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Hybrid Electric vs Conventional Vehicle: Life Cycle Assessment and External Costs

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Abstract
The environmental damage costs ("external costs") of the hybrid electric vehicle (HEV) are compared with those of a conventional gasoline vehicle, by carrying out a life cycle assessment (LCA) coupled with an impact pathway analysis based on the ExternE project series of the EC. The LCA data for vehicle production are from Delucchi et al of UC Davis and from studies by MIT, those for fuel production from the well-to-wheel software of Argonne National Laboratory. The HEV is a Toyota Prius whereas the Toyota Camry and Corolla are taken to specify the conventional alternative. The damage costs of the HEV are significantly lower than those of a conventional vehicle, especially in urban driving. The difference is proportional to the fuel savings since the damage costs from the vehicle production stage are nearly the same. The damage costs of the Camry, Corolla and Prius are, respectively, 0.88, 0.77 and 0.53 Eurocents/km. The damage cost of a hypothetical plug-in version of the Prius varies between 0.18 and 0.53 Eurocents/km, the result depending strongly on the contribution of fossil fuels in the electricity generation mix.
Key words: *life cycle assessment, hybrid electric vehicle, conventional vehicle, external costs, ExternE, impact pathway analysis, greenhouse gases*

1 - Introduction and Objectives

The hybrid electric vehicle (HEV) appears as an attractive option until fuel cell vehicles are sufficiently well developed to compete in the market. But whereas several manufacturers, especially Toyota and Honda, are already actively selling HEVs in the USA, that has not yet been the case in the EU. The objective of the present paper is therefore to quantify the environmental advantages of the HEV for European conditions.

For a correct evaluation and comparison of different vehicle options a life cycle assessment (LCA) is required because one could get a very misleading picture by just looking at the pollutants emitted during driving. Whereas numerous LCA studies of cars have been carried out [e.g. Delucchi 2003, MacLeana & Lave 2003, Weiss et al 2000], their results for the environmental effects have been reported only in terms of the quantities of the emitted pollutants. That makes the interpretation and use of these studies difficult because it is not obvious how harmful the different pollutants are. The results would be much simpler to understand if the impacts of all the pollutants were expressed in terms of a single common metric. The most convenient metric is monetary, i.e. in terms of damage costs (such costs are often called "external costs" because they are not included in market prices). The monetary valuation of environmental damage has the great advantage of allowing a direct comparison with conventional costs and a cost-benefit analysis. For that one needs an impact pathway analysis (IPA). By contrast to most conventional LCA's, we calculate real impacts and costs rather than a set of non-commensurate indices of "potential impacts".

In recent years much progress has been made in this regard, thanks to several major research projects in the USA and the EU. The most extensive and up-to-date research of the kind is the ExternE project series of the EU, a large multinational and interdisciplinary undertaking in which the present authors are active participants (a detailed description of the IPA methodology is found at http://www.externe.info). Environmental cost-benefit analysis is now used extensively in the USA and the EU. In particular, the European Commission recently decided that the internalization of external costs is a key tool for the attainment of sustainable development (one of the main policy goals of the EU). This obliges industries that manufacture or sell their products in Europe to take into account their environmental damage costs.

2 - Methodology

LCA

The life cycle of a vehicle can be broken down into the following stages:

- Production of the materials needed for making the vehicle,
- Assembly of the materials,
- Fuel feedstock (e.g., extraction of petroleum from the ground),
- Fuel supply (refining of petroleum and transport of fuel),
- Utilization of the vehicle, and
- Disposal of the vehicle at the end of its useful lifetime.

Disposal of the vehicle at the end of its life involves as major steps i) dismantling the vehicle, ii) recycling the recyclable fraction, and iii) disposing the rest fraction in a landfill or incinerator. The impacts due to the dismantling of the vehicle are entirely negligible, as can be seen from the fact that the energy for dismantling is about ⅓ of the energy requirement for vehicle assembly [Stodolsky et al., 1995] and that the impacts of the assembly are only a very small part of the total life cycle impact. For the present study we have found it most convenient to include the recycling in the first stage (Production of the materials), because the available data for materials include the emissions due to recycled materials. The impacts from the disposal of the rest fraction are negligible if current environmental regulations for waste disposal are respected [see Rabl, Spadaro & McGavran 1998]. For these reasons we do not show explicit results for the disposal stage.

