European Union Action to Fight Environmental Crime

Qualitative and monetary analysis of the impacts of environmental crime: Overview

Deliverable 3.2a

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ABSTRACT

This report provides a summary of the conclusions of twelve case studies undertaken within the EFFACE project. The conclusions are structured around major themes of the analytical framework of the project.

The survey of data sources within earlier work within EFFACE showed that the data on environmental crime are usually highly dispersed with limited detailed data collations. The most likely sources of consolidated data are international institutions (such as Conventions and the EU). However, even here data are often limited. As a result it is not possible to provide a robust estimate of the overall impacts of environmental crime as a total figure. There are simply too many gaps for this to be done with any confidence. Therefore, it is important to focus on quantifying the impacts of environmental crime in areas where there are sufficient data for this to be done robustly and with confidence. As a result, the quantitative and monetary analysis has been undertaken for the following five subjects:

- The impacts of arson events
- The impacts of illegal wildlife trade in rhino and elephant
- The impacts of marine pollution
- The impacts of illegal WEEE shipments from the EU to China
- The impacts of illegal wildlife trade in Horsfieldii Tortoise

A common framework to guide data collection, analysis and presentation was agreed. This framework involved three analytical steps on the quantitative assessment of levels of illegal activity, the quantitative assessment of the impacts of that illegal activity and the economic valuation of the impacts of the illegal activity.

The results identified good examples of information that can be used to understand impacts of environmental crime. The most useful are good, coherent databases with information about the scale of illegal events (a fires database being a clear case). Another is the linking together of good data from different sources, such as that on illegal elephant and rhino poaching and that on population changes – thus enabling conclusions to be drawn on whether the criminal activity is affecting populations in the wild. Data from different types of sources can help paint a picture of different types of impacts (as seen with the waste shipment case).

The work had variable success in determining the quantitative impacts of environmental crime. Problems encountered in doing this have included:

- Barriers to determining what level of crime is occurring, where, trends, etc. In some cases there is poor recording of criminal events. However, in other cases it may be difficult to distinguish between legal and illegal activity.
- Information about impacts may prove difficult to move from anecdotal to quantitative.
- Where crime levels are known, the impacts from such crimes may be mixed with those from legal activities, so that distinguishing impacts is difficult.
- There is poor monitoring and recording of changes to environmental quality, health, etc., so that quantitative impacts of criminal activities are not known.

As a result, for much of the work here, assessments of quantitative impacts are limited to specific areas where there is sufficient data (or data of sufficient confidence) to provide those estimates.

The analyses in this study present different approaches to economic analysis. In some cases the quantitative information is insufficient to develop further economic assessments to any degree of confidence. Some used valuations of the natural environment (e.g. on natural capital loss due to poaching) or to health (e.g. for waste shipment). The fires case has sought to estimate the value of assets lost. Several cases have included information on the financial losses and benefits from those engaged in or affected by the illegal activity. In all cases, the economic analysis does not provide a total value for the impact of the type of environmental crime covered, but economic values for specific impacts.

ABBREVIATIONS

CITES	Convention on International Trade in Endangered Species
EA	Environment Agency
EFFACE	European Union Action to Fight Environmental Crime
EU	European Union
EUTR	European Union Timber Regulation
FTE	Full time equivalent
IQ	Intelligence quotient
IUU	Illegal, unreported and unregulated (fishing)
IWT	Illegal wildlife trade
MS	Member State
PAH	Polycyclic aromatic hydrocarbons
PBDE	Polybrominated diphenyl ethers
PCB	Polychlorinated Biphenol
PCDD/Fs	Polychlorinated dibenzo dioxins/furans
UNECE	United Nations Economic Commission for Europe
UNODC	United Nations Office on Drugs and Crime
WEEE	Waste Electrical and Electronic Equipment (Directive)
WP	Work Package

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1 Introduction

This deliverable is the conclusion of Tasks 2 and 3 of WP3 of EFFACE. The aim of WP3 is to understand the impacts (quantitative and monetary) of environmental crime. The purpose of the first task of WP3, Task 1, was to review and collect data on the extent and impact of different types of environmental crime. It did not produce estimates of the impact of environmental crime or otherwise quantify that impact. Rather it summarised the data sources available for different types of environmental crime and the type and extent of the data these sources contain. Task 2 was to draw on the available data and set out the quantitative impacts of environmental crime, while Task 3 was to set out the monetary impacts. This report describes the results of both the quantitative and monetary analysis. These are presented together as the monetary analysis often draws on the quantitative analysis.

The survey of data sources within Task 1 of WP3 of EFFACE showed that the data on environmental crime are usually highly dispersed with limited detailed data collations. The most likely sources of consolidated data are international institutions (such as Conventions and the EU). However, even here data are often limited. For many Conventions data collation is limited to those data reported by Parties and such data are often limited, of uncertain quality and with significant gaps. At EU level there has been limited data gathering on this issue (in contrast to other data sets on environmental quality and pressures).

While consolidated data sets are uncommon (e.g. that on fires is an exception), there are many examples of data on impacts in specific cases, such as for individual countries, individual instances, sites, etc. As a result it is not possible to provide a robust estimate of the overall impacts of environmental crime as a total figure. There are simply too many gaps for this to be done with any confidence. Even doing this for certain areas of environmental crime is problematic (particularly when the full range of potential impacts is considered). Therefore, it is important to focus on quantifying the impacts of environmental crime in areas where there are sufficient data for this to be done robustly and with confidence.

Following the examination of the data sets described in this deliverable, it was agreed that it would be most suitable for quantitative and economic analysis to be undertaken in selected areas given the availability of data. Thus the quantitative and monetary impact analysis of this WP should focus on those areas of environmental crime where there are likely to be sufficient data to perform such analysis and which could lead to robust conclusions. As a result, the quantitative and monetary analysis has been undertaken for the following five subjects:

- The impacts of arson events: D3.2b
- The impacts of Illegal wildlife trade in rhino and elephant: D3.2c
- The impacts of marine pollution: D3.2d
- The impacts of illegal WEEE shipments from the EU to China: D3.2e
- The impacts of illegal wildlife trade in Horsfieldii Tortoise: D3.2f

This overview report provides a short summary of the results of these analyses and some consolidating conclusions. The reader is referred to the more detailed analyses contained in these reports for further analysis.

2 Methodology

It was agreed that a common approach across the different subjects should be taken to help deliver results which were comparable. However, it is important to stress that the work undertaken in the project was neither the de novo collection of data on the impacts of specific types of environmental crime, nor the collection of specific economic data. Were the work to have involved the collection of such raw data, a detailed common methodology for data collection may have been applicable. Instead, the research has involved the collation and analysis of data already collected for other purposes. Thus there is not a common method for data collection itself.

Instead, a common framework to guide data collection, analysis and presentation was agreed. This framework involved three analytical steps:

1. Quantitative assessment of levels of illegal activity: e.g.
    - Numbers of incidents
    - Different types of illegal activity
    - Timespan – changes over time
    - Variability between countries, etc.

2. Quantitative assessment of the impacts of that illegal activity: e.g.
    - Area impacted
    - Species/ecosystems impacted
    - Health impacts
    - Financial impacts

3. Economic valuation of the impacts of the illegal activity:
    - Monetisation of specific impacts from Step 2 (noting that not all impacts may be amenable to monetary analysis)

The research would conclude with conclusions including overall quantitative and economic assessments of the impacts of the illegal activity examined and comments on how good the link is between crime data and impact data.

Each of the sub-reports of this report contains sections providing details of the methodology used in the data collection and analysis. Most studies report issues with data availability or ability to interpret data. This is particularly so in separating illegal from legal activities, understanding specific impacts, including valuation of economic impacts of crime (i.e. the impact on economic activities such as economic growth, tourism, sales of particular products or destruction of goods and services that have a market value) and, in turn, having confidence to undertake economic valuation of impacts (i.e. the monetary valuation of all impacts related to environmental crime and not only at economic impacts or not only at monetary consequences).

