UK rail network. It estimates the embodied material, process and transport emissions linked with the lifecycle activities of construction, relay/renewal and end-of-life of these variants at low and high traffic tonnage. The analysis suggests that at low traffic loads, the softwood sleepers perform the best over the whole simulated-period. At high traffic loads, the concrete sleepers outperform all other variants in terms of lifecycle CO2e emissions, followed by hardwood, softwood and steel. Regardless of the scenario examined, the steel sleepers perform the worst due to the carbon intensive nature of their manufacturing process. This performance gap is amplified at high traffic loads, as their service life is excessively compromised. The analysis reveals that the end-of-life pathway of timber is a critical determinant of its footprint. Results suggest that the impact of disposing of these sleepers results in their footprint being magnified. Nevertheless, if a minimum of 50% follows the combustion pathway with subsequent heat recuperation, then a GHG reduction potential of between 11-18% of their footprint is feasible. From a whole-lifecycle cost lens, for higher tonnage routes, the choice of concrete sleepers results in considerable financial savings. If the infrastructure manager was to install sleepers with stiff under sleeper pads (USPs), it may achieve additional economic and GHG savings, with potential for increasing the latter using recycled carbon-neutral USPs.

***Keywords:*** *Life Cycle Assessment; Life cycle costing; Carbon footprint; Greenhouse gas emissions; Rail track; Railway sleepers.*

# **Introduction**

Life Cycle Assessment (LCA) studies often produce recommendations for strategies to reduce the cost of, or emissions from, the subject of the study in the future, and studies of railway track are no exception to this. One area which has proven particularly controversial is the question of which sleeper material minimises whole-life CO2 emissions, as wood tends to be viewed as being less carbon-intensive material than concrete. To date numerous studies explored this area (detailed in **Section 2**), however, their findings are often conflicting. This is mainly a result of differences within their wider scope, such as the range of environmental impacts considered, the functional units and system boundaries used for the analysis, the LCA methodology used, the technical specifications of the track forms, and geographical and temporal variations in the datasets used for analysis.

Against this background, the aim of this paper is to bridge the knowledge gaps and address the limitations identified from previous studies in order to provide a more definitive study of the carbon footprint and financial costs of different sleepers types. First, a modelling framework is developed for environmental and financial appraisal both at an asset and component level. Second, the applicability of the model is tested through an environmental and economic case study of the four most common sleeper types present in the UK rail network. Third, the results from this case study are compared against those of previous studies, examining the reasons behind potential variations, and quantifying the impact of different modelling choices on the results. While the conclusions from this work will be only directly applicable for the system boundaries of sleepers manufactured in the region of Great Britain, the modelling framework developed could be easily transferred to other geographic regions, and could therefore play a significant role in efforts to further reduce rail’s environmental impacts around the world. Finally, although the application is UK-based, the analytical approach draws on international evidence.

The remainder of this paper is structured as follows. **Section 2** presents a brief overview of previous studies in the field of railway infrastructure environmental modelling, with a particular focus on railway sleepers, emphasizing on the current research progress, and setting the scene for further research. **Section 3** describes the methodology adopted in this study, discussing the goal and scope, inventory, functional units, and model structure. **Section 4** details the results of the study, exploring their significance for the infrastructure managers. **Section 5** concludes the paper.

# **Study Background & Previous Research**

## **Research Background**

Considering the rail infrastructure itself, several concerns arise during manufacture, treatment using preservatives, and disposal of life-expired sleepers. The raw material and energy resources required for their manufacturing process are responsible for a relatively large share of these emissions (Crawford, 2009). Esveld (2001) suggests that the energy consumption during the production process of steel and concrete is significant and results in 10 to 200 times more CO2 dispersed into the atmosphere compared to that of hardwood sleepers. Aside of the emissions originating from the manufacturing process, transport and installation processes also generate emissions, also a sizeable portion of waste is generated mainly from the harvesting of timber (Esveld, 2001). Considering timber, the extent of its embodied externalities strongly depends on the mode(s) of transport used for the intermediate upstream-chain processes from cradle-to-the-factory-gate, which in turn are strongly dependent on the location of wood harvesting, processing, treatment and use (van den Dobbelsteen and Alberts, 2001). Moreover, following the timber harvesting process, carbon sequestration ceases and the process of decay onsets, with the subsequent effect being the progressive release of locked-up carbon back to the atmosphere. This is due to the fact that during tree growth, carbon is being absorbed from the atmosphere. This is being maintained locked-up in the timber’s structure, and the subsequent structure of its downstream by-products, up until the wood is either naturally decomposed or incinerated, resulting to the subsequent formation of biogenic CO2. According to Crawford (2009), the CO2e emissions during the service life of timber sleepers can be up to six times greater than these of concrete. Additionally, other concerns with timber sleepers are associated with their disposal when they become life-expired. This is due to the toxic substances used to extend their useful life, and prevent biological degradation. Despite being widely used, these do not easily decompose in the natural environment and are classified as volatile, posing subsequent risks to the environment and human health (Bilec et al., 2006; van den Dobbelsteen and Alberts, 2001). In the past, numerous studies have been conducted in an attempt to inform environmental decision-making, particularly with respect to which type of sleeper yields the best environmental performance. However, their findings are often conflicting, raising questions regarding their completeness and credibility.

For example, two studies targeting the appraisal of different sleepers used in the Japanese Shinkansen system concluded that timber sleepers have marginally lower emissions compared to concrete, with steel and synthetic sleepers performing the worst (Ueda et al., 2003, 1999). Another study conducted by Werner (2008), concluded that timber sleepers (e.g. beech and oak wood) result in considerably less Greenhouse Gas (GHG) emissions compared to the ones made of concrete and steel. Conversely, Milford and Allwood (2010) suggested that replacing all timber and steel sleepers on the UK rail network with concrete sleepers could yield 6-9% less CO2 emissions. Similarly, an assessment of the reinforced concrete and timber sleepers for the Australian network over a 100-year lifecycle, revealed that concrete sleepers (30-50 year service life) produce 70% less CO2 compared to the best timber sleeper scenario examined (30-year service life with 96% reuse of steel fastenings) (Crawford, 2009). Nevertheless, Owens (2007) showed that timber sleepers are more environmental friendly than concrete sleepers due to the reduction in carbon emissions. The difference between these results is mainly explained by the figures used to evaluate the amount of storage of carbon and carbon dioxide in timber sleepers. Similarly, Bolin and Smith (2013), concluded that creosote impregnated sleepers offer better environmental performance in terms of fossil fuel and water use, and lower environmental impacts compared to plastic composite (P/C) or concrete sleepers. With the exception of the eutrophication impact for P/C sleepers, all impacts were found to be favourable to timber sleepers. The methodology used consisted of a cradle-to-grave LCA underpinned with a gravity analysis to identify the most influential processes, and both uncertainty and sensitivity analysis of the results.

Considering the environmental impact of wooden sleepers made of beech compared to ones made of oak, Werner (2008) suggested that their impacts do not vary considerably, as the marginally lower oven dry weight of the oak sleepers and their lower creosote insertion rate is being compensated with a lower heating value per sleeper, which translates to a lower fuel substitution effect as a result of the thermal utilisation and heat recovery of wood.

## **Evidence from Previous Research**

The differences in conclusions (see Table 1) from previous studies arise due to a series of factors: (i) the mass of components is different between each study due to the different design characteristics being used in each country (Table 2), (ii) the service lives assumed vary between studies (Table 3), (iii) the carbon factors (CFs) display a variation between studies as they are drawn from different sources (Table 4), (iv) the methodological framework and the subsequent modelling choices largely differ between studies.

Considering the service lives selected, these can have a major influence to the results of such studies, as they determine the amount of interventions required over a studied period. For example, timber sleepers, were found to have a variation ranging from 10 to 45 years, suggesting that their CO2 emissions when normalised on a per sleeper year basis can vary by as much as 4.5 times. Similarly, the selection of CFs has a large impact to the outcomes of these studies, for example, *ceteris paribus*, the use of the hardwood sleeper CF by Crawford (2009) as opposed to the one by Milford and Allwood (2010) will result to 7 times more CO2 per hardwood sleeper installation. Adding to this, when considering the component mass and spacing differences between studies, these emissions variations may be set further apart.

**Table 1: Comparison of the environmental profile of different sleepers from selected studies (\*hardwood or softwood) – [✓✓✓ or ✓✓]: best performance, [✓]: better performance, [✘]: worse performance, [?]: not appraised.**

[Table 1 app. Here]

**Table 2: Comparison of the mass characteristics of different sleepers from selected studies – [✘]: not appraised.**

[Table 2 app. Here]

**Table 3: Service life of track components from selected publications (\*service life assumed for the whole track, \*\*upper/lower bound dependent on annual traffic load expressed in EMGTPA[[1]](#footnote-1) (10 low/60 high)).**

[Table 3 app. Here]

**Table 4: Component emission factors (EFs) from selected publications.**

[Table 4 app. Here]

# **Methodology**

## **Scope**

Given the above, this paper critically appraises and compares the lifecycle environmental impacts associated with the four most common sleeper types present in the UK rail network, namely, the G44 mono-block concrete, hardwood, softwood and the W560H steel sleepers. Broadly speaking, the most commonly quantifiable environmental impacts when it comes to the appraisal of railway sleepers are indicated by the GHG emissions and energy consumption. For this analysis, a spreadsheet model has been developed to evaluate the carbon footprint of different sleepers by adopting a streamlined lifecycle approach and the results are presented using a CO2e metric. This approach is in essence a slimmed down version of a complete LCA adopting all its core processes and excluding low contributing processes that would be otherwise included in a full LCA.

The subsequent evaluated emissions have been normalised over a km of single railway track (t.CO2e per km), to permit the summation of the environmental impacts of the different processes modelled within the examined products’ lifecycle. The key processes included in this analysis are broken down in the three core phases of the infrastructure’s life: construction, relay/renewal and end-of-life, with the associated emissions being based on the devised table of EFs (see Table 5). The methodology adopted in this work is based partially on the framework described by Ashby (2009) and the conceptual LCA framework guidelines designated by the ISO (2006a, 2006b) and BSI (2011).