Sources of information for the LCA phase include:

- A large and comprehensive review paper by MacLeana & Lave [2003],
- A major LCA of many vehicle and fuel technologies by MIT [Weiss et al 2000],
- The most comprehensive well-to-wheel analysis (the GREET software of ANL [2004]),
- The most comprehensive LCA of vehicle and fuel technologies [Delucchi 2003], and
- For on-the-road emissions and fuel economy USEPA and independent test organizations.

IPA

The principal steps of the impact pathway analysis include:

1. Emission: specification of the relevant technologies and the environmental burdens they impose (e. g. kg of NO_x emitted per km by vehicle);
2. Dispersion: calculation of increased pollutant concentration in all affected regions (e. g. incremental concentration of ozone, using models of atmospheric dispersion and chemistry for ozone formation due to NO_x);
3. Damage in physical units: calculation of the dose from increased exposure and calculation of impacts () from this dose using a dose-response function (e. g. number of cases of asthma due to this increase in ozone);
4. Cost: monetary valuation of impacts (e. g. multiplication by the cost of a case of asthma).

We have taken into account all the impacts that have been quantified in the latest phases of ExternE [2000 and 2004], in particular: health costs, changes in agricultural yield, damage to materials and buildings, and impacts of climate change. There are of course additional types of impact, for instance acidification and eutrophication of ecosystems, but in monetary terms they turn out to be far less important for the pollutants emitted during the life cycle of vehicles. Since the present analysis has been concerned only with damages due to pollutants, we have not considered noise. Preliminary estimates by the ExternE team suggest that in European cities noise can impose external costs of a magnitude comparable to those of pollution. Since the HEV is much quieter it enjoys another major advantage in terms of external costs. The result of an IPA is the damage cost per kg emission, or marginal damage cost (MDC). Multiplication of the emission by MDC gives the damage cost by pollutant; and aggregation across all pollutants gives the total damage cost. Excluding global warming impacts, health damages typically account for more than 95% of the total cost. An uncertainty analysis based on the methodology of Rabl and Spadaro [1999] indicates that the damage cost function has a lognormal distribution with geometric standard deviation σ_G = 3. The 68% confidence interval is ⅓ to 3 times the geometric mean (median) of the distribution. The ratio of median to mean is about 0.55.

The mean damage costs per kg of pollutant are listed in Table 1; they are typical values for central Europe. Based on numerous site-specific calculations for ExternE we estimate that the cost per kg of $PM_{2.5}$ for urban driving is a factor of twenty higher than for rural driving. However, for NO_x and SO_2 the difference between urban and rural emissions is negligible because their impacts are believed to arise from secondary pollutants (aerosols and ozone) that are created gradually and further away from the emission site. For CO and Pb the variation with local population density can also be neglected because the dispersion of these pollutants in the environment covers very large distances (Pb through food chain pathway, and CO has a long atmospheric residence time).

The cost of the greenhouse gases (CO_2, CH_4 and N_2O) does not depend on the emission site. For greenhouse gases other than CO_2 the cost are expressed in terms of equivalent emissions of carbon, using the global warming potential (GWP). The damage cost per kg of CO_2 remains uncertain and controversial. The estimates by various experts fall in the range of about 0.002 to 0.050 €/kg of CO_2. Here we follow ExternE [2004] in using 0.019 €/kg of CO_{2equiv}. Not only is this a very reasonable choice in view of the various estimates but it equals the abatement cost in the EU that is implied by the acceptance of the Kyoto protocol. Because of the growing development of a worldwide market for trading CO_2 emissions, the effective cost per kg of CO_2 permits is fairly close to this value.