3 Summary of the quantitative impacts of environmental crime

3.1 Introduction

This chapter provides a summary of the findings of the studies undertaken within this part of the EFFACE research. The findings are presented according to the three major sections of the analytical framework:

- Quantitative assessment of levels of illegal activity

- Quantitative assessment of the impacts of that illegal activity
- Economic valuation of the impacts of the illegal activity

3.1 Quantitative assessment of levels of illegal activity

Each of the analyses within the WP3 EFFACE research had a specific scope of illegal activity it examined. For some (such as that on illegal hunting of elephants and rhino), the scope of illegal activity is defined by the range of the species and the type of activity. For the study on illegal waste shipment between the EU and China, the particular geographic relationship limits the scope. In contrast, the study on marine pollution is more open-ended.

Arson events

The study of the impact of arsons noted that in order to understand the level of illegal activity, it is important not only to know the cause of individual fires, but their location, severity, etc.. Without this additional information, analysis of impacts is not possible. To improve the availability of information and to support the fire prevention activity in the EU, the Joint Research Centre (JRC) and Directorate General for the Environment (DG ENV) of the European Commission have developed and implemented the European Forest Fire Information System (EFFIS). This provides comparative data on the number, extent and causes of fires. This is summarised in Table 1 below, which focuses on five southern MS and shows considerable variability between MS and over time.

Therefore, **given the broad heterogeneity of the fire events, the existing literature on the impact evaluation of fires typically focuses on a case-study approach**. This seems to be the most effective strategy to gather detailed information regarding different level of damages in order to develop regional fire management measure to minimize negative economic, social and environmental impacts of fires.

Table 1. Number of arsons **in five Southern Member States over the last decade**

	ITALY	SPAIN	PORTUGAL	FRANCE	GREECE	**TOTAL**
Number of Arsons						
2003	6720	10123	8101	554		**25818**
2004	4823	8402	6657	355	290	**20527**
2005	3422	7867	5210	562	231	**17292**
2006	4238	8723	3212	455	180	**16808**
2007	8384	15168	2997	1345	724	**28618**
2008	4250	12123	9990	567	423	**27353**
2009	3251	5423	4234	352	196	**13456**
2010	2475	6702	6455	398	320	**16350**
2011	5296	7093	4478	575	312	**17754**
2012	5246	7656	5069	870	823	**19664**

Average	**4810.5**	**8928.2**	**5640,3**	**603,3**	**381,9**	**20364.2**
Total	**48105**	**89282**	**56403**	**6033**	**3819**	**203642**

Illegal poaching of elephant and rhino

The study of illegal poaching of elephant and rhino was able to use good data from CITES monitoring and a number of other specialist monitoring studies. These provide data for overall illegal activity and well as data by country and data over time. For example, there is the African Elephant Database, which is maintained by the IUCN/ Species Survival Commission (SSC) and the African Elephant Specialist Group (AfESG) which has produced five reports to date (i.e. 1995, 1998, 2002, 2007 and provisionally in 2015). For rhino, the relatively low number of rhino and their earlier near extinction has led to meticulous monitoring and statistics. It is important to note that here the data are on animals killed, which closely matches illegal events. The findings for elephant are **Africa lost a total of 100,000 elephants to poaching, that could have provided legal income for African countries**.

The trends have been very different in different regions of Africa. While in South Africa nearly no population loss occurred, the population losses were substantial in Central Africa. For Rhino, the data for four countries show:

- South Africa: For 2006-2014 3,827 Rhinos were poached which reduced the overall population growth but did not lead to a reduced population. Only in 2013 and 2014 (more than a 1.000 animals per year) the overall poaching was close to the level where a population decrease could be expected.
- Namibia: Only five animals were poached from 2006-2012 which did not lead to any reduction in population.
- Kenya: 101 animals were poached between 2006-2012 which reduced the increase in population but did not lead to a population decrease.
- Zimbabwe: between 2006-2012, 378 animals were poached. The population decreased during that time by 67 animals or 8% of the population.

Marine pollution

The study on marine pollution examined data on incidents in Europe's regional seas. There are data on oil incidents, discharges, shipping incidents, etc.. However, it is difficult to identify specific instances of environmental crime as opposed to legal activities or accidents. Therefore, the research on quantification of marine pollution has been limited. The work includes information from all of Europe's regional seas, but comparative analysis is problematic due to the diversity of data sources.

E-waste

The e-waste study found that quantifying the illegal export of e-waste from the EU (to China) is especially challenging as there is very little clear information upon which estimates can be based. There are data on the amounts of e-waste generated in the EU and on the amounts imported into China and also estimates of overall illegal e-waste exported from the EU. Overall, **the study estimated that that for 2005 and 2012 respectively around 0.74 and 1.16 million tonnes of e-waste have been imported in China from the EU**. However, because of uncertainties in the data, the study also proposed a 'minimum China import scenario' and a 'maximum China import scenario'.

The e-waste study, therefore, provides an overall estimate of the quantity of illegally moved waste. It was, however, unable to quantify the number of specific illegal activities or actors. **The data do suggest an increase over time and this would reflect huge increases in available EEE in recent years in the EU** (thus reflecting increased opportunity). The study did not examine variability across the EU Member

States. This is problematic in any case as much e-waste is moved within the EU to major ports before shipment, so determining particular roles is highly problematic.

Illegal wildlife trade in Horsfieldii Tortoise

The study on the impacts of illegal wildlife trade in Horsfieldii Tortoise found it difficult to have confidence in understanding the levels of illegal activity as importation of this species into the EU is legal from farmed sources (so simply looking at import figures is not appropriate). There are three potential avenues of illegal activity:

1. **It is likely that many more individuals are involved in the trade than are actually reported in CITES data**.

2. It is suspected that the improper use of CITES labels that differentiate between wild and captive bred specimens, results in a much higher number of wild caught specimens existing in trade than the data reported would suggest.

3. The vast geography in which the species exists is compounded by the fact that several range states are not Party to CITES which makes it likely that many individual tortoises are illegally transported and smuggled through non-Party countries or countries with less stringent environmental and enforcement standards.

However, the study concluded that it is possible to use available information to make estimates about the proportion of illegal trade as compared to the legal trade. For example, one source suggested in 2000 that the annual illegal export was around 7,000 tortoises from Uzbekistan, 25,000 from Kazakhstan and 40,000 in total from Central Asian Countries, but the Uzbek government gave a much higher figure in 2007 of 35,000 not accounted for in the trade statistics. The study concluded that in Uzbekistan at least 50-75% of the Horsfieldiis labelled as captive bred are actually illegally wild caught, equivalent to 20,000 to 30,000 animals per year.

3.2 Quantitative assessment of the impacts of that illegal activity

Introduction

Arson events

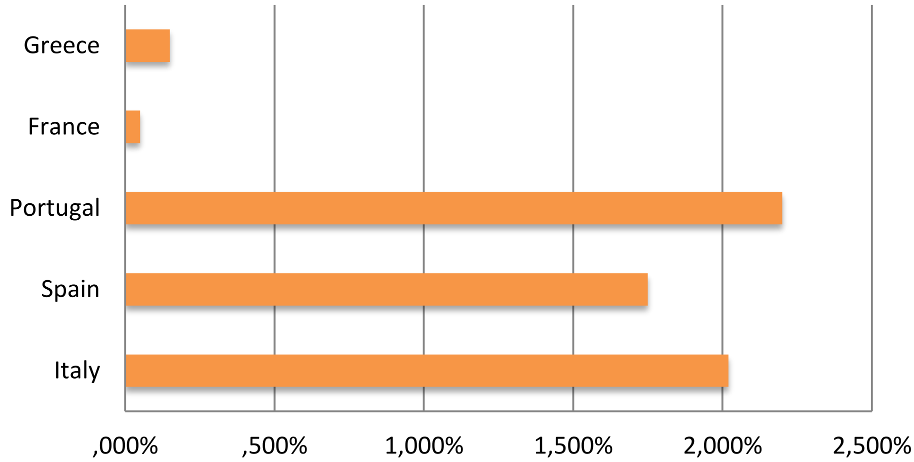
The extent of environmental, social, and economic impacts of arsons is a function of several factors such as the size, intensity, location and causes (deliberate and negligence) of the event. Thus the environmental impact was assessed using the following indicator:

$$\text{Environmental Impact (ha)} = \text{Average Burnt Area per fire (ha)} * \text{Number of arsons} \qquad (1)$$

This was further combined with the land use type (for example, forest vs. non forest) and for forest characteristics (i.e. protected areas, national parks and so on).