## **Inventory analysis and Functional Unit**

The system boundary selected for the inventory analysis has been summarised in the Figure 1 below, the processes highlighted in red represent the upstream and downstream stages which are not scoped in the appraisal. The methodological choices of this work are purposed for screening and preliminary evaluation (or initial ‘hot-spotting’) of these design alternatives, this is mainly a result of data limitations. Finally, it should be pointed out that the in scope activities, CFs, geographical coverage boundaries and track specifications adopted are chosen to represent the UK region.

As already discussed, the environmental indicator metric on this appraisal is CO2 equivalent, with the associated emissions being normalised over a km of single railway track (stkm). The chosen functional unit includes only the railway sleeper component (either manufactured from concrete, steel, hardwood or softwood timber) excluding any auxiliary equipment such as fastenings, baseplates, fixings and so on. The primary reason behind the exclusion of these components is that in the UK routes there is a wide variety of baseplate and fastening combinations currently in use on various timber and concrete sleepers (approximately 17 types of fastening and 25 types of baseplates) (SUSTRAIL, 2012). According to SUSTRAIL (2012), a relatively small number of combinations of fastenings and baseplates are used in primary passenger routes in the UK as these are fairly standardised.

**Figure 1: Simplified flow diagram and associated system boundary for the streamlined LCA carbon footprint model.**

[Figure 1 app. Here]

Considering the lifecycle adopted for this study, a 60-year appraisal period has been chosen in line with WebTAG recommendations (DfT, 2018), with the associated in scope processes being the raw material extraction, manufacturing of sleeper components, transport on site via rail freight, infrastructure use including the dismantling of obsolete components and the subsequent relay/renewal of new ones and finally, the end-of-life phase of the life-expired sleepers depending on the material present following the appropriate downstream pathway.

## **Model construction**

### **Embodied Emissions**

The embodied carbon dioxide (EC) impact of the construction phase for each examined sleeper variant was estimated based on the material and carbon intensity values and the respective service life estimates based on the simulated load scenarios (see Table 5). The service life estimates in Table 5 are given in ranges to reflect the impact of different traffic loads on the sleeper renewal regimes.

**Table 5: Basic infrastructural component characteristics (\*upper/lower bound dependent on annual traffic load expressed in EMGTPA (10 low/60 high)), Sources: (Milford and Allwood, 2010; RSSB, 2018; SERCO, 2007).**

[Table 5 app. Here]

The service life condition information are being drawn from the Network Rail Vehicle and Track Interaction Strategic Model (VTISM) (SERCO, 2014, 2007) and is based on a mix of age and accumulated tonnage data recordings from UK routes. The material breakdown by mass is primarily based on the work done by Milford and Allwood (2010), with data drawn from their study being cross-correlated with other academic publications and manufacturer brochures, as well as data present within the RSSB’s Rail Carbon Tool (RSSB, 2018). The embodied CO2e emissions, $EC\_{j}^{c-g}$ ‘cradle-to-gate’ associated with the manufacturing of each sleeper variant were estimated by multiplying each material mass with the associated CF (equation (1)) quoted in RSSB’s Rail Carbon Tool. These CFs are based on the Bath Inventory of Carbon and Energy (Hammond and Jones, 2011).

|  |  |
| --- | --- |
| $EC\_{j}^{c-g}={LT}/{s}×\sum\_{i=1}^{n}M\_{i}×C\_{i}×CF\_{i}$ | (**1**) |

Where, $EC\_{j}^{c-g}$is the aggregate embodied ‘cradle-to-gate’ (*c-g*) CO2e emissions for a track length *LT* laid with a type *j* sleeper in kgCO2e/functional unit selected; *i* is the material index; *n* is the total number of materials used in the construction of the railway sleeper; *Mi*is the mass of material *i* for each sleeper component given in kg; *Ci* is the % composition of each material to the total mass of the component; *CFi* is the EC factor for each material *i* given in kgCO2e/kg; *LT* is the chosen length of track given in metres; *s* is the UK sleeper spacing between centres given in metres.

In order to normalize the evaluated EC emissions per stkm, the standard sleeper spacing used in the UK rail network (600 to 700 mm between centres) has been used to calculate the number of sleepers per stkm (1,538 sleepers) and multiply them with the appropriate quantity of EC (in kgCO2e/sleeper) calculated for each sleeper type.

### **Component Processing**

Additionally to the material manufacturing processes, components processing during track installation procedures including tasks such as rail cutting, welding, ballast scattering among others will also produce emissions. However, specific estimates of such figures are scarce, with the only set of information on component processing being the ones provided by Lee et al. (2008). Milford and Allwood (2010) posit that the emissions from the processing of track components typically contribute less than 8% of the aggregate footprint. In this study, the labour and plant emissions arising during construction have been estimated based on the machinery specifications of a Matisa TCM60 construction train with a sleeper laying productivity of 16 sleepers/minute and an engine size of 285kW (MATISA Matériel Industriel S.A., 2018a). Considering the labour and plant emissions arising during relay/renewal, these were estimated based on the machinery specifications of a P95 Matisa renewal/track-laying train (engine size 400kW), which is capable of installing concrete, steel and timber sleepers with an average working speed productivity of 1.1 km/hr (MATISA Matériel Industriel S.A., 2018b). The calculation of these emissions has been made indirectly using the equation (2) shown below.

|  |  |
| --- | --- |
| $$EC\_{L\\_P}^{s}=\sum\_{i,j=1}^{n}{LT}/{M\_{P\_{i}}}×{PR\_{ij}}/{100}×{PL\_{ij}}/{100}×CF\_{ij}$$ | (**2**) |

Where, $EC\_{L\\_P}^{s}$ is the CO2e emissions arising from the use of machinery and equipment for the construction, maintenance and relay/renewal on site in tonnes of CO2e; *i* is the machinery index; *j = 1, 2, 3,…,n* corresponding to a rated power of *100kW, 300kW, 400kW,…,n*; *n* is the total number of equipment used in the installation/renewal of the railway sleepers; *LT* is the length of track to be processed given in km; *MPi* is the track-laying train productivity given in km/hour; *PRij* is the % runtime of the engine *j*; *PLij* is the % load of the engine depending on the pre-defined engine size *j*; *CFij* is the CF for machine *i* with engine size *j* given in kgCO2e/hour of combustion (e.g. 237 kgCO2e/hour and 316 kgCO2e/hour for a 300kW and 400kW diesel engine respectively).

### **Component Transport Emissions**

Emissions will also be produced from the transportation of components and materials associated with the construction, maintenance and/or renewal processes. Such emissions are highly ‘site-specific’, as they are dependent on the distance from the material source to the construction site, but also to the transport mode used. Milford and Allwood (2010) assumed an arbitrary selected transport distance of 200 km for all components, assumed to arise via rail freight. On this basis, they estimated emissions using Department for Environment, Food and Rural Affairs (DEFRA) recommended EFs. Milford and Allwood (2010) highlighted that even with an arbitrary selected distance of this magnitude, transport from the factory gate to site typically accounts for less than 3% of the aggregate footprint. The CFs drawn from the RSSB’s Rail Carbon Tool are quoted as ‘cradle-to-gate’ rather than ‘cradle-to-site’ boundaries, and as this analysis does not consider any specific location, an arbitrary selected transport distance of 50 km has been chosen for all sleeper components, with the subsequent direct emissions assumed to arise via rail freight. Consequently, in order to evaluate the direct embodied emissions arising from the transportation of the sleeper components, the Department for Business, Energy and Industrial Strategy (DBEIS) rail freight ‘all scope’ EF (DBEIS, 2018) quoted in kgCO2e/tonne km has been used (using equation (3)).

|  |  |
| --- | --- |
| $$EC\_{T}=\sum\_{j}^{}\sum\_{i}^{}M\_{ji}×(T\_{ji}×CF\_{i})/1000$$ | (**3**) |

Where, *ECT* is the direct carbon emissions arising from the transportation of materials, equipment and waste expressed in tonnes of CO2e; *Mji*is the amount of building material, component, equipment or waste *j* (in tonnes) to be transported by vehicle *i*; *Tji* is the total transport distance for item *j* transported via vehicle *i* (in km); *CFi* is the EF to transport an item using vehicle *i* (in kgCO2e/tonne km).

### **Carbon Emissions due to Temporary Works**

Aside of the embodied emissions associated with the permanent building materials used on the railway infrastructure construction, maintenance and renewal operations, the rail industry consumes a sizable quantity of temporary materials to support different construction activities. This is particularly the case for the construction activities carried out for the installation of ballast-less track designs. For example, temporary materials such as propping elements and formworks are required to support the concrete slab during the curing process until it reached the desired strength and being able to support its weight. These materials are often reusable for a certain time period up until they turn into waste and they need replacement. Consequently, the EC emissions originating from these materials follows a gradual depreciation over time and use. The exact same concept applies to the equipment used for supporting the construction, renewal, inspection and maintenance operations, where the initial embodied emissions of the equipment depreciates over time as it approaches its life-expiration time. Akbarnezhad et al. (2014) suggests that when the carbon footprint of construction is being evaluated, the depreciation of the EC of the equipment and temporary materials should be taken into account. Akbarnezhad and Xiao (2017) suggested the use of the following formulation in order to determine the amount of CO2 emissions attributed to the gradual reduction in the remaining service life of the construction equipment and temporary materials:

|  |  |
| --- | --- |
| $$C\_{C}^{TM/E}=\sum\_{i}^{}\left(EC\_{i}^{M/E}- EC\_{i}^{S}\right)d\_{i}^{u}/D\_{i}^{S}$$ | (**4**) |

Where, $C\_{C}^{TM/E}$is the construction carbon emissions attributed to the depreciation in the EC of equipment or temporary materials (in tonnes CO2e); $EC\_{i}^{M/E}$is the EC of the temporary materials or equipment *i* as reported by the manufacturing company (in tonnes of CO2e); $EC\_{i}^{S}$ is the salvage EC of the temporary materials or equipment *i* at the end of its serviceable life (in tonnes of CO2e); $d\_{i}^{u}$ is the operating duration of the temporary materials or equipment *i* on the project given in hours, days or years; $D\_{i}^{S}$ is the service life of the temporary materials or equipment *i* given in hours, days or years.