3 - Assumptions and Data

Reference vehicle

Ideally, the HEV should be compared to a gasoline version that is identical except for the drive train. However, since the difference in damage costs is crucially dependent on the fuel consumption, we have chosen the Toyota Prius as the HEV since it was the first to enter the market and its consumption and emissions data are more reliable than for more recent models that are offered in both versions. Unfortunately for the present analysis there is no exact conventional equivalent. The Toyota Camry comes closest, although it is slightly larger; for that reason we have also analyzed the data for the Toyota Corolla. We have also considered a plug-in (grid-connected) version of the Prius or PHEV; at this time, this vehicle is not available. In our assessment, we assume the batteries of the PHEV are charged overnight using baseload electricity, and there is enough energy stored in the batteries to travel 90% of the distance as an all-electric vehicle (EV). The remainder of the time the vehicle operates as a hybrid car would. Regarding the electricity generation mix, two scenarios were analyzed: i) fossil fuels contribute 60% of baseload electricity (PHEV, high), and ii) electricity supply is from nuclear power and renewable fuels (PHEV, low).

Production of vehicles

We have taken the inventory data of Delucchi [2003], Weiss et al [2000] and AMM [2003] (they are very similar) and scaled them in proportion to the actual mass of the cars under consideration. The first two of these sources provide separate inventories for conventional and for hybrid cars.

The vehicle disposal stage involves the dismantling of the vehicle and recovery of any materials that can be reused, and the treatment of the remainder. Metals are easy to recycle, and at the present time already 90% to 95% of the metals are reused, either in the same or in different sectors of the economy. The quantity of glass in a car is small and so are the associated impacts. Plastics are much more difficult to recycle, and it is not clear what fraction will find another use. Fluids in cars are either water or petroleum based. Most of the latter are already recycled in some form or other (including thermal recycling via incineration). In any case the recycled fractions are bound to increase under the growing pressure from governments, especially in the EU, to increase the recycling of waste. The MIT study [Weiss et al 2000] assumes optimistically that 50% of plastics are recycled by 2020; Delucchi [2003] does not try to estimate a percentage. In the present study we take a current estimate of 5% for plastics and rubber.

For this analysis it does not matter whether the materials are reused in vehicles or in other sectors, because the emissions avoided by recycling are essentially independent of the sector in which the production of virgin materials is avoided. We assume that an average passenger car is driven 200,000 km during its 15-year lifetime.

Emissions during use of vehicle

The assumptions regarding fuel consumption and tailpipe emissions are summarized in Table 2. Energy efficiency results by the EPA are based on standard and supplemental federal testing procedures (http://www.fueleconomy.gov), but are not representative of real-world driving conditions, due to such factors as personal driving habits, on-road driving conditions, on-board equipment malfunction, air-condition use, and extra load. We have therefore looked at the results of actual driving tests by independent testing organizations such as Consumer Reports (http://www.ConsumerReports.org), Consumer Guide (http://www.auto.consumerguide.com) and the US Department of Energy (advance Vehicle Testing Activity, http://www.eere.energy.gov/), and we have chosen as representative values the mean of these results. For emissions from vehicle operation, we have used, whenever available, USA published data by ACEEE, CEC, EIA, INEEL, NREL, US DOE, US DOT, US EPA and Toyota Motors. In Europe, additional test results are available from the ARTEMIS project of the EU (http://www.trl.co.uk/artemis/). In the absence of data, we assume the regulatory limit values of the national Tier 2 emission standards of the USA. Turns out that perfect knowledge of actual emissions is not critical because the external costs are dominated by CO_2 releases, which are dependent on fuel consumption.