The following Figure 1 shows how much of five MS total surface has been burnt due to arsons during the last decade. Overall, Portugal has been the most affected country considering the ratio (2.02%) between burnt area (201,210.9 ha) and total surface area (9,209,000 ha), followed by Italy (2.02%), Spain (1.75%) and, to a lesser extent, by Greece (0.15%) and France (0.05%).

Fig. 1. The evironmental impact by country total surface



	Italy	Spain	Portugal	France	Greece
Burnt Area/Total Surface	2,020%	1,750%	2,200%	,050%	,150%

Source: authors' elaboration from European Fire Database

The environmental impact of arsons is particularly problematic for protected areas. Impacts include those on biodiversity, tourism, those living in and near the areas (property and life) and others. In Italy, in 2012, there were a total of 696 arsons that have covered about 11826 hectares of protected areas. During 2012, the most affected regions in terms of protected area burnt by arsons were Campania and Puglia.

The loss of human lives is the worst impact arising from forest fires. However, other issues concerning health are related to injury and pathologies affecting fire-fighters and people with respiratory problems. However, data limitations were found which meant that only deaths could be determined and these only in Italy. Between 2003-2012 the loss of human lives amount to 55 people and 442 injured. The most dramatic season was in 2007 with 23 deaths and 26 injured.

Illegal poaching of elephant and rhino

The information given above on levels of illegal activities was presented in terms of animals killed. Therefore, there is no need to repeat the figures here as these demonstrate quantitative impacts on those animals. In some cases the levels of loss of individual have become unsustainable for population maintenance. For elephants, in 2012, the killing rate was 7.4% compared to an average annual population growth for elephants of 5% (in the absence of illegal killing), which means that more animals are being killed than are being born. Thus, the criminal activity is reducing populations.

For rhino, from 1990 to 2007 poaching was limited and populations recovered. However, illegal poaching is increasing. The total population of white and black rhino in Africa increased by 17.5% between 2007 and 2012 with an average rate of population growth of 4.9% per. This growth rate decreased from 2010 to 2012 to 0.9% per annum. Thus poaching is not yet reducing rhino populations, but it is slowing recovery.

E-waste

The e-waste study found the illegal export of e-waste from the EU to China has resulted in the release of large amounts of contaminants in the local environment, such as heavy metals, PBDEs, PCDD/Fs, PCBs, CFCs and PAHs. It has caused among others high concentrations of heavy metals such as lead, cadmium, mercury, copper and zinc in the surrounding air, dust, soils, sediments and plants. The potential annual emissions of some environmental contaminants were estimated, e.g. it was for instance estimated that respectively 10 and 16 tonnes of PCBs from EU e-waste were potentially released in the Chinese environment in 2005 and 2012. Given the complexity of the e-waste problem, the report presents only select data as to the environmental impacts of the e-waste crime. The focus was mostly on the environmental impacts (i.e. pollution levels) related to heavy metals and lead and zinc in particular. And mostly studies investigating or results regarding adverse environmental impacts in Guiyi and Taizhou, as the most representative locations for informal e-waste recycling, are being referred to.

The environmental impacts lead to economic losses and additional costs. For example, due to local contamination of soils and water resources, drinking water needs to be brought in from other regions. It is estimated that in Guiyu, with a population of about 150,000 in the year 2013, establishing a piped water supply resulted in annual additional cost of around €1.6 million.

The e-waste study found that illegal exports from the EU are resulting in increasing incidences of chronic disease in China, threatening not just workers but also current residents of e-waste recycling areas and adjacent regions and future generations. Illegal exports from the EU result (through the informal recycling and dumping) in high prevalence of skin, gastric, respiratory, hematic, neurological, prenatal, natal and infant diseases in China. Select scientific studies (in China) show associations between exposure to e-waste and physical health outcomes such as:

- decreased lung function (i.e. lower forced vital capacity);
- decreased physical growth of children (i.e. lower weight, height and body-mass index);
- reduced reproductive health (i.e. increases in spontaneous abortions, stillbirths, and premature births, and reduced birth weights and birth lengths);
- changes in cellular expression and function (i.e. increased DNA damage).

Negative associations were also shown for blood lead levels and IQ in children. Studies of the impacts in these areas of China do not address local biodiversity impacts, if any.

For China as a whole it is conservatively estimated that around 81,300 children (58,000-93,000) born in the period 1995-2013 have been affected in their neurological development as a result of e-waste exposure. It was subsequently estimated that these children in China lost about 97,560 IQ points (69,600-111,600) as a result of informal e-waste recycling and dumping activities. This amounts to an average reduction of intelligence of 1.2 points per child.

Illegal wildlife trade in Horsfieldii Tortoise

The study on the impacts of illegal wildlife trade in Horsfieldii Tortoise noted that data limitations on population and replacement rate are a significant obstacle to calculate efforts the rate of extinction and sustainability of current trade levels. Horsfieldii mature slowly and have modest reproduction capacities and so are particularly susceptible to collapse in the presence of illegal trade. Moreover, it is estimated that 95% of Horsfieldiis that enter the pet trade die within a year, thus harvest rates may be significantly higher than those corresponding to that which is documented in CITES trade data. A further problem is that information on population density is limited and outdated, partly due the species' extensive range and reclusive habits and also to its perceived insignificance compared to a flagship or keystone species such as a panda, rhino or elephant. Having said this, the study did find evidence of significant declines in population levels in China, Kazakhstan and Uzbekistan and cases where the species had disappeared altogether. Illegal capture and trade is likely to be a major contributing factor, but other factors such as land-use change cannot be ruled out.

It is also worth noting that the value of the Horsfieldii tortoise for the overall ecosystem cannot be valued properly. It is not a keystone species and particular existence on the Central Asian steppes is not well understood in terms of its role within this ecosystem. The study did look into existing estimates for the value of the turtles and tortoises to ecosystems more generally but found there to be few studies and not applicable to the case.

3.3 Economic valuation of the impacts of the illegal activity

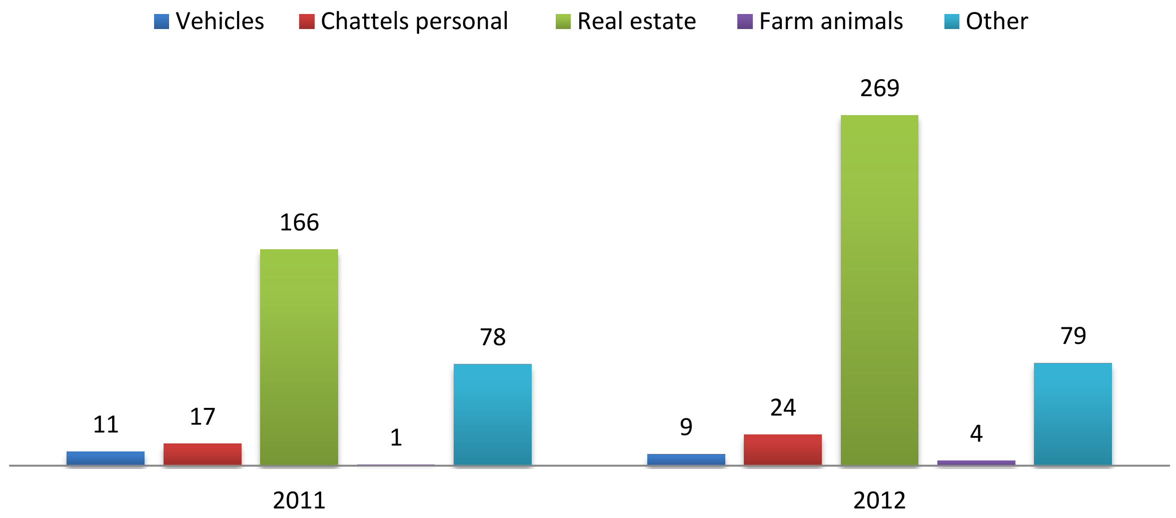
Introduction

This section summarises the findings on the economic valuation of the impacts of the illegal activities studied. It is important to note that for one study area (marine pollution), this proved problematic to determine and for the other areas there were issues for which there were insufficient data to perform an economic analysis. This was typically the case for biodiversity impacts, but also explains why some areas (e.g. tourism) could be analysed for some types of crime (e.g. poaching of elephants), but not others (e.g. arson events).