Akbarnezhad and Xiao (2017) advised that the salvage EC of temporary materials or equipment *i* at the end of their serviceable life can be evaluated by investigating the final usability and/or fitness for use of the remaining components. They suggested that in case all the elements are to be recycled, the following expression should be used:

|  |  |
| --- | --- |
| $$EC\_{ij}^{S}=\sum\_{j}^{}\left(EC\_{ij}^{R}- C\_{ij}^{RP}\right)$$ | (**5**) |

Where, $EC\_{ij}^{R}$is the EC of the recycled product obtained following the recycling of the component *j* of equipment or temporary material *i*; $C\_{ij}^{RP}$is the CO2 emissions arising from the processes involved on the recycling process of component *j* belonging to the equipment or temporary material *i*.

Whilst there was the intention of including the embodied emissions from the manufacturing of the equipment used for the construction, maintenance and relay of the railway infrastructure, it has been decided for these emissions to be scoped-out of the analysis, as manufacturer quotes for this equipment could not be traced in the academic literature.

### **Track Maintenance and Renewal**

Most studies considering railway infrastructure omit emissions resulting from track maintenance, and even in studies which do estimate the resulting emissions from such activities, there is often little detail of what is considered as forming part of such maintenance. Milford and Allwood (2010) estimated maintenance emissions from the three main maintenance activities carried out on the ballasted track (e.g. tamping, stoneblowing, rail grinding etc.). The maintenance frequencies adopted in their study were based on estimates provided by Network Rail, with the emissions being modelled indirectly based on the fuel consumption of the vehicles involved in these operations. Similarly, Chester (2008) estimated maintenance emissions (from material replacement, inspection(s), rail grinding) based on the equipment used and the labour productivity, as well as utilizing rail maintenance EFs from the SimaPro software. More recently, Krezo et al. (2018) conducted a field-based study to evaluate the CO2 emissions from the three main railway resurfacing activities (e.g. tamping, ballast regulator, and stabilizer). Based on their results, they found vast differences (-55% to +26%) on fuel consumption and subsequent CO2 emissions compared to earlier studies by Milford and Allwood (2010) and Kiani et al. (2008). They suggested that these arise due to differences in (i) costruction speeds (e.g. accounting for real-time delays) and also in (ii) diesel fuel EFs between Australia and the UK. Krezo et al. (2016) appraised that the CO2 emissions from renewal/maintenance interventions are dominated by the EC (90-98%) of the materials used, with the remaining 2-10% being attributed to the machinery utilized (Krezo et al., 2016).

In this study, the focus is placed on the appraisal of railway sleepers excluding any auxiliary equipment as well as other components such as rails and ballast that require frequent maintenance over their service life which will also produce a sizeable portion of emissions. Consequently, the only recurring emissions arising from the maintenance/renewal of the infrastructure were assumed to arise from the renewal of the sleeper components based on the service life threshold values shown in table 5. Consequently, these life expectancy values define the amount of renewal operations required for each sleeper component. There is a large uncertainty surrounding these estimates as in reality the renewal of these components is heavily dependent on a wide range of factors. This means that in absence of recorded data the prediction of their service life is highly subjective and virtually impossible. As it can be inferred from table 3, the service lives assumed for railway sleepers in different studies are highly variable with for example certain publications assuming a service life of as low as 20 years (Kiani et al., 2008) for concrete sleepers, while others suggesting a service life in excess of 50 years (Chester, 2008; Crawford, 2009; Ueda et al., 2003, 1999). SUSTRAIL (2012) suggests that concrete sleepers have an average service life in excess of 50 years, however, in the UK they are usually being replaced at the same time as rail at approximately 20 to 25 years, utilising the sleepers dismantled from the primary routes in secondary lines (SUSTRAIL, 2012). Suggesting that an average of 31 to 52% of the sleepers on UK routes do not exceed 50% of their design life at renewal. This industry practice may contribute to additional financial, environmental and operational performance burden imposed to the responsible infrastructure authority.

In summary, the renewal emissions have been calculated using equation (6) shown below. The $EC\_{L\\_P}^{s}$component of the equation which refers to the CO2e emissions arising from the use of the machinery for the renewal of the infrastructure, includes the emissions from the sleeper dismantling, as these have been assigned to the renewal phase of the LCA instead of the end-of-life. This has been decided based on the capability of Matisa P95 track laying train to perform parallel dismantling and complete track bed renewal operation (MATISA Matériel Industriel S.A., 2018b).

|  |  |
| --- | --- |
| $$EC\_{R}^{s}=EC\_{j}^{c-g}+EC\_{T}+ EC\_{L\\_P}^{s}$$ | (**6**) |

Where, $EC\_{j}^{c-g}$is the aggregate embodied ‘cradle-to-gate’ (*c-g*) CO2e emissions for a track length *LT* laid with a type *j* sleeper in tonnes of CO2e per functional unit selected; *ECT* is the direct carbon emissions arising from the transport of materials, waste and equipment expressed in tonnes of CO2e; $EC\_{L\\_P}^{s}$ is the CO2e emissions arising from the use of machinery and equipment for the construction, maintenance and relay/renewal on site in tonnes of CO2e.

### **End-of-life**

A factor which is often overlooked in rail-related LCA studies, is the disposal of infrastructure when it becomes life-expired. Milford and Allwood (2010) posit that the emissions arising from the ‘end-of-life’ disposal phase of the infrastructure are predominantly related to transport. Nevertheless, the associated processes of dismantling, incineration and landfilling will also generate a sizeable portion of emissions. However, most of the existing rail-related studies largely omit the ‘end-of-life’ stage due to scarcity of process-specific information related to the exact pathway of infrastructural components. In this study, three end-of-life options for the sleeper components have been selected: (i) incineration, (ii) landfilling, (iii) recycling and subsequent re-use to lower grade applications. Considering the end-of-life characteristics for concrete sleepers, it has been assumed that 50% of the material is going to be recycled, while the remaining material is going to be landfilled. On the other hand, steel sleepers are assumed to be 100% recycled. The model excluded the emissions associated from the process of recycling as these lie outside the system boundary. These emissions would be included in the construction/manufacturing phase of a downstream recycled product. Consequently, the carbon dioxide equivalent factors adopted to represent the recycling process for concrete and steel only consider the transport to an energy recovery or materials reclamation facility. This approach is in line with the Greenhouse Gas Protocol (2004). The factors have been sourced from the latest greenhouse reporting database by DBEIS (2018) quoted in kgCO2e per tonne of concrete or steel landfilled or recycled respectively.

|  |  |
| --- | --- |
| $$E(CH\_{4})=MCF×DOC×DOCF×F×16/12×(1-OX)^{6}$$ | (**7**) |

Where, $E(CH\_{4})$is the methane EF given in tonnes of CH4 per tonne of waste wood landfilled; $MCF$is the methane correction factor which is recommended by IPCC (1996) to have a value of 1.0 for managed landfills; *DOC* is the degradable organic carbon which is recommended by IPCC (1996) to have a value of 30% for wood waste; *DOCF* is the fraction of *DOC* dissimilated (i.e. converted to gas), recommended by IPCC (1996) to have a value of 0.5; *F* is the fraction of CH4 in landfill gas – default value of 0.5 (IPCC, 1996); 16/12 is the conversion factor from carbon to methane (IPCC, 1996); *OX* is the oxidation rate which essentially accounts for the CH4 that is oxidised in the uppermost layers of the waste mass and in cover material, where there is presence of oxygen - IPCC (1996) suggests a value of 0 as they note that there is currently no internationally accepted factor that can be applied to take into account the CH4 oxidation – in this work a 10% oxidation rate was assumed based on the work done by Primary Power International (2004).

Following the estimation of the methane EF from the landfill of wood waste, 85% of these landfill gas emissions were assumed to be captured and burnt (substituting natural gas) as recommended by DEFRA (2006). This is proven to be a conservative estimate, as according to Milford and Allwood (2010) if the landfill gas was utilised as a substitute for grid electricity, the subsequent CO2e emissions from the landfilling process of timber would be a net carbon sink. Finally, based on the aforementioned relationships, in order to derive the avoided CH4 emissions from the landfill gas capture process equation (8) has been used.

|  |  |
| --- | --- |
| $$E\left(CO\_{2}e\right)=E(CH\_{4})×W×GWP×C$$ | (**8**) |

Where, $E\left(CO\_{2}eq\right)$is the avoided CO2e emissions from the capture of the landfill gas given in tonnes of CO2e/tonne of wood; $E(CH\_{4})$is the methane EF given in tonnes of CH4 per tonne of waste wood landfilled; $W$is the quantity of wood to be landfilled given in tonnes; $GWP$is the global warming potential of CH4 in CO2e which is equal to 25 (Pachauri et al., 2014).

The second examined end-of-life pathway option for timber sleepers was incineration with energy recovery; however, due to the presence of carcinogenic coating attributed to the creosote impregnation process of timber sleepers, the base case scenarios excluded recycling or incineration as this practice is currently restricted in the European Union. Nevertheless, a ‘what-if-scenario’ has been tested on the worst timber sleeper scenario found in terms of environmental performance, in order to examine the extent of incineration required in order to offset the environmental dis-benefits of timber sleepers when compared with other sleeper alternatives. An issue with the evaluation of these emissions and also the subsequent benefits from the energy recovery during incineration was that the initial locked-up carbon resulted from the carbon sequestration during hardwood and softwood tree growth was unknown. Moreover, it was unknown if this parameter was included in the embodied CFs quoted by RSSB (2018). In light of this, Milford and Allwood (2010) assumed no energy recovery from timber incineration to account for the carbon absorbed during tree growth. Nevertheless, in this analysis it is assumed that the energy recovery is feasible, with the subsequent calculation being made by assuming a heat content from the combustion of industrial wood of 3.306 kWh/kg of wood (Carbon Trust, 2016). The subsequent heat energy recovered was assumed to be once again used as a substitute for natural gas (0.183997 kgCO2e/kWh) (Carbon Trust, 2016).