4 - Results

Vehicle production

The damage cost due to the vehicle production stage is shown in Figure 1, with breakdown by pollutant. It represents very roughly one percent of the price of the car. Greater detail for the contribution of the individual materials and of vehicle assembly is provided in Figure 2 for the Camry. The corresponding numbers for the Prius are very similar, slightly smaller in proportion to the vehicle mass; if the rest of the car were identical the only difference would come from the battery (Pb for the Camry, a larger quantity of the much less toxic Ni for the Prius). Compared to the total cost of this stage the contribution of vehicle assembly is only about a quarter. NMVOCs are relatively more important for assembly than for the individual materials because of emissions during painting and coating (Toyota has recently switched to water-based paints, which should reduce emissions, and consequently damage costs, by a factor two). In terms of damage costs, the most important materials are iron, steel, aluminum and plastics.

Well-to-tank and tank-to-wheel

For the present study, the GREET model (version 1.5a, dated April 2001) was used for the well-to-wheel stages. This software was developed by ANL [2004]. Since its first release in 1996, GREET has been used extensively by industry, government and academia. Although the default input data and assumptions are for the United States (e.g.., electricity mix for feedstock production, fuel specifications, upstream boiler emission factors, etc.), the technologies are sufficiently similar in the EU. Figure 3 shows the breakdown by pollutant for the stages of the well-to-wheel analysis and vehicle production. We do not show the damage cost of CO because it is so small that its contribution to the total would not even be visible in the graph. Only the Camry is shown, because these impacts are proportional to fuel consumption.

Comparison of total cost per vkm

The results for the total damage cost, expressed in Eurocents per vehicle km (vkm), are shown in

Figure 4. These are mean estimates, with an uncertainty range extending between 20% of the estimate at the lower bound and up to 150% at upper bound (lognormal distribution with geometric standard deviation of 3). The hybrid vehicles cause by far the lowest damage. Vehicle utilization accounts for the largest contribution to the total damage cost in the case of the Camry, Corolla and Prius, but its share of the total drops drastically to between 10% and 20% for the plug-in hybrid. It is quite evident that the well-to-tank damage cost of the plug-in Prius is sensitive to the assumption of the share of fossil fuels in the electricity supply mix, a factor of three separates PHEV low and high estimates. Damages from vehicle production are essentially proportional to vehicle mass since the relative composition does not vary much. Finally, Figure 5 compares the damage costs of European gasoline and diesel (direct injection, DI) light duty passenger cars around 2010. Although diesel vehicles have lower $CO_{2,equiv}$ emissions per km compared to gasoline engines, particulate and NOx tailpipe emissions are higher (of particular concern is the potential impact on health from exposure to increasing emissions of NO_2 from diesel cars equipped with a particulate filter, DPT.) The net effect is that the diesel car has a slightly higher total damage cost than the gasoline car on the basis of ¢/km. Hybrid (NiMH battery) options, of course, have significantly lower damage costs.

Conclusion

The HEV has much lower damage costs than the conventional alternative , the difference being proportional to the fuel savings (e.g., 0.53 ¢/km for the Prius vs. 0.88¢/km for the Camry, or a 40% damage cost reduction). A further advantage can be gained from the plug-in option (charging the HEV with baseload electricity at night); this option is strongly dependent on the driving pattern and share of fossil fuels in the electricity generation mix.

Acknowledgments

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Table 1. Marginal damage costs in € per kg of pollutant. $PM_{2.5}$ results are higher because emissions occur at ground-level; also the particles are smaller and more toxic.

Pollutant	**€ per kg pollutant**
Particulate Matter, PM	
Tailpipe emissions ($PM_{2.5}$ urban)	750
Tailpipe emissions ($PM_{2.5}$, rural)	37.4
Tall, stationary sources (PM_{10}, urban)	44.8
Tall, stationary sources (PM_{10}, rural)	22.4
Vehicle production sources (PM_{10})	22.4
SO_2 (via sulfate aerosols, crops and material damages)	4.6
NOx (via nitrate aerosol formation)	2.4
CO_2	0.019
NMVOC (non-methane volatile organic compounds)	1.14
CO	0.0016
Pb (includes both inhalation and oral dose)	600
Ni (inhalation dose only)	3.8

Adapted from the ExternE [2004]. Estimates based on life-cycle calculations for typical European conditions using data from the EcoInvent emissions inventory.