Arson events

The total impact of fires includes fire extinguishing costs, environmental damage and external damage to physical assets (e.g. infrastructures, building, etc.) in terms of reconstruction costs, etc. The study on arsons focused the economic analysis on direct costs associated with fire fighting and the loss of income due to the destruction of human physical assets in the surrounding areas. To quantify the material impact of fires based on the factors which have been recorded (e.g. there is not an estimated of losses to tourism, etc.), different categories of asset were considered: vehicles, material assets, real estate and farm animals. These economic impacts are summarised in figure 2 below (chattels being personal possessions).

Fig. 2. Economic and Material Impacts (2011-2012)



Illegal poaching of elephant and rhino

The study of the impacts of illegal poaching on elephant and rhino looked at the economic impact through two aspects of income provided by the ecosystem with elephants and rhinos:

- If the poaching does not lead to reduced numbers of the species, the societal loss is valued by estimating the alternative legal income that the host society could reap from the animals through tourism income, if they would not be poached.
- If the poaching reaches a level that leads to a reduction of the population, the loss is valued as a loss of natural capital. The wildlife is the wealth of the source countries on which basis they can attract wildlife tourism and the associated annual income from it.

The following tables 2-3 summarise these economic impacts for each type of animal. Overall the poaching of rhinos and elephants causes significant damage to African economies both by taking away legal present income opportunities for African economies but also by reducing the natural capital on which all future income opportunities are based.

Table 2. Economic value lost due to elephant poaching

	Africa
Total population of Elephants in Africa 2010	500,000
Number of elephants poached 2010-2012	100,000
Lost potential legal income per Elephant	€22,331 - €31,264
Total loss of potential legal income 2010-2012	**€ 2.23 billion – € 3.12 billion**
Total loss of population 2010-2012	25.000 (5% of population)
Value of 1% population loss	€ 2,4 billion to € 3,6 billion
Total loss of natural capital 2010-2012	**€ 12 billion to € 18 billion**
Total economic loss per year	**€ 4,7 billion to € 7 billion**

Table 3. Economic value lost due to rhino poaching

	South Africa	**Namibia**	**Kenya**	**Zimbabwe**
Total population of rhinos 2012	20.954	2214 (2010)	914	792
Number of rhinos poached 2006-2014	3.827	5 (2006-2011)	101 (2006-2012)	378 (2006-2012)
Lost potential legal income per rhinos	€ 312.640	€ 312.640	€ 312.640	€ 312.640
Total loss of potential legal income per year	**€133 million**	**€0.26 million**	**€4.5 million**	**€16.9 million**
Total loss of population 2010-2012	0	0	0	67 (8%)
Value of 1% population loss	€790-1,180 million	€37- 56 million	€150 - 230 million	€45- 68 million
Total loss of natural capital 2006-2012	0	0	0	€360-544 million
Total loss of natural capital per year	**0**	**0**	**0**	**€51-76 million**
Total economic loss per year	**€133 million**	**€0.26 million**	**€4.5 million**	**€68 - 93 million**

The economic losses caused by rhino poaching are much smaller than the losses caused by elephants mainly due to the much higher occurrence of elephant poaching and, except for Zimbabwe, rhino poaching does not yet exceed the natural growth of population. The estimates only cover a small part of the overall societal costs of rhino and elephant poaching. The illegal trade does cause other costs which cannot be valued.

E-waste

The e-waste study estimated that the 2.98 million tonnes of illegally exported e-waste from the EU in 2012 correspond roughly with €31.2 million to €37.5 million loss in income to the EU e-waste recycling industry. If one looks at the e-waste exports to China only (1.16 million tonnes in 2012), the EU recycling industry is estimated to have lost €12.2 million to €14.6 million in profits in 2012. Assuming that the average intrinsic value of WEEE (i.e. income available to the recycling business) is about €300 per tonne, the economic value lost to the EU as a result of illegal exports to China is roughly estimated at €348 million for 2012 only. The economic value lost to the EU as a result of all illegal exports out of the EU is estimated at €892 million for 2012.

As to the impact on jobs, the illegal export of e-waste from the EU in 2012 is estimated to represent a potential loss of about 38,000 FTE recycling jobs in the EU. Assuming a typical multiplier of 2, these direct recycling jobs would result in another 38,000 indirect and induced jobs, for a total of 76,000 jobs. The illegal export to China in particular is estimated to represent a potential loss of circa 14,900 FTE jobs in the industry and another 14,900 indirect and induced jobs, for a total of 29,800 jobs. A loss of 14,900 FTE jobs goes along with an estimated loss of economic value added of around €780 million. Though this figure needs to be treated with caution due to data availability and quality issues, it is indicative of the significance of losses in economic terms. It should also be noted that the assessment of FTE jobs lost does not mean a total net loss of jobs in society as some people will have alternative jobs available.

Table 4 provides an overview of the estimated economic impacts for the EU.

Table 4: Overview of estimated economic impacts in the EU for 2012

Loss in profits for the EU recycling industry	Arising from illegal EU exports to China		€ 12.2m - € 14.6m	
	Arising from total illegal EU exports		€ 31.2m - € 37.5m	
Lost economic value to the EU	Arising from illegal EU exports to China		€ 348m	
	Arising from total illegal EU exports		€ 892m	
Potential job loss in the EU (FTE)	Arising from illegal EU exports to China	*Direct jobs*	14,900	29,800
		Indirect and induced jobs	14,900	
	Arising from total illegal EU exports	*Direct jobs*	38,000	76,000
		Indirect and induced jobs	38,000	

Some of the health impacts in China arising from illegal e-waste shipments (and informal recycling and dumping in particular) have direct economic costs and others can usefully be represented by economic or monetary values to help communicate the importance of preventive and remedial action. A monetary valuation of the impacts on children's IQ might include an assessment of: opportunity costs in terms of *lost productivity* (i.e. decreased current value of expected lifetime revenues); *direct resource educational costs* related with compensatory education; opportunity costs of *lost income during remedial compensatory education*; *medical treatment costs*; and, *disutility* resulting from human development disabilities. Given time and resource constraints and limited availability of or access to Chinese data, it was not possible within the context of this project to estimate or calculate these direct costs and opportunity costs.

Illegal wildlife trade in Horsfieldii Tortoise

The study on the impacts of illegal wildlife trade in Horsfieldii Tortoise was unable to monetise impacts of illegal trade in wild population due to difficulties in determining changes to those populations, the cause of changes (i.e. the contributing factor of illegal trade) and the role of the species in the ecosystem (as noted above). There is certainly an economic value from the trade. However, while a *Horsfieldii* tortoise is sold as a pet for between $25 and $100 USD, the price paid to exporters/collectors in source countries was estimated in 1997 at €0.45 per individual. It is clear that the majority of the earnings stay with the importing country and pet dealership. Thus the overall value of the pet trade for the source countries is small. Overall countries are exporting around 80,000 live animals and this provides an overall value of less than € 40,000.

4 Conclusions

4.1 Introduction

The case studies make a series of conclusions and recommendations. Many of these are highly specific to the individual case study. However, others have wider consequences to addressing environmental crime and it is these which are summarised here. Given the focus of the case studies it is not surprising that many of recommendations are addressed to EU level institutions.

This conclusions section begins by examining some of the lessons learned from undertaking the work.

4.2 Limitations on assessing quantitative impacts

The work undertaken in this study has had variable success in determining the quantitative impacts of environmental crime. Problems encountered in doing this have included:

- Barriers to determining what level of crime is occurring, where, trends, etc. In some cases there is poor recording of criminal events. However, in other cases it may be difficult to distinguish between legal and illegal activity.
- Information about impacts may prove difficult to move from anecdotal to quantitative.
- Where crime levels are known, the impacts from such crimes may be mixed with those from legal activities, so that distinguishing impacts is difficult.
- There is poor monitoring and recording of changes to environmental quality, health, etc., so that quantitative impacts of criminal activities are not known.