### **Non-Energy Related Emissions**

#### **Locked-up Carbon**

During tree growth, carbon is being absorbed from the atmosphere. This carbon is being maintained ‘locked-up’ in the timber’s structure and the subsequent structure of its downstream by-products up until the wood is either naturally decomposed or incinerated, resulting to the subsequent formation of CO2 (Crawford, 2009). Crawford (2009) posits that this amount of carbon can be deducted from the carbon in the embodied GHG releases related with the fabrication of wooden sleepers. However, it was unknown if the locked-up carbon was included in the derived CFs for timber sourced from RSSB (2018). Therefore, it was decided to exclude this parameter from the model.

#### **Roundwood Conversion Emissions**

Aside of ‘locked-up’ carbon in wood, other non-energy related emissions include the emissions associated with the roundwood conversion of wood. “When forests are harvested, underbrush is disrupted, bark leaves and branches are stripped, and off-cuts and sawdust result” (Crawford, 2009). Crawford (2009) suggests that it is a common practice to either chip the roundwood conversion waste for use in wood products, or burn it either on-site or kilns that are used for the process of drying timber or for other process heat, or alternatively leave it where it is. In his study, he assumed that the CO2 originating from the roundwood conversion waste enters the atmosphere as GHG by assuming a factor of 40% as the share of biomass that is converted into hardwood sleepers, with the remaining 60% representing the timber waste to be treated as roundwood GHG emissions. However, as pointed out by Snowdon et al. (2000), this factor can display a large variation ranging from 17 to 80% depending on the available market, type of forest and method of harvesting adopted. Aside from this, it can be argued that the locked-up carbon from the reclaimed waste for other products such as wood chips should be credited to them. Adding to this, if the purpose of forest harvesting is not only for the production of timber sleeper, which is usually the case, and the decision to harvest trees is linked with partially producing other products, then a subsequent portion of these aggregate emissions should be credited to them. These emissions have not been directly included in this study, however, there are indications suggesting that the embodied CFs adopted for timber include biomass process energy which is most likely originating from roundwood conversion.

#### **Timber Decay Emissions**

Following the timber harvesting process, carbon sequestration ceases and the process of timber decay onsets, with subsequent effect being the progressive release of the locked-up carbon back to the atmosphere. Crawford (2009) calculated the emissions from the eventual decay of new hardwood sleepers by assuming a carbon mass content for hardwood of 50%. Hence based on the derived CF for timber of 3.67 kgCO2/kg of carbon and a weight of 80.2 kg per timber tie, the eventual decay CO2 emissions were estimated as 147 kg CO2 per tie, with the complete decay assumed to be occurring after an 100-year period at a uniform rate over the sleepers’ entire life (Crawford, 2009).

#### **Carbonation of Concrete**

It is well documented that concrete has the ability of chemically reacting with airborne CO2 (see Salas et al. (2016)). However, the carbonation by concrete during its primary or secondary life is often omitted in LCA studies (Collins, 2010). Collins (2010) suggests that the carbon capture by built concrete during its primary life is more or less negligible. However, he stated that the CO2 capture during the concrete’s secondary life is significantly greater as the crushed product has a greater exposure to CO2, as the greater surface area relative to volume results in greater levels of carbonation (Collins, 2010). This amount during the secondary life of the recycled concrete aggregate can be up to 41% of the carbon dioxide released during the production of 100% Portland cement binder (Collins, 2010). Collins (2010) posits that if carbonation is omitted, emissions figures can be overestimated by as much as 13 to 48%. However, he suggested that this range of overestimation will depend on the type of cement binder, as well as the destined application of the recycled downstream concrete product during its secondary life (Collins, 2010).

In this model only the CO2 uptake during the concrete sleeper’s primary life has been considered. This decision has been taken due to the fact that the recycling of concrete sleepers for the same application has been scoped out of the analysis as it is unknown if this is a common practice in the UK rail industry. Nevertheless, if waste concrete is being crushed in order to be used as a substitute for ballast material, it will contribute not only to avoiding the production of ballast from stone materials, but also it will reduce the amount of landfill waste, save resources and reduce the amount of CO2 emitted from the production of virgin ballast (Stripple, 2013).

Aside of these benefits, crushed concrete materials can enable a rapid CO2 uptake (in less than one year), while thicker concrete layers will carbonate in a slower pace (Stripple, 2013). Nonetheless, in order to realize the benefits from the CO2 uptake of ballast made of recycled concrete, the waste material should be crushed and sieved accordingly, so its void ratio is considerable to enable air circulation (Stripple, 2013). The CO2 uptake from the potential recycling of concrete for lower grade application such as for railway ballast was not included in the model, as the ballast component was omitted from the selected functional unit. However, it is worth pointing out that if the crushed concrete achieves the desired specifications to be used as a substitute for railway ballast, then the environmental profile of the concrete railway sleeper track is likely to be significantly improved.

Concrete sleepers are common products that are expected to carbonate during their service life. For the model calculations, the G44 mono-block sleeper has been considered with a weight of 309 kg per unit excluding fastenings, a length of 2.5 m, width of 0.285 m, depth of 0.2 m on the corners and a mid-span depth of 0.175 m. Using these dimensions the surface area distribution exposed to (topside, sides and ends = 1.8265 m2) or sheltered from rain (bottom = 0.7125 m2) has been estimated. The main surfaces have a good potential exposure to CO2 even with 80% of the sleeper’s surface being covered with railway ballast (Stripple, 2013). Hence, although the high quality dense concrete utilised for railway sleepers hinders the rate of CO2 uptake of concrete during its use phase, the potential magnitude of uptake is high due to the high cement content in the concrete (Stripple, 2013). In summary, in order to calculate the annual CO2 uptake for concrete sleepers, the derived estimates by Nilsson (2011) for exposed concrete structures to rain and sheltered structures from rain have been adopted and used to derive the relationships shown in Figure 2.

These relationships have been used to estimate the carbonation (in kgCO2/m2 of concrete) of the exposed and sheltered portion of the concrete sleeper per annum which were multiplied by the exposed or sheltered area of the sleeper to determine the annual amount of carbonation (in kgCO2/sleeper).

**Figure 2: Derived relationships for the carbonation of concrete structures exposed to and sheltered from rain based on data estimates made by Nilsson (2011).**

[Figure 2 app. Here]

# **Results and Discussion**

## **Simulated Scenarios**

In total eleven programmed scenarios have been tested. These scenarios have been programmed using Visual Basic (VBA) excel spreadsheet models in order to enable variation in the modelling parameters with ease. The simulated scenarios differentiate in terms of: (i) track design (e.g. concrete, steel, hardwood, and softwood sleepers), (ii) traffic load conditions (either 10 or 60 EMGTPA), (iii) end-of-life pathway (e.g. assuming that timber sleepers are either 100% landfilled or partially landfilled and incinerated with energy recovery), (iv) installation of novel interventions (e.g. concrete sleepers fitted with stiff under sleeper pads (USPs)). These scenarios are detailed in table 6 below.

**Table 6: Overview of simulated scenarios (L: landfilling; R: recycling; L|R: partial landfilling and recycling; L|E: partial landfilling and incineration with energy recovery).**

[Table 6 app. Here]

## **LCA Modelling Results**

### **Carbon Footprint Results: *Cradle-to-Site***

The CO2e emissions associated with the construction phase (e.g. cradle-to-site) of each sleeper type are common across all the simulated scenarios. These emissions are split in three primary areas: (i) EC emissions associated with the materials used, (ii) direct emissions associated with the transportation of materials from the factory gate to the construction site, and (iii) the emissions originating from the labour and plant (e.g. construction vehicles and machinery) during the track construction process. It has been found that the share of transport emissions to the aggregate footprint of construction ranges approximately between 0.1-0.8% (c. 0.19 to 1.0 t.CO2e/stkm in absolute terms) depending on the weight of the sleepers to be transported (as the mode of transport and the distance from the factory gate were capped across all scenarios and sleeper types). This translates to sleepers with higher unit weight (e.g. concrete) having a higher share of CO2e emissions from transport compared to lighter alternatives (e.g. softwood timber). Similarly, the emissions originating from the construction equipment had a miniscule share of between 0.2-0.5% (c. 0.4 t.CO2e/stkm in absolute terms) of the construction footprint. The greatest share on the CO2 footprint of construction has been found to be due to the EC associated with the production of materials (see figure 3), ranging between 54.6 to 167.3 t.CO2e/stkm depending on the type of sleeper considered.

**Figure 3: % contribution of each LCA phase (gate-to-site boundaries) to the CO2 footprint of construction.**

[Figure 3 app. Here]

From figure 4 it can be inferred that the steel sleepers have the greatest environmental impact during construction, which is approximately 36.6% greater than the next sleeper alternative (e.g. concrete sleepers). This should not come as a surprise, considering the sizeable mass of steel required and the highly carbon intensive process of manufacturing this component. At the other end of the environmental spectrum, the softwood sleepers made of French Maritime Pine, have three times smaller carbon footprint during construction compared to that of steel sleepers. Comparing the sleepers made of hardwood and the ones made of softwood, the former type has 60% more EC compared to the latter. The difference between the two can be explained partly due to their 9.8 kg mass difference and the 47.5% greater carbon intensity of the hardwood sleeper as expressed by its embedded CF.

**Figure 4: CO2 impact from the construction (materials, labour and plant, transport) of a stkm for each sleeper type (in tonnes of CO2e emissions per stkm).**

[Figure 4 app. Here]

### **Carbon Footprint Results: *Cradle-to-Grave***

#### **Scenario 1: 10 EMGTPA**

The first simulated scenario assumed a low traffic tonnage of 10 EMGTPA. This scenario is broadly reflecting the tonnage of an inter-urban primary route. Under this scenario the softwood sleepers performed the best out of all the alternatives over a 60-year period. Hardwood sleepers displayed the second best environmental profile, however, they emitted 36.8% more CO2e emissions compared to the softwood sleepers. The concrete sleepers exhibited marginally worse performance by emitting 5.5% more carbon compared to the hardwood sleepers. Notably, steel sleepers displayed by far the worst footprint by emitting 29% to 57% more CO2e emissions compared to the other variants. The aggregate carbon footprint of each tested sleeper alternative is being displayed in figure 5.