Table 2: Vehicle data, consumption (liters per 100km) & tailpipe emissions (grams per vehicle km).

Vehicle	**Energy efficiency L/100km**	**$PM_{2.5}$ g/km**	**NOx g/km**	**CO g/km**	**NMVOC g/km**	**SO_2 120ppmS g/km**	**$CO_{2,equiv}$ g/km**
Toyota Camry 2.4L, 4-cyl (mid-size car, 1435 kg)	10.1	0.0012	0.02	0.62	0.006	0.018	240
Toyota Corolla 1.8L, 4-cyl (compact car, 1177 kg)	8.2	0.0012	0.12	2.61	0.19	0.014	195
Toyota Prius 1.5L, 4-cyl (1311 kg)	5.5	0.001	0.02	0.13	0.006	0.01	131
Plug-in Prius (PHEV) (HEV driving mode, 1538 kg)	6.2	0.001	0.02	0.13	0.006	0.011	147
Plug-in Prius (PHEV) (EV driving mode)	2.6 (L of gasoline equivalent)	0	0	0	0	0	0

(1) Energy efficiency expressed as gasoline equivalent for "real-world" driving conditions; vehicle distance traveled 200,000 km over lifetime; useful vehicle lifetime is 15 years.

(2) Brake and tire wear particulate emissions (PM_{10}) are 0.013 g/km, but are not included here because these particles are much less toxic than tailpipe emissions.

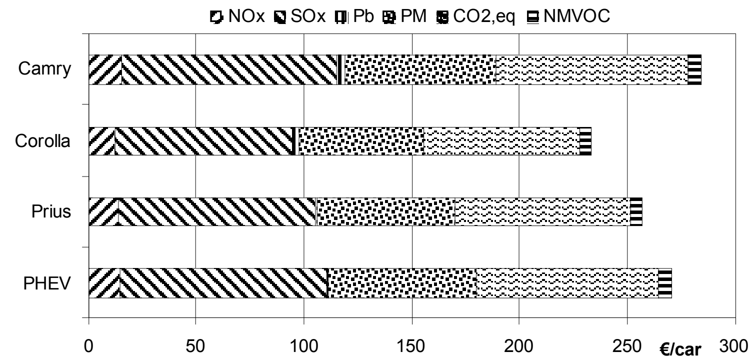

Figure 1: Damage cost due to the vehicle production stage.