As a result, for much of the work here, assessments of quantitative impacts are limited to specific areas where there is sufficient data (or data of sufficient confidence) to provide those estimates.

It is important to note, however, that the work has also identified good examples of information that can be used to understand impacts of environmental crime. The most useful are good, coherent databases with information about the scale of illegal events – the fires database being a clear case. Another is the linking together of good data from different sources, such as that on illegal elephant and rhino poaching and that on population changes – thus enabling conclusions to be drawn on whether the criminal activity is affecting populations in the wild. Data from different types of sources can help paint a picture of different types of impacts (as seen with the waste shipment case).

However, overall while the work has identified a wide range of different quantitative impacts of environmental crime, there are frustrations in working towards any systematic overview of such impacts.

4.3 Impacts are a pyramid

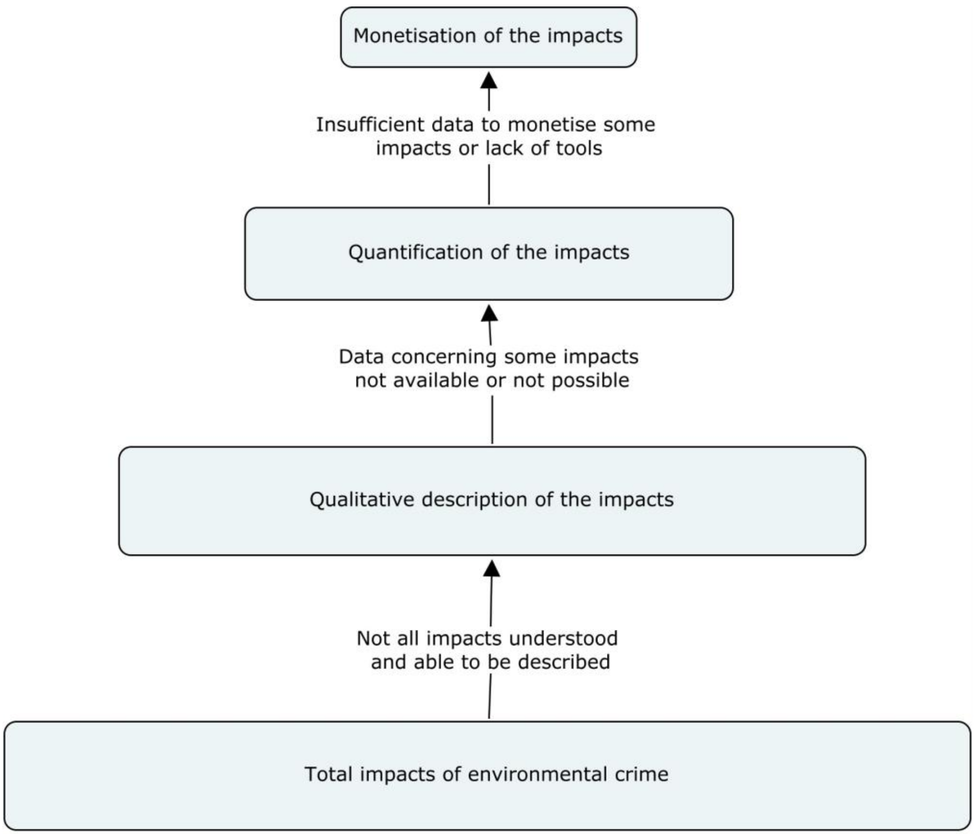
Environmental crime has many different impacts. The analysis supporting this overview report has examined quantitative and monetary impacts. However, an important lesson from the examination of the impacts of environmental crime is to recognise that there are different degrees to which impacts can be described and analysed and different ways in which these can be presented.

This is illustrated by the following figure. An environmental crime (individual or collective) has a range of different impacts. These are understood to different extents and it is likely that some will not be known. Therefore, only a proportion of these can be described in a qualitative way. Of those impacts that for which qualitative descriptions are possible, only a sub-set of these can be quantified. For the others there may be insufficient information (e.g. because of lack of monitoring, difficulty in collecting data in a criminal environment, problems in linking cause and effect, etc.) to provide numbers to impact events. Finally, only a sub-set of the quantified impacts will be able to be monetised, again due to data limitations as well as possible methodological limitations for specific types of impacts.

For environmental crime, there is even perhaps an additional lower layer to the pyramid – where total impacts are included, whether legal or illegal. For some of our work this is not an issue (e.g. elephant poaching is readily distinguished, or fires where causes are recorded), but for some areas such as marine impacts, identifying the illegal component is problematic.

This deliverable focuses on the quantitative and monetary impacts, but it is important that all impacts, even if they can only be qualitatively described, should be communicated to the public. Only in this way can a full picture be presented. Quantification is important, e.g. to communicate scale of impact, and monetisation enables the impacts to be considered within wider economic contexts. However, the biggest impact might be the one that has not been quantified and not monetised.

Figure 3. A schematic overview of understanding impacts as a pyramid



4.4 Scoping the impacts of environmental crime

An important lesson from the work on impacts on environmental crime in EFFACE has been the challenge of scoping those impacts. Most environmental crimes have an immediate focus for understanding the impact – that which is the crime itself. For example, the loss of animals protected under CITES can be determined, timber illegal felled can be estimated, quantities of waste illegally shipped can be analysed. However, these first order impacts are only the start of understanding the impacts of those crimes.

Figure 4 provides a generalised overview of the different types of impacts that might occur in considering a specific criminal activity. It is not meant to be comprehensive, but illustrative of the range of impacts that might occur.

The figure explores different types of direct environmental impacts. These includes impacts on environmental quality (which should be considered across air, water, soil quality) and direct impacts on biodiversity (e.g. hunting on protected species populations). However, it is also important to ensure indirect impacts are included, e.g. impacts on environmental quality through pollution causing impacts of biodiversity.

There may also be impacts on health (short and long term), caused by changes in environmental quality. Resources might be impacted (e.g. loss of useable soil, depletion of forest resources, etc.). Public administrations are also impacted, not least the diversion of funds to fight crime. Criminal activity may also negatively affect the financial interests of legitimate economic actors. Finally, the full scope of impacts should include positive impacts, such as economic benefits from the criminal business[1].

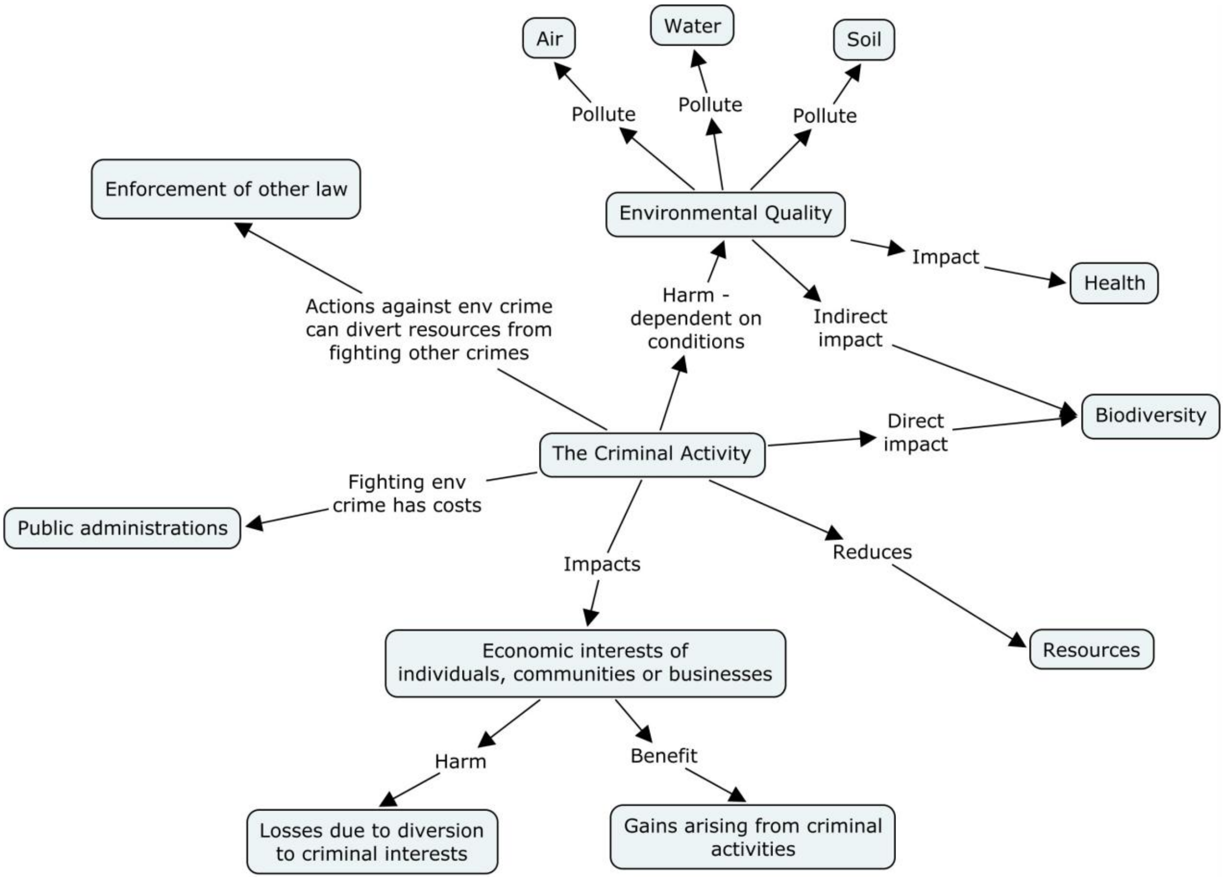
It is important to note that each of these elements of scoping can, themselves, contain many elements. It might be possible to identify a change in air quality causing a health impact, but other air pollution changes might cause further health impacts and these need to be acknowledged in any scoping to ensure a comprehensive picture is provided.