Looking at the breakdown of the overall carbon footprint by LCA phase (see figure 6), it can be inferred that at low traffic loads the impact of construction is the greatest (e.g. accounting for between 38.9-51.4% of the overall footprint followed by the impact of track renewal which is marginally lower, as all the variants undergo only one major renewal during the whole simulated period. The impact of end-of-life phase for both the concrete and the steel sleepers is negligible ranging from 0.03 to 0.23% of the overall footprint. This is a direct implication from the modelling choice of recycling 100% of the steel sleepers and 50% of the material from the concrete sleepers. Conversely, the impact of landfilling of the timber sleepers’ results to a sizeable portion of emissions due to the releases of methane. This figure is ranging between 16.79 to 22.86% of the overall footprint for the hardwood and softwood sleepers respectively. As it was expected the impact of carbonation of concrete during its primary life is miniscule, accounting for approximately 3% of the concrete’s aggregate footprint. Nevertheless, the carbonation of the recycled concrete product during its secondary life is expected to be significant, however, this calculation has not been incorporated into the model.

#### **Scenario 2: 60 EMGTPA**

The second simulated scenario assumed a high traffic tonnage of 60 EMGTPA. This scenario is generally reflecting a high tonnage urban primary route with high service frequency in terms of commuter flows tied with a significant presence of freight traffic. At high traffic tonnage, following a 60-year appraisal period, the concrete sleepers perform the best by emitting 26-46% less CO2e compared to the other sleeper variants. Once again hardwood sleepers displayed the second best environmental profile by emitting 16% and 27% less CO2e over a 60-year period compared to the softwood and steel sleepers respectively (see figure 5). Whereas, the steel sleepers performed the worst due to the carbon intensive nature of their manufacturing process (e.g. 13-46% more CO2e emissions). Another important consideration that it is worth pointing out is the impact of the selected service lives. For example, when the impact of the service life of the concrete sleepers is taken into account, and their subsequent emissions originating from construction are being normalized on a per km year basis, then it can be seen that these sleepers exhibit the best environmental profile due to their prolonged service life under high traffic tonnage. This highlights that the significantly increased service life of the concrete sleepers under this scenario, results to their EC share being spread out over the examined lifecycle period, as the subsequent amount of renewal operations required will be much smaller than, for example, that of softwood sleepers (e.g. 10-year service life at 60 EMGTPA, translating to five major renewals). Conversely, the significantly reduced service life of the steel sleepers under 60 EMGTPA can explain largely the substantial size of their footprint. The renewal figures from this scenario suggest that the steel sleeper renewal operations emitted 2, 1.8 and 1.5 times more CO2e emissions compared to these of concrete, hardwood and softwood (see figure 7). Having said this, the literature confirms that this type of sleeper should only be utilised for more lightly trafficked lines and are reported to be suitable only for speeds at or below 160 km/hr (Manalo et al., 2010).

Considering the impact of the end-of-life phase for both the concrete and steel sleepers, it is deemed negligible, accounting for less than 1% of their total footprint. This is marginally greater for concrete, as a proportion by mass of the secondary product is being treated as waste and subsequently, landfilled. In contrast, the impact of disposing the timber sleepers when they become life-expired, results in their footprint being magnified. The figures suggest that the end-of-life phase for the hardwood and softwood sleepers has a share of 23.2% and 33.7% of their aggregate footprint. This is due to the sizeable releases of CH4 from landfilling wood, which has 25 times higher GWP when compared with CO2. The impact from the disposal of timber has been found to be 100 to 180 times higher than that of concrete recycling. When these figures are being compared with the emissions from steel recycling, timber sleepers appear to emit a staggering amount of 325 to 560 times that of steel sleepers. Whilst softwood sleepers have lower mass than the hardwood ones, which would imply a smaller footprint from their disposal phase, their service life difference of 6 years meant that hardwood sleepers have to undergo less frequent interventions (e.g. three complete renewals as opposed to the five relay operations required for softwood sleepers), translating to less waste material ending up on landfill. Once again the carbon uptake of concrete sleepers during their primary life was miniscule (e.g. c. 2.3% of the total footprint).

**Figure 5: Aggregate carbon footprint of each sleeper design for a 60-year appraisal period for a low and high traffic tonnage scenario.**

[Figure 5 app. Here]

**Figure 6: Carbon footprint for each sleeper type broken down by LCA phase for a 60-year appraisal period (10 EMGTPA).**

[Figure 6 app. Here]

**Figure 7: Carbon footprint for each sleeper type broken down by LCA phase for a 60-year appraisal period (60 EMGTPA).**

[Figure 7 app. Here]

Taking a lifecycle perspective at this stage, it is important to point out that irrespective of the scenario considered, typically concrete sleepers will last longer, with the high tonnage scenario signifying their advantage in terms of service life. At high traffic tonnage, following 31 years of traffic operation, the concrete sleepers achieve a break-even point resulting in a positive environmental profile compared to the other sleeper alternatives. Their dominance can be better understood by looking at the magnitude of emissions associated with their use, which is comparable with the best steel sleeper scenario at low traffic tonnage (e.g. emitting just about 8.4% more CO2e compared to the best steel sleeper option). Regardless of the scenario considered the steel sleepers consistently perform the worst. Even at low traffic loads their prolonged service life of 40 years is proven inadequate to offset their sizeable footprint originating from their manufacturing. This is due to the fact that metal products require considerably more processing when compared with concrete or wood products. Adding to this, the production of the appropriate end-products from base steel is also highly energy-demanding, resulting to a sizeable portion of CO2 emissions. Timber sleepers appear to be a more desirable option from a GHG emissions reduction perspective at least for lightly trafficked routes. Softwood sleepers have the best environmental profile at low traffic loads, while hardwood sleepers display marginally better performance when compared with the concrete ones (e.g. 5.5% less CO2e).

#### **Scenario 3: What if scenario? – Timber Incineration**

The third scenario was an attempt to examine the amount of incineration with energy recovery required to yield a better environmental profile for both timber sleeper types compared to the concrete ones. Following this scenario, it has been found that a minimum of 50% of the timber sleepers have to be incinerated with energy recovery in order to yield a positive performance. The GHG emissions reduction potential due to the energy recovery process represents approximately 11% (hardwood) and 18% (softwood) of their aggregate footprint (under 60 EMGTPA) (see figure 8). This end-of-life strategy results to a 7.3% and 12.9% smaller footprint for these variants, when compared to the concrete sleepers (see figure 8). Nevertheless, in reality incineration and/or combustion practices are not suitable due to the toxic chemical compounds including polycyclic aromatic hydrocarbons (PAHs) contained within the timber sleepers due to their creosote impregnation process. Additionally, these processes are also unsuitable from an economic point of view and impractical (Manalo et al., 2010). This is due to the fact that they are treated as hazardous waste and their subsequent disposal and storage is unsuitable from an economic point of view compared to ordinary waste.

**Figure 8: Carbon footprint for each sleeper type broken down by LCA phase for a 60-year appraisal period.**

[Figure 8 app. Here]

#### **Comparison of Results**

Attempting to compare the overall environmental impact of the sleepers examined in this study against the values found in different studies has proven difficult. This is due to the different methodological assumptions, as well as differences in the drawn system boundaries and selected functional unit. However, it was possible to compare the EC emissions from this study against other studies by converting the estimated EC to kgCO2e/(sleeper×year) (see table 7). When comparing the results from this study against the values found by Milford and Allwood (2010), it can be seen that there is a sizeable difference between the emissions for hardwood and softwood sleepers, with the values calculated in the current study being 1.9 and 1.3 greater than the estimates made by Milford and Allwood (2010). This difference is not attributed to the service lives or the mass values adopted as the service lives were drawn from the same source, while the mass values are nearly identical as they apply to country-specific track components. The observed differences between the two studies arise due to the adopted CFs (see table 8). In the present study, the most up to date CFs for both materials have been selected, while the CFs adopted by Milford and Allwood (2010) are being classified by RSSB (2018) as obsolete. Considering the steel sleepers, the current study found the embodied emissions from these sleepers to be 20% smaller than the ones cited by Milford and Allwood (2010). The value for steel sleepers used in the current study is more reliable as it has a specific geographical data coverage for the UK. Whereas, the value adopted by Milford and Allwood (2010) is a global average with unknown scope of coverage. Moreover, the value adopted on this study assumes a 41:59 split of virgin and recycled steel content, while the study by Milford and Allwood (2010) doesn’t specify any details on the steel content composition of the analysed sleepers.

**Table 7: Comparison of the embodied carbon impact of different sleepers.**

[Table 7 app. Here]

**Table 8: Comparison of the embodied CFs adopted in this study against the values adopted in other studies (\*EF refers to 100% virgin steel fastening as steel sleepers were not appraised).**

[Table 8 app. Here]

When the values of this study are being compared with the predicted values by Ueda et al. (2003, 1999), the differences are significant, as (i) the mass values of the Japanese sleepers are considerably smaller than the respective UK alternatives. Adding to this, (ii) the assumed service lives (e.g. 15 years for timber, 50 years for both concrete and steel) by Ueda et al. (2003, 1999) are different than the ones assumed in the current study (see table 5). Finally, (iii) the CFs for timber sleepers in their study are 0.2 to 0.3 times smaller compared to ones in the current study. Similarly, the steel sleeper embodied factor assumed in this study is approximately 1.7 times the one derived by Ueda et al. (2003, 1999) (see table 8). Finally, comparing the values of this report against the values predicted by Crawford (2009), it can be inferred that there are some sizeable differences. Unlike the study by Ueda et al. (2003, 1999), the observed differences with Crawford (2009) are not predominantly attributed to the mass and service life estimates assumed. These arise mainly due to the difference between the EF estimated in their study, which are 3.8 (for hardwood) and 4.0 times (for steel) higher than the values adopted in the current study. Moreover, unlike this study, the estimated embodied emissions per sleeper, include these of the fastening system, using an EF for virgin steel which is 4.0 times higher than the one estimated in this study.