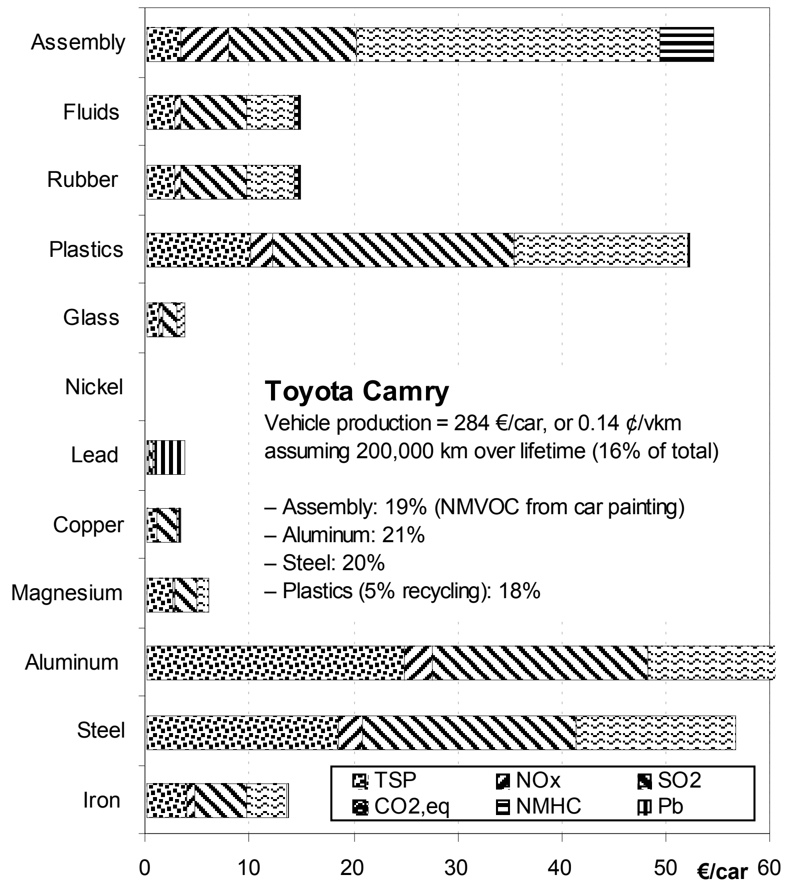

Figure 2. Contribution of materials and vehicle assembly to damage cost of vehicle production.

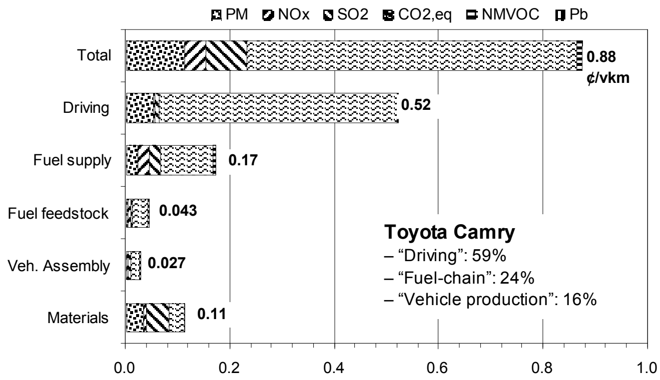

Figure 3. Contribution of the pollutants to the well-to-wheel and vehicle production stages.

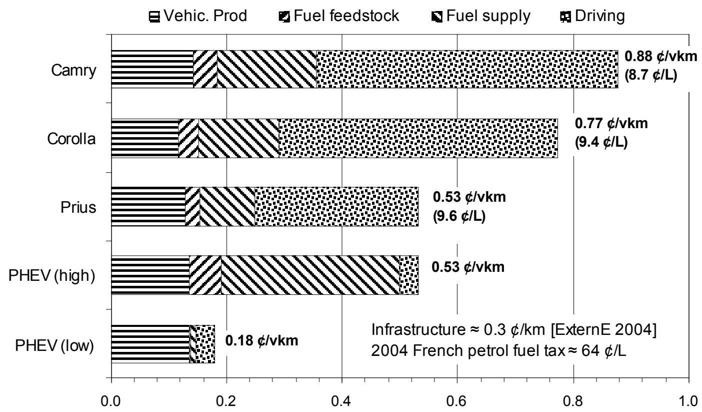

Figure 4: Total damage cost Eurocents/vkm.

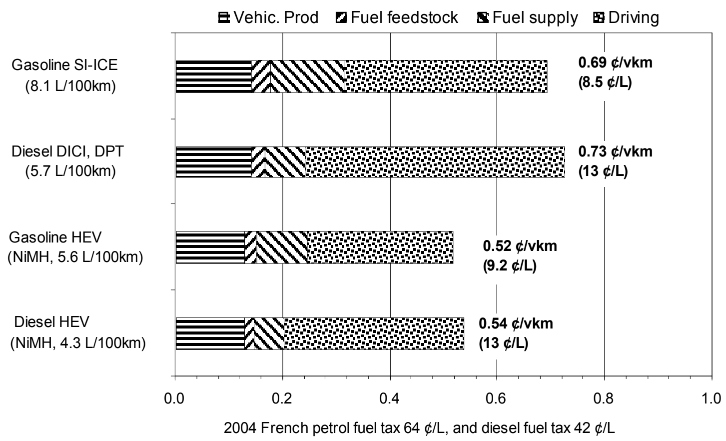

Figure 5: Damage costs of gasoline and diesel passenger cars in Europe (2010 time horizon).