Overall, this scoping exercise is important to ensure all impacts are accounted for. It may not be possible to follow-up all impacts in further analysis, but the acknowledgement of a (potential) impact ensures they are not ignored, which can be important in communicating results to policy makers and the public.

The cases studied in this work present different approaches to the scoping of impacts. That for waste shipment, for example, has sought to identifying different types of impacts in order to then focus on those for which further analysis is appropriate. That for fires recognises the range of land-use, property and health impacts that occur, while that on illegal trade in Horsfieldii Tortoise notes the limited impacts that occur in that instance.

[1] This was noted in the scoping work within Task 1 of this WP, where studies have found waste electrical equipment illegal shipped to west Africa can have enter the local economy and have positive economic consequences.

Figure 4. A schematic illustration of scoping of the different types of impacts of environmental crime



4.5 Identifying an impact

The first scoping of impacts concerns the direct impacts on the environment. In the figure these are divided between environmental quality and impacts on biodiversity. Although biodiversity is an aspect of environmental quality, separating air, water and land quality is useful as these can also lead to health impacts. However, while these immediate environmental impacts may arise from an environmental crime, whether an change in air, water or soil quality causes harm will depend on a range of factors such as pre-existing conditions, nature of organisms dependent on the medium, etc. Further, where there are environmental quality thresholds of concern some degradation in quality resulting from a crime might not breach that threshold. However, care needs to be taken in understanding the nature of 'thresholds', particularly where these are in law. Some might be a real threshold beyond which an impact starts to appear. However, EU air limit values, for example, are not thresholds of impact and changes in air quality below these values has positive or negative health effects.

Even if one argues that change is quality is inherently undesirable, it is still necessary to scope the context of that change as this will certainly affect any attempt to quantify the impact after scoping is complete.

Impacts on biodiversity may be complex and may either be indirect (e.g. pollution affecting individuals and populations) or direct (e.g. from illegal hunting). The relationship for direct impacts on individuals may be relatively easy to conceptualise (not necessarily determine), but indirect impacts may be much harder to describe. A link between change in quality and biodiversity impact may be known, but that link may not be clear.

A second issue in scoping biodiversity impacts is the link between impacts on individuals and impacts on populations. Environment crimes might result in the death of individuals, but does this have any impact on populations[2]? This issue is explored in the analysis on hunting of elephants and rhinos, where it is clear that hunting levels are beyond those consistent with sustainable populations. Overall it would be expected that species that are slow reproducing would be at greatest risk from the removal of individuals from a population.

4.6 Relationships between drivers, pressures and impacts

Understanding the link between drivers, impacts and pressures is important in understanding impacts, particularly as a key driver for much environmental crime is economic. Therefore, these economic outcomes for the criminal may need to be included in an impact analysis. Understanding these interactions is also important in other contexts, such as in helping to direct more effective law enforcement, but that is a subject for other analyses.

The link between drivers, pressures and impacts might not always be clear. For many times of environmental crime this is often not an issue for immediate pressures. The loss of individual elephants due to hunting is relatively precisely documented. Similarly, local toxic contamination due to recycling activities from illegal imported electronic waste can be directly linked to that criminal activity. In other cases the exact relationship with a pressure is more difficult to determine, particularly if environmental crime is contributing to a problem, but is not the only cause. Illegal waste dumping in Europe removes the benefit of potential resources from the recycling system, but so do other activities.

4.7 Approaches to economic analysis

The analyses in this study present different approaches to economic analysis. In some cases the quantitative information is unsufficient to develop further economic assessments to any degree of confidence. Some have used valuations of the natural environment (e.g. on natural capital loss due to poaching) or to health (e.g. for waste shipment). The fires case has sought to estimate the value of assets lost. Several cases have included information on the financial losses and benefits from those engaged in or affected by the illegal activity.

It is important to note that this work did not include de novo collection of data and, therefore, could not be designed to fit into the most appropriate economic analysis tool for the issue being addressed. As a result, there is further work to be undertaken in this Work Package. This work will, therefore, continue by examining the range of different economic analysis tools available and determining which are most appropriate in examining different types of impact from environmental crime.

4.8 Use of data on impacts

Of course, the design of scoping for information on impacts and the effort into gathering such information is dependent on why such information is needed. Data on the impacts of environmental crime are needed for a variety of reasons, including the following:

- To help target actions, resources, etc., of enforcement bodies to target enforcement actions.
- To help understand impacts on victims and so guide attention to liability and restoration.
- To design welfare maximizing sanctions that internalize the external effects associated with environmental crime.
- To help guide policy review and policy development.

[2] Some might argue about the ethical issue of killing of individuals, but this would take us into the realm of animal rights and comparing the death of an elephant, for example, with millions of cattle every day. This is not part of this analysis.

Environmental enforcement bodies are usually resource constrained and, therefore, it is important that these resources are used most efficiently. One criterion to target those resources is to focus them on where the impacts of environmental crime are greatest or most severe. This might not be the same as numbers/levels of crime, which is also a legitimate criterion for targeting resources. However, understanding the extent of impacts and the limitations of such understanding is important in guiding enforcement strategies. Further, if data on impacts can be compared with the results of the application of different enforcement approaches, then such data can be important in guiding the development of enforcement strategies and the development of smart instrument mixes for tackling environmental crime.

Information on impacts is important for actions to be taken once those impacts have occurred. Where the offender is identifiable, then liability rules may apply and impact data can be used to determine the extent of liability. Such information, therefore, helps to empower victims (where these can be identified as some environmental crime can be viewed as 'victimless') by providing solid evidence. Information on impacts is also important to guide restoration initiatives, including helping to cost those initiatives.

The research within other areas of EFFACE has examined the suit of legislation and policies on environmental crime in different contexts (EU, Member State, International). There is much debate on whether these policies are well designed. However, in order to improve these policies, it is important to have evidence of their effectiveness and their efficiency. Information on impacts is an important part of this evidence (along with other types of evidence). Policies should be leading to reductions in impacts and these should, ideally, be focused where those impacts are most severe while taking into account the costs of the policy/enforcement measures. However, is there evidence that this is the case?

It can, therefore, be seen that there are many challenges in gathering qualitative, quantitative and economic data on the impacts of environmental crime. This is due to the wide range of many different types of impacts, the complexity of criminal activity and methodological challenges. However, the gathering of such information is important to help target enforcement activity and improve environmental legislation. Thus further effort is needed to improve the gathering of impact information. This should build on the strengths identified in this research and address the weaknesses. Identifying the strengths, weaknesses, opportunities and threats regarding data and information is undertaken in later work within the EFFACE project, which will lead to identification of specific policy recommendations on this issue.

The Quantitative and Monetary Impacts of Forest Fire Crimes

WP3 Quantitative Analysis

Deliverable 3.2 b

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DISCLAIMER

ABSTRACT

Wildfires have been recognized as one of the most significant environmental threats in Europe and particularly in the Mediterranean regions, giving rise to a multitude of environmental, social and economic impacts. However, forest fires are often not the result of a natural disaster or causes of fatality, but rather an anthropogenic phenomenon with direct dependence on social behavior, both voluntary and involuntary. Therefore the willful and malicious burning of forests represents a fire crime. This report focuses on the current status of wildfire impacts at the European and Italian regional levels, looking mainly at fire crimes in both quantitative and monetary terms and, to the same extent, at the relation between geographical distribution of such crimes and organized crime.

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LIST OF ABBREVIATIONS

WP	Wood Production
AAM	Analytical Assessment Model
AOT	Aerosol Optical Thickness
DGs	Directorates General
DG ENV	Directorates General for Environment
EFFIS	European Forest Fire Information System
EFCC	European Fire Causes Classifications
JRC	Joint Research Center
NIAB	Nucleo Investigation Anti Bosch
NWP	Non-wood Products
PCC	Protection from Climate Change
QAM	Quick Assessment Model
TEV	Total Economic Value
WP	Wood Production

1 Introduction

Wildfires have been recognized as one of the most significant environmental threats in Europe and, particularly, in the Mediterranean regions (Requardt et al., 2007). According to the data gathered by the European Forest Fire Information System (EFFIS), the number of wildfires has moderately increased in recent years. The annual amount of burnt area fluctuates with no well-defined trend, although critical fire seasons in recent years have heavily impacted Mediterranean countries. Consequently, it is not surprising that during the last decade wildfires have gained more attention and increased concern about their environmental and economic effects and the loss of human lives. In Europe, the increasing awareness of the gravity of the problem was triggered by large-scale wildfires and their effects in southern European countries. Most of the total burnt area in Europe concentrates in this region where approximately 35,000 events occur per year (JRC, 2009). The huge number of forest fires and their systematic geographical location clearly show that they are not only an environmental issue, but have also significant social dimensions, affecting millions of people, by having major economic impacts and causing significant human casualties (Gonzalez Caban, 2007).

According to the National Wildfire Coordinating Group (2012), wildfires can be defined as uncontrolled fires caused naturally or by humans with susceptibility to expand into forested areas, arboreal or bushy, including any structures and infrastructures placed within such areas, or on cultivated or uncultivated land and pastures adjacent to these areas. However, forest fires are often not the result of a natural disaster or causes of fatality, but rather an anthropogenic phenomenon, with an exclusive, direct dependence on social behavior, both voluntary and involuntary (Leone and Lovreglio, 2003). Fire crime is defined as the willful and malicious burning of forests. It may be distinguished from other causes such as spontaneous fires and natural wildfires.

The European Union does not yet have a common forest fire policy for tackling fire crime, but it is involved in the development and monitoring of measures in the field of information, prevention and restoration of burned surfaces through various Directorates General (DGs)[1].

[1] DG Environment (forest fires prevention and monitoring, LIFE+ Regulation (EC) No 614/2007) and DG Agriculture and Rural Development (for supporting the restoration and fire prevention activities, Regulation (EC) No 1968/2005).

However, intentional and deliberate fire has recently been recognized as an environmental crime and has been included in the legal code of some Member States (i.e. Spain, Italy). For instance, in Italy, fire crimes are subject to prosecutions in a court of law. Particularly, in the Italian legal system, the criminal protection of the environment is almost entirely left to a series of misdemeanors, which fall outside the Penal Code. Among the few cases that do include the felony, there is art. 423-bis CC, which punishes anyone who causes a fire in forests or woods with four to ten years imprisonment. If the fire is caused by negligence, the punishment shall be imprisonment from one to five years. The penalties shall be increased if the fire endangers buildings or harms protected areas; the penalties shall be increased by half if the fire caused serious, widespread and persistent environmental damage. Also, it should be noted that Italy represents a quite important country study because of the strong presence of *organized crime systems*: the Legambiente Report (2014) argued that more than 51% of fire crime areas were concentrated in the four Italian regions where the presence of mafia clans is highest.

Policy and fire management measures at different levels are being developed and implemented to minimize the negative economic, social and environmental impacts of wildfires. The application of policy measures needs significant investments in terms of financial, human and organizational resources, which have to be justified and effective (Mavsar et al., 2011). The assessment of the use of financial resources in wildfire related measures (e.g. deterrence from fire crime) is crucial to justify and to select the most effective alternative in order to stem the different impacts of wildfires as much as possible (Mavsar et al., 2011). Lack of information on the environmental, social and economic damages arising from fire crime limits decision-making and investments for the prevention and the fight against wildfires (Nautiyal and Doan, 1974) and could lead to the misallocation of financial and organizational resources.

1.1 Some difficulties in evaluating forest crime impacts

The analysis of the impact of fire crimes in both quantitative and monetary terms presents several difficulties relating to both methodological and data availability issues. From the methodological point of view, the impact evaluation of fires has to take into consideration the fact that each fire produces several impacts (environmental, health, economic and social), which are very specific to the particular area where it occurs. Moreover, it is worth noting that fires directly impact benefits and resources that people receive from the environment, including for instance, provision of food, water and fiber, regulation of floods, drought, and land degradation (nutrient cycling). However, only a minor part of them exhibit a market price that can be used as a possible proxy for assessing their value, while the majority of goods and services is not marketed. Statistics on fires quite often do not distinguish between natural and intentional causes of fires (fire crimes). As we have seen in deliverable D.3.1., to improve the availability of information and to support the fire prevention activity in the EU, the Joint Research Centre (JRC) and Directorate General for the Environment (DG ENV) of the European Commission have developed and implemented the European Forest Fire Information System (EFFIS). However, although EFFIS represents a useful and effective effort to collect data in a harmonized way among the Member States, it lacks adequate indicators to measure the economic, social and health impact of forest fire. To better cope with the lack of data and related indicators on the impact of fires, the JRC decided to develop two operational models for the evaluation of the social and economic impacts of fires (Mavsar et al., 2011). The first one (*Quick Assessment Model, QAM)* is used for small fire events (i.e., those mapped in the EFFIS database > 40 ha) and is designed to evaluate the fire damage cost using the reconstruction cost approach. The second model *(Analytical Assessment Model, AAM)* is conceived for the evaluation of the fire damage cost of large fire events (e.g., fire size > 500 ha) and is based on the economic analysis of the missed flow of forest goods and services. However, despite the general approach of both models to take advantage of the information available in the EFFIS database, they nonetheless seem to be quite limited in: (i) understanding other types of impacts (e.g. social and health); (ii) considering fire events below 40 ha of burnt area; and (iii) analyzing fire crime since no fire causes are taken into account. Therefore, given the broad heterogeneity of the fire events, the existing literature on the impact evaluation of fires typically focuses on a case-study approach. This seems to be the most effective strategy to gather detailed information regarding different levels of damage in order to develop regional fire management measures to minimize negative economic, social and environmental impacts of fires.

1.2 A brief survey of the literature

The literature dealing with the issue of fire events is, essentially, described by two phases: a pioneer one, during which there were very few studies regarding this topic, and a more recent phase where the wildfire criticalities were faced with detailed approaches and techniques. The pioneer phase publications discussed fire without thoroughly studying its different impacts on the ecosystems. For example, Ducamp (1932) and Laurent (1937) tried to explain the effects of wildfires on vegetation, considering their impact on erosion and forest degradation. While, during the more recent phase, most scientists, under the influence of non-neoclassical social science ideas, were anxious to analytically demonstrate the different dimensions of wildfire impacts providing methodologies for the evaluation of forest fire risks and additionally policy implications. Therefore, it was only about 30 years ago that the subject was first objectively approached. Presently, several researchers have started to analyze the Mediterranean countries and the results give new perspectives on environmental, economic, social, and health impacts. Moreover, during the last decade, several meetings and conferences have also brought new knowledge on wildfire effects, considering also aspects such as post-fire vegetation, recovery costs, loss of infrastructure, safety issues for fire-fighters and for those who live and work in affected areas. As underlined, the existing literature on wildfire impacts mostly relies on case study methodology to assess short and long-term impacts and focuses on the nature and extent of impacts on wild-land in order to incorporate this information into wildfire policies, risk assessments, and management practices. Table 1 below summarizes some empirical contributions aiming at quantifying wildfire impacts:

Table 1: Empirical contributions on wildfires impacts

Author(s), Years	*Research Question*	*Main Argument*
Morton et al., 2003	Assessing the environmental, social and economic impacts of wildfire in U.S.	Total environmental, social, and economic impacts are a function of multiple factors, including fire size, location, and burn intensity.
Ortuno-Perez et. al., 2004	Assessing socio-economic deterioration that large forest fires have produced in the municipalities of the Tietar Valley (Spain).	Most affected socio-economic indicator was the number of holiday homes, which led authors to conclude that tourism is the most affected industry by forest fires.
Ciancio et al., 2007	Methodological framework for the economic evaluation of specific impacts arising from wildfire.	Total economic damage is divided into three components: (i) *fire extinguishing costs*; (ii) *environmental damage*; and (iii) *external damage*.
Valese et al., 2011	Quantify the economic impact arising from forest fires in North Eastern Italy (Veneto) in 2006, 2007, and 2008.	On average, every year in Veneto, the economic cost of wildfire is about 50 million Euros (i.e. every Veneto citizen pays about 10 Euros per capita due to forest fires).
Barnaba et al., 2011	Estimating wildfires' health impacts by looking at their contribution to the European aerosol optical thickness (AOT).	Fire-related aerosols play a major role in shaping the AOT yearly cycle at the continental scale.

To our knowledge, the only contribution in the literature that, following a holistic approach, tries to simultaneously quantify different dimensions of wildfire impacts is the *Wildfires*

Impacts Report edited by the Global Institute of Sustainable Forestry & Environmental Studies - Yale University (Morton et al., 2003). This report summarizes the current status of wildfire impacts in the United States by taking into consideration information from federal, state and local sources. Based on a survey of 10 large wildfires that occurred during the previous decade, the report provides examples on the magnitude of a variety of wildfire impacts by providing an opportunity to assess the utility for policy formulation and wildfire management. Economic impacts were, generally, obtained by considering fire suppression costs, damages to infrastructures and private properties; environmental impacts by taking into account the increased levels of carbon dioxide in the atmosphere, and damages caused to vegetation (burnt area), peat and soils, which results in loss of valuable habitat for autochthonous wildlife; while social impacts, including loss of income for the tourism industry, emotional stress associated with the fire events and damages to cultural or historic sites, are sometimes difficult to assess in detail. In the report, total environmental, social, and economic impacts are a function of multiple factors, including fire size, location, and burn intensity.

Using long term (2002–2007) satellite-based fires and aerosol data, coupled with atmospheric trajectory in Europe, Barnaba et al. (2011) estimates the wildfires environmental/health impacts by looking at the contribution of the European aerosol optical thickness (AOT)[2]. Essentially, the authors found that fire related aerosols play a major role in shaping the AOT yearly cycle at the continental scale. In general, the regions most impacted by wildfire emissions and/or transports are the Eastern and Central European regions as well as Scandinavia. Conversely, a minor impact is found in Western Europe and in the Western Mediterranean regions. Moreover, they estimate that during springtime, 15% of the European fine fraction of AOT is attributable to wildfires. The estimated impact maximizes in April (20–35%) in Eastern and Central Europe as well as in Scandinavia and in the Central Mediterranean. An important contribution of wildfires is also found in summer over most of the continent, particularly in August over Eastern Europe (28%) and the Mediterranean regions, from Turkey (34%) to the western Mediterranean (25%). The results suggest that this fire related, continent-wide haze plays a non-negligible role on the European radiation budget and, possibly, on European air quality, thus, representing a clear target for policy intervention.

Generally, economic costs range from direct costs associated with fire fighting to loss of income from the land following wildfire incidents and damage to property. Restoring damaged habitats is also becoming an important component of post-wildfire recovery in sensitive

[2] Degree to which aerosols prevent the transmission of light by absorption or scattering of light.

environments, which is typically a very costly and time-consuming process. In this respect, monetization of specific impacts may be difficult to achieve, since not all impacts may be amenable to monetary analysis. The components of the damage resulting from wildfires have been subject to extensive analysis by various authors (Valese et al., 2011; Ciancio et al., 2007), both for the monitoring and reconstitution aspects and for environmental and social impacts. In this regard, Valese et al. (2011) measured the economic damage arising from forest fires in north-eastern Italy (Veneto region) looking at 10 selected large fires that occurred in 2006, 2007 and 2008.

Based on the information provided by the Italian Forestry Corp (Corpo Forestale dello Stato), for every single fire event it has been estimated that, in Veneto, on a yearly average, the economic cost of wildfire is about €50 million (equivalent to every Veneto citizen paying about €10 per capita). The applied methodology takes into account several indicators including: (i) costs related to the regular staff (manna employee of the Italian Forest Corp has a gross average salary of €1,700 per month) and extraordinary staff (volunteers are not paid, but the equipment they use has an average price of approximately €1,500), (ii) the cost of maintenance of helicopter and extinguishing means, (iii) costs incurred for the restoration of forests (€1,500-2000 per hectare), and (iv) damage caused by the decreased production of woodland products (€3,500 per hectare). Moreover, the Italian Academy of Forest Fire and the Italian Forest Corp (Ciancio et al., 2007) provided a methodological framework for the economic evaluation of specific impacts arising from wildfire. In particular, according to this study, the total impact is divided into three components: (i) *fire extinguishing costs* can be estimated by knowing the average personnel cost per hour, the number of people who were employed, the duration of the extinction operations and the cost of the equipment used. Usually, all of this information is collected for each fire event in Italy, (ii) *environmental damage*; a conventional approach for an analytical estimation of the environmental damage can refer to the cost of reconstruction or restoration, and (iii) *external damage* to physical assets (e.g. infrastructures, building, etc.) in terms of reconstruction costs; damage to human capital, or disability (temporary or permanent) and, in extreme cases, loss of human lives.

From a social point of view, according to Ortuno-Perez et al. (2004), the disappearance of forests as a source of wealth, due to fire events, involve the loss of many traditional jobs, such as conveyors, carpenters, tappers, loggers and charcoal burners. Particularly, the aim of their work was to determine the socio-economic deterioration that large forest fires have produced in the municipalities of the Tietar Valley (Spain) where large wildfires occurred in 2003. The study found a close relationship between natural resources and economic development, especially in Pedro Bernardo and Guisando. This was shown by the fact that the number of holiday homes (the most important indicator for determining tourist activity) was lower in these towns than in other municipalities. They found that the most affected socio-economic indicator in the case of the Tietar Valley was the number of holiday homes, which led authors to conclude that tourism is the industry most affected by forest fires.

In conclusion, the traditional perspective in assessing wildfire impacts relies mostly on a case-study methodology. This approach seems to be the most effective strategy to obtain detailed information regarding different dimensions of such impacts in order to acquire the necessary knowledge to prevent and reduce negative economic, social and environmental impacts of wildfires. However, one point needs to be stressed: forest fires are often not the result of a natural disaster or causes of fatality, but rather an anthropogenic phenomenon, with an exclusive, direct dependence on social behavior, both voluntary or involuntary (Leone and Lovreglio, 2003). In this context, to our knowledge, no studies have been carried out so far in order to investigate the impacts of wildfires caused intentionally or not by humans (fire crimes).

1.3 Objective of the report

The main objective of this report is to analyze the extent of fire crimes and their impacts in both quantitative and monetary terms. To some extent, we also try to estimate whether the geographical impact of fire crimes is correlated to the presence of organized crime across Italian regions. Due to data limitations, our analysis will be conducted at two different geographical