### **Sensitivity Analysis**

Following the initial appraisal, a sensitivity analysis has been performed using a ‘one-at-a-time’ approach (OAT). This localised approach enabled the input parameters to be changed one-at-a-time in order to investigate their influence on the final result. Using this method and given the results from the literature review, seven factors were tested for both the high and low load traffic scenarios. These were: the impact of the service life (±30%); sleeper spacing (US and Australian specifications); transport distance (+100-300%); amount of sleepers incinerated with energy recovery (+75-100%); gate-to-site transport mode (modal shift from rail-to-road and *vice versa*); proportion of virgin and recycled steel used (±100%); and productivity of installation equipment (±50%) (see figure 9).

Based on our results, a number of conclusions can be drawn. First, at low traffic loads, regardless of the magnitude of overestimation softwood sleepers will consistently exhibit the best performance. On the contrary, by assuming that the service life of hardwood sleepers is overestimated by more than 20%, results in their performance being the worst compared to softwood and concrete. At high traffic loads, concrete sleepers perform the best even by assuming a 30% overestimation of their service life, while an overestimation of 10% of the service life of hardwood would result in the softwood sleepers outperforming them.

Second, the choice of end-of-life pathway for timber sleepers can reshape their footprint as irrespective of the traffic load scenario examined, these variants outperform both concrete and steel sleepers when a minimum of 50% them follows the combustion pathway with energy recovery. Similarly, although it may not be realistic in practice, the use of 100% secondary steel will result to the steel sleepers outperforming all other variants, regardless of the traffic load scenario examined.

Third, both gate-to-site transport distance and mode choice have an impact to the lifecycle emissions of these structures, but it is deemed negligible compared to their total footprint. For example, the mode choice appears to have an impact of less than 1%. Equally, even by increasing the transport distance for all materials from 50 to 200 km results to an increase of about 0.7 to 2.5% of the total footprint of the examined structures. Similarly, by overestimating the productivity of construction equipment by as much as 50%, will have an impact of <1% to the lifecycle emissions of these sleepers.

Finally, as it is expected the use of Australian track standards has a considerable impact in the carbon footprint of both timber (-5%) and concrete (-9%) sleepers due to the wider spacing requirements (685 mm and 714 mm), resulting on less sleepers installed per stkm. On the contrary, the adoption of North American standards, results on an increase of about +7% of the footprint of concrete sleepers due to the tighter spacing intervals of Class 1 mainlines in the US. This effect is even more pronounced for timber sleepers (+31%), where the spacing is around 495 mm compared to an equivalent 650 mm for UK routes, resulting to an additional 482 sleepers installed per stkm.

**Figure 9: Sensitivity analysis – Tornado chart.**

[Figure 9 app. Here]

### **Carbon Footprint Costs**

Once the GHG emissions resulting from each scenario have been quantified, these are given a monetary value. In valuations for appraisal, the UK government adopts a target-consistent approach, basing this on estimates of the abatement costs required so as to meet specific emissions reduction targets. Considering this, the carbon values given by the DfT (2019) are used to monetise the changes in emissions from each option.

Considering the high traffic load scenario, the use of concrete sleepers, instead of hardwood, softwood, or steel can bring about a benefit of £9,436, £13,886, or £22,231 per stkm, respectively. Conversely, for the low traffic load scenario, the choice of the softwood variant results in a welfare benefit of approximately £4,377 to £10,304 per stkm, depending on the sleeper variant they substitute. For these calculations, the base year for discounting is 2019, with the base test discount rate taken as 3.5% for the first 30 years of the appraisal and a lower discount rate of 3% used thereafter (as recommended by HM Treasury (2018)).

It is worth highlighting that there are some inherent risks associated with the economic evaluation of the costs of CO2e on this study. Thus, in order to overcome this, a stochastic approach has been selected, instead of a deterministic one. This has been done by conducting a Monte Carlo Simulation (MCS). The goal of the adopted methods was to assess the risk associated with our estimates, choosing randomly modifiable values in each iteration. In terms of simulation details, MCS samples were of size 10,000 for each of the models. It was assumed that the evaluated annual cost of carbon is a triangularly-distributed random variable; with the minimum, maximum, and mode being calculated based on the target-consistent marginal abatement costs given as a three-point estimate by DfT (2019). The cumulative probability distribution curves for each sleeper variant (and load scenario) are displayed in Figure 10 below.

**Figure 10: Costs of carbon dioxide cumulative distribution function (for each sleeper variant), [a]: High tonnage scenario [top], [b]: Low tonnage scenario [bottom].**

[Figure 10 app. Here]

A number of conclusions can be drawn from the results of the MCS presented in Figure 10. First, for the lower tonnage routes the expected lifecycle carbon costs for each sleeper type are considerably lower than those for the higher track category routes. Additionally, for the latter route scenario, there is a higher uncertainty for these cost estimates as it can be inferred from the higher width, and flatness of their S-Curves, indicating higher standard deviation (σ). Second, at high traffic loads, there is a clear difference in terms of carbon costs between each sleeper type, with the concrete sleepers bringing the lowest minimum (£12.8k/stkm) and maximum (£40.7k/stkm) carbon cost per stkm compared to the other three variants. Third, for the lower track category routes, the softwood sleepers, have the lowest carbon costs, both in terms of minimum and maximum possible value, with a higher certainty around these estimates, as indicated by the lower value of σ. Then again, for the case of concrete and hardwood sleepers, there is an evident overlap between their S-Curves. Considering this, the former brings a lower minimum cost (£8.1k/stkm (concrete) < £8.5k/stkm (softwood), whereas, its maximum possible value is marginally greater than that of the latter (£26.8/stkm (concrete) > £26.4k/stkm (softwood)). Finally, under high traffic loads the expected carbon costs for the best performing variant (concrete) can be set between £18.9k to £34.6k/stkm with an 80% probability, and an μ value (population mean or expected value) of £26.8k/stkm. Whereas, under the same route scenario, the second best performing alternative (hardwood) will have its carbon costs set between £25.8k to £46.7k at the same probability, and an μ value of £36.2k/stkm. While, for the low traffic load scenario, the best performing variant (softwood) will have its carbon costs set between £7.3k to £13.6k per stkm (80% probability), and an μ value of £9.9k/stkm.

### **Life Cycle Costs (LCC)**

So as to examine prospective trade-offs between the one-off and the on-going financial and environmental externalities throughout the useful life of these structures, their life cycle costs (LCC) were calculated for each scenario. Other elements of the wider social cost such as the impact of air-borne and ground-borne noise have been excluded as data was not readily available. Accordingly, this calculation included only the direct activity costs of each sleeper variant (materials, transport, plant, and time-on-tools labour hours) excluding any indirect costs or overheads, these were sourced directly from Network Rail[[2]](#footnote-2) in 2017/18 prices as per Control Period 5 (CP5: 2014 to 2019). These cost elements were grouped at the standard job activity level and adjusted for inflation since original estimates, using the Consumer Price Index (CPI) based GDP-deflator sourced from the DfT (2019). This conversion is equivalent to multiplying the annual values based in 2017/18 prices with the value of CPI based GDP-deflator in 2019 (103.920) over the baseline value of 2017 (113.920/100 = 1.0392). This implies that over a two-year period the prices have gone up by 3.92%. It is worth noting, that the productivity values for different activities are not constant over time and may require adjustment, however, for the purposes of this research, these have been assumed constant over time as per CP5 estimates (Williams, 2018). Once again, for these calculations the base year for discounting is 2019, with the base test discount rate taken as 3.5% for the first 30 years, and a lower discount rate of 3% used thereafter. The results of the LCC analysis broken down by cost type for each simulated load scenario are displayed in Figure 11 below.

**Figure 11: LCC (2019 prices) of each sleeper variant for a 60-year appraisal period broken down as CapEx, OpEx, and cost of carbon for a low [top] and high traffic tonnage scenario [bottom].**

[Figure 11 app. Here]

Some conclusions can be drawn from the results of the LCC analysis presented in Figure 11. First, irrespective of the sleeper variant or traffic scenario examined, the costs of carbon are small compared to CapEx and OpEx of these structures over a 60-year period, representing just about 1.9-5.5% of the total LCC, depending on the sleeper variant and scenario examined.

Second, for a high traffic tonnage scenario, the concrete sleepers appear to outperform the remaining variants, with the associated benefits of choosing concrete sleepers as opposed to softwood or steel, being approximately £317,790/stkm and £263,639/stkm, respectively. However, these savings are considerably lower when substituting hardwood sleepers, with the equivalent figure being just about £1,487 per stkm. Conversely, for a low traffic tonnage scenario, softwood sleepers outperform all other options in terms of LCC, with the savings per stkm (over a 60-year period) being at c. £95,234 and £163,452 compared to the concrete and steel sleepers, respectively. Once again, hardwood sleepers perform marginally worse than the best performing alternative, with the LCC savings per stkm of the softwood variant compared to hardwood being just around £4,377.

Third, for the low traffic tonnage scenario, CapEx is the highest contributor to the LCC regardless of the sleeper variant, with this phase accounting between 74.7-79.2% of the whole-life cycle costs (WLCC). In contrast, for the high traffic scenario, the share of OpEx dominates the WLCC for all variants (c. 48.9-65.7%) apart from the concrete sleepers. Considering the latter, their CapEx share is approximately 58.3%, compared to an equivalent 37.7% attributed to ongoing expenditure by the infrastructure manager. This result is partly due to the greater service life of these sleepers, translating to fewer interventions compared to the remaining variants, but also due to the heavy discounting used, placing heavier weight on the investment made during the earlier years of the appraisal.

#### **Scenario 4: Installation of Concrete Sleepers with USPs**

In the course of the second half of CP6 (2019-2024), concrete sleepers are anticipated to be in routine production with USPs in the UK. A USP is a highly elastic (resilient) element attached underneath the sleepers, so as to provide an intermediate elastic layer between the sleeper and the ballast. Made from polyurethane, rubber, EVA (ethylene-vinyl acetate), USPs were first developed in the 1980s and are widely used across Europe but not in the UK, other than on London Underground.

The infrastructure manager expects concrete sleepers fitted with USPs to last 50% longer than the in-service concrete variants, for an additional cost of £15/sleeper (in 2009 prices) or £17.8/sleeper (in 2019 prices). Against this background, an additional scenario of installing concrete sleepers with USPs (stiff 4-mm thick) was simulated, resulting in an additional cost of £27k/stkm (c. £22-23k/stkm in 2009 prices) representing a cost increase in material supply per stkm of between 26-32% (to the present-day sleeper cost), and a carbon surplus of 49.76t.CO2e for installing USPs per km of single track.

The simulated scenarios assumed that the inclusion of USPs prolong the concrete sleeper service life by 50% (for example, for the high traffic load scenario, from 24 to 36 years). Based on our analysis (Figure 12), the inclusion of USPs will improve the economic profile of concrete sleepers over a 60-year period by c. £65,000 to £100,000 per stkm, as well as an equivalent carbon saving of about 23 to 73t.CO2e per stkm, depending on the load scenario examined. These improvements are in effect due to the enhanced service life of the concrete variant, reducing the number of interventions required over the examined lifecycle, but also delaying the time of initial investment for track relay by 12 (high) to 23 years (low), depending on the examined route. Finally, by assuming the installation of carbon-neutral USPs, these carbon savings would rise up to about 122t.CO2e per stkm for both the low and the high tonnage route. Considering this, end-of-life tyres can be an appealing option for treatment, and subsequent use as USPs (Sol-Sánchez et al., 2014), while maintaining the track at an adequate track quality standard (Sol-Sánchez et al., 2016).

**Figure 12: LCC (2019 prices) of each sleeper variant, including the scenario of concrete sleepers with USPs fitted for a 60-year appraisal period broken down as CapEx, OpEx, and cost of carbon for a low [top] and high traffic tonnage scenario [bottom].**

[Figure 12 app. Here]

# **Conclusions**

This study evaluated and compared the lifecycle GHG emissions associated with the four most common sleeper types present in the UK rail network. It estimates the embodied material, process and transport emissions linked with the lifecycle activities of construction, relay and end-of-life of these variants at low and high traffic loads. Based on the results of this work the following conclusions can be drawn. Firstly, under the low traffic scenario, the softwood sleepers appear to be the most favourable option from a GHG emissions point of view. In contrast, at high traffic loads, the concrete sleepers are preferred as their prolonged service life, results in their sizeable embodied emissions’ share being spread out over the examined lifecycle period, this means that the volume of required interventions will be much smaller than, for example, that of softwood sleepers. The analysis revealed that the already burdensome footprint of steel sleepers is being magnified at high traffic loads, primarily, due to their unsuitability for heavy trafficked applications. The renewal figures from this scenario suggest that the steel sleeper interventions emit almost twice more CO2e compared to the other variants. Considering timber sleepers, it has been found that the choice of end-of-life pathway following life-expiration is a critical factor of their environmental performance. Depending on the scenario selected the end-of-life phase of hardwood and softwood accounts for between 17-33.7% of their footprint. When these impacts are compared with the associated impact of steel and concrete recycling, the figures suggest that timber landfilling result in 100-180 times higher GHG emissions compared to that of concrete recycling and a staggering amount of 325-560 times more CO2e emissions compared to that of steel recycling. Nevertheless, it has been shown that if a minimum of 50% of the timber sleepers follow the combustion pathway with heat recuperation, then a GHG reduction potential of between 11-18%, measured as a percentage share of their total footprint can be realized. Finally, despite the inclusion of concrete’s carbonation during its primary life, the effect of this phenomenon has been confirmed to be negligible, accounting for less than 3% of its overall footprint. Nonetheless, it is believed that the carbon uptake of the recycled downstream concrete products during their secondary life can be significant, thus, it is recommended to scope in this activity in future analyses.

This work also included an LCC analysis of these sleepers. Based on our results the following conclusions can be drawn. First, irrespective of the sleeper variant or traffic scenario examined, the costs of carbon are small (representing 1.9-5.5% of the total LCC) compared to the lifecycle CapEx and OpEx of these structures. Second, for a high traffic tonnage scenario, the concrete sleepers appear to outperform the remaining variants, displaying benefits of up to £317,790/stkm; whereas, for a low traffic tonnage scenario, softwood sleepers result on a maximum WLCC saving of £163,452/stkm, with this varying depending on the variant they substitute. Finally, by installing concrete sleepers fitted with stiff USPs at high tonnage routes, their expected benefits are magnified. Based on our results, the inclusion of this intervention will achieve additional LCC savings in the magnitude of c. £65,000-100,000 per stkm of installation, and an equivalent reduction in their carbon footprint of about 23-73t.CO2e/stkm, depending on the annual tonnage of the route. Then again, it has been shown that the use of carbon-neutral USPs can amplify these savings even further, offering a more appealing option from an environmental viewpoint. In a future update, it is envisaged to include an analysis of the recently developed composite sleepers (see Ferdous et al. (2015)), which were not included in the present study as data was not readily available.

A number of limitations remain which are worth investigating further of which two are considered key. First, by assessing any infrastructure design purely from a GHG lens, environmental superiority cannot be fully claimed. Considering this, in future studies these variants should be examined against a more extensive list of impact indicators to provide more definitive comparisons. Secondly, from a WLCC perspective, indirect costs and overheads, as well as other elements of the wider social cost, such as the impact of air-borne and ground-borne noise, were not included in the current study. These can be included in future studies.

# **Declarations of interest**

None.

# **Acknowledgments**

This work was supported by the EPSRC-funded Track to the Future (EP/M025276/1) project. The authors would like to express their gratitude to Lucie Anderton and Lauren Brown from Rail Safety and Standards Board for kindly granting us access to the RSSB’s Rail Carbon Tool. The authors would also like to thank Network Rail and in particular Julian Williams for his support in providing data to carry out this research.

# **Bibliography**

Akbarnezhad, A., Ong, K.C.G., Chandra, L.R., 2014. Economic and environmental assessment of deconstruction strategies using building information modeling. Automation in Construction 37, 131–144. https://doi.org/10.1016/j.autcon.2013.10.017

Akbarnezhad, A., Xiao, J., 2017. Estimation and Minimization of Embodied Carbon of Buildings: A Review. Buildings 7, 5. https://doi.org/10.3390/buildings7010005

Ashby, M.F., 2009. Materials and the Environment Eco-Informed Material Choice. Butterworth-Heinemann, Oxford, UK.

Baron, T., Tuchschmid, M., Martinetti, G., Pépion, D., 2011. High Speed Rail and Sustainability. Background Report: Methodology and results of carbon footprint analysis. Paris.

Bilec, M., Ries, R., Matthews, H.S., Sharrard, A.L., 2006. Example of a Hybrid Life-Cycle Assessment of Construction Processes. Journal of Infrastructure Systems 12, 207–215. https://doi.org/10.1061/(ASCE)1076-0342(2006)12:4(207)

Bolin, C.A., Smith, S.T., 2013. Life Cycle Assessment of Creosote-Treated Wooden Railroad Crossties in the US with Comparisons to Concrete and Plastic Composite Railroad Crossties. Journal of Transportation Technologies 3, 149–161. https://doi.org/10.4236/jtts.2013.32015

BSI, 2011. PAS 2050: 2011, Specification for the assessment of the life cycle greenhouse gas emissions of goods and services. London, United Kingdom.

Carbon Trust, 2016. Conversion factors guide 2016 update (CTL153).

Chester, M., 2008. Life-cycle environmental inventory of passenger transportation in the United States. Institute of Transportation Studies. University of California, Berkeley, Institute of Transportation Studies.

Collins, F., 2010. Inclusion of carbonation during the life cycle of built and recycled concrete: Influence on their carbon footprint. International Journal of Life Cycle Assessment 15, 549–556. https://doi.org/10.1007/s11367-010-0191-4

Crawford, R.H., 2009. Greenhouse gas emissions embodied in reinforced concrete and timber railway sleepers. Environmental Science and Technology 43, 3885–3890. https://doi.org/10.1021/es8023836

DBEIS, 2018. 2018 Government GHG Conversion Factors for Company Reporting: Methodology Paper for Emission Factors [WWW Document]. URL https://www.gov.uk/government/publications/greenhouse-gas-reporting-conversion-factors-2018 (accessed 8.8.18).

DEFRA, 2006. Carbon Balances and Energy Impacts of the Management of UK Wastes - Defra R&D Project WRT 237 Final Report. Oxford, United Kingdom.

DfT, 2019. WebTAG: TAG data book, May 2019 [WWW Document]. Transport modelling and appraisal. URL https://www.gov.uk/government/publications/tag-data-book#history (accessed 10.28.19).

DfT, 2018. TAG Unit A1.1, Cost-Benefit Analysis [WWW Document]. A1 - Cost-Benefit Analysis. URL https://www.gov.uk/guidance/transport-analysis-guidance-webtag (accessed 10.28.19).

Esveld, C., 2001. Modern Railway Track, 2nd ed. MRT-Productions, Zaltbommel, The Netherlands.

Ferdous, W., Manalo, A., 2014. Failures of mainline railway sleepers and suggested remedies - Review of current practice. Engineering Failure Analysis 44, 17–35. https://doi.org/10.1016/j.engfailanal.2014.04.020

Ferdous, W., Manalo, A., Van Erp, G., Aravinthan, T., Kaewunruen, S., Remennikov, A., 2015. Composite railway sleepers - Recent developments, challenges and future prospects. Composite Structures 134, 158–168. https://doi.org/10.1016/j.compstruct.2015.08.058

Hammond, G.P., Jones, C., 2011. Inventory of Carbon & Energy (ICE) Database Version 2.0. Bath: Sustainable Energy Research Team (SERT).

HM Treasury, 2018. The Green Book: Central Government Guidance on Appraisal and Evaluation. Her Majesty’s Treasury (HM Treasury).

ICE, Mott MacDonald, BRE, 2010. CESMM3: Carbon & Price Book 2011 (CESMM3 Series), 2010/2011 ed. ICE Publishing, Thomas Telford, London, United Kingdom.

IPCC, 1996. Chapter 6: Waste, in: Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories: Reference Manual. Intergovernmental Panel on Climate Change (IPCC), p. 32.

ISO, 2006a. Environmental Management: Life Cycle Assesment - Principles and Frameworks, ISO 14040 (E). Geneva, Switzerland.

ISO, 2006b. Environmental Management: Life Cycle Assesment - Requirements and Guidelines, ISO 14044 (E). Geneva, Switzerland.

Kiani, M., Parry, T., Ceney, H., 2008. Environmental life-cycle assessment of railway track beds. Proceedings of the Institution of Civil Engineers - Engineering Sustainability 161, 135–142. https://doi.org/10.1680/ensu.2008.161.2.135

Krezo, S., Mirza, O., He, Y., Makim, P., Kaewunruen, S., 2016. Field investigation and parametric study of greenhouse gas emissions from railway plain-line renewals. Transportation Research Part D: Transport and Environment 42, 77–90. https://doi.org/10.1016/j.trd.2015.10.021

Krezo, S., Mirza, O., Kaewunruen, S., Sussman, J.M., 2018. Evaluation of CO2 emissions from railway resurfacing maintenance activities. Transportation Research Part D: Transport and Environment 65, 458–465. https://doi.org/10.1016/j.trd.2018.09.019

Künniger, T., Richter, K., 1998. Comparative life cycle assessment of Swiss railroad sleepers, 29th Annual meeting of the international research group on wood preservation (IRG/WP). Maastricht.

Lee, C.K., Lee, J.Y., Kim, Y.K., 2008. Comparison of environmental loads with rail track systems using simplified life cycle assessment (LCA). WIT Transactions on the Built Environment 101, 367–372. https://doi.org/10.2495/UT080361

MAINLINE, 2014. Work Package 5, Deliverable 5.5: MAINLINE Life Cycle Assessment Tools (LCATs) - Plain track model.

Manalo, A., Aravinthan, T., Karunasena, W., Ticoalu, A., 2010. A review of alternative materials for replacing existing timber sleepers. Composite Structures 92, 603–611. https://doi.org/10.1016/j.compstruct.2009.08.046

Mason, L., 2013. Life Cycle Assessment of different forms of railway track. Southampton: University of Southampton, Faculty of Engineering and the Environment.

MATISA Matériel Industriel S.A., 2018a. Track Construction Trains [WWW Document]. 100% Trains de construction Matisa [English]. URL http://www.matisa.ch/ (accessed 8.15.18).

MATISA Matériel Industriel S.A., 2018b. Track Renewal Trains [WWW Document]. 100% Trains de renouvellement Matisa [English]. URL http://www.matisa.ch/ (accessed 8.15.18).

Milford, R.L., Allwood, J.M., 2010. Assessing the CO2 impact of current and future rail track in the UK. Transportation Research Part D: Transport and Environment 15, 61–72. https://doi.org/10.1016/j.trd.2009.09.003

Network Rail, 2009. Comparing Environmental Impact of Conventional and High Speed Rail. London, UK.

Nilsson, L.-O., 2011. A new model for CO2-absorption of concrete structures. CO2-cycle in cement and concrete Part 7: Models for CO2-absorption TVBM-3158.

Ortega, A., Blainey, S., Preston, J., 2018. Assessing Whole-Life Carbon Footprint of Under Sleeper Pad Installation for Ballasted Track. Journal of Transportation Engineering, Part A: Systems 144, 04018073. https://doi.org/10.1061/jtepbs.0000192

Owens, M., 2007. Timber – The Low Carbon Footprint for Rail, in: AusRAIL PLUS, 4-6 December 2007. Australasian Railway Association (ARA), Sydney, pp. 1–9.

Pachauri, R.K., Allen, M.R., Barros, V.R., Broome, J., Cramer, W., Christ, R., Church, J.A., Clarke, L., Dahe, Q., Dasgupta, P., Dubash, N.K., Edenhofer, O., Elgizouli, I., Field, C.B., Forster, P., Friedlingstein, P., Fuglestvedt, J., Gomez-Echeverri, L., Hallegatte, S., Hegerl, G., Howden, M., Jiang, K., Cisneroz, B.J., Kattsov, V., Lee, H., Mach, K.J., Marotzke, J., Mastrandrea, M.D., Meyer, L., Minx, J., Mulugetta, Y., O’Brien, K., Oppenheimer, M., Pereira, J.J., Pichs-Madruga, R., Plattner, G.-K., Pörtner, H.-O., Power, S.B., Preston, B., Ravindranath, N.H., Reisinger, A., Riahi, K., Rusticucci, M., Scholes, R., Seyboth, K., Sokona, Y., Stavins, R., Stocker, T.F., Tschakert, P., van Vuuren, D., van Ypserle, J.-P., 2014. Climate Change 2014: Synthesis Report, in: Pachauri, R.K., Meyer, L. (Eds.), Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. IPCC, Geneva, Switzerland, p. 151.

Primary Power International, 2004. Emission Reduction/Removal Project Document. GHG Reductions through Offset of Canadian Grid Electricity and Methane Avoidance through Waste Wood Diversion Wood Diversion.

RSSB, 2018. RSSB Rail Carbon Tool [WWW Document]. Version: 1.1.3.0. URL https://www.railindustrycarbon.com/KnowledgeBase/Account/LogOn?ReturnUrl=%2fKnowledgeBase%2f

RSSB, 2010. Whole Life Carbon Footprint of the Rail Industry. London, United Kingdom.

Salas, D.A., Ramirez, A.D., Rodriguez, C.R., Petroche, D.M., Boero, A.J., Duque-Rivera, J., 2016. Environmental impacts, life cycle assessment and potential improvement measures for cement production: A literature review. Journal of Cleaner Production 113, 114–122. https://doi.org/10.1016/j.jclepro.2015.11.078

Schmied, M., Mottschall, M., 2010. Treibhausgasemissionen durch die Schieneninfrastruktur und Schienenfahrzeuge in Deutschland, (FKZ 363 01 244). Büro Berlin.

SERCO, 2014. VTISM Version 2.6.6 User Guide (No. SERCO/SRTS/E.006693.01/004). Hook, United Kingdom.

SERCO, 2007. Track Strategic Planning Application: Technical Basis (No. SA/TSS/14870/W006). Warrington, Cheshire, United Kingdom.

Snowdon, P., Eamus, D., Gibbons, P., Khanna, P.K., Keith, H., Kirschbaum, M.U.F., 2000. Synthesis of Allometrics, Review of Root Biomass and Design of Future Woody Biomass Sampling Strategies (No. 17), National Carbon Accounting System Technical Report. Canberra, Australia.

Sol-Sánchez, M., Moreno-Navarro, F., Rubio-Gámez, M.C., 2014. Viability of using end-of-life tire pads as under sleeper pads in railway. Construction and Building Materials 64, 150–156. https://doi.org/10.1016/j.conbuildmat.2014.04.013

Sol-Sánchez, M., Pirozzolo, L., Moreno-Navarro, F., Rubio-Gámez, C., 2016. Reducing Railway Maintenance: The Effectiveness of Combining the Stoneblowing Technique with Rubber Elements from Waste Tires, in: Transportation Research Board 95th Annual Meeting, January 10-14, 2016. Washington, DC, United States, p. 12.

Stripple, H., 2013. Greenhouse gas strategies for cement containing products. Part of the research project CO2 cycle in cement and concrete (No. B2024). Göteborg, Sweden.

Stripple, H., Uppenberg, S., 2010. Life cycle assessment of railways and rail transports - Application in environmental product declarations (EPDs) for the Bothnia Line (No. B1943). Göteborg, Sweden.

SUSTRAIL, 2012. The sustainable freight railway: Designing the freight vehicle – track system for higher delivered tonnage with improved availability at reduced cost (No. 265740 FP7 - THEME [SST.2010.5.2-2.], D2.3 Track design requirements for reduced maintenance).

The Greenhouse Gas Protocol, 2004. A Corporate Accounting and Reporting Standard | The Greenhouse Gas Protocol, Revised. ed, Greenhouse Gas Protocol Webpage. World Resources Institute and World Business Council for Sustainable Development, c/o Earthprint Limited, USA.

Tuchschmid, M., Knörr, W., Schacht, A., Mottschall, M., Schmied, M., 2011. Carbon Footprint and environmental impact of Railway Infrastructure.

Tuchshmid, M., 2009. Carbon Footprint of High-Speed Railway Infrastructure (Pre-Study). Zurich.

Ueda, H., Miyauchi, T., Tsujimura, T., 2003. Application of lifecycle assessment to Shinkansen vehicles and cross ties in Japan. Proceedings of the Institution of Mechanical Engineers, Part F: Journal of Rail and Rapid Transit 217, 271–278. https://doi.org/10.1243/095440903322712874

Ueda, H., Takai, H., Tsujimura, T., Emoto, M., 1999. Fundamental Investigation of LCA of Cross Tie. Quarterly Report of Railway Technical Research Institute (RTRI) 40, 210–213. https://doi.org/10.2219/rtriqr.40.210

van den Dobbelsteen, A., Alberts, K., 2001. Milieueffecten van bouwmaterialen. Duurzaam omgaan met grondstoffen (In Dutch). Delft, The Netherlands.

von Rozycki, C., Koeser, H., Schwarz, H., 2003. Ecology profile of the German high-speed rail passenger transport system, ICE. The International Journal of Life Cycle Assessment 8, 83–91. https://doi.org/10.1007/bf02978431

Werner, F., 2008. Life Cycle assessment (LCA) of railway sleepers. Comparison of railway sleepers made from concrete, steel, beech wood and oak wood. Study by Umwelt & Entwicklung for SGH. Zürich, Switzerland.

Williams, J., 2018. Track Maintenance Model, V1.1 (No. Issue 1 Draft 3). United Kingdom: Network Rail.

1. Equivalent Million Gross Tonnes Per Annum [↑](#footnote-ref-1)
2. Williams, J., 2019. Personal communication (Network Rail). [↑](#footnote-ref-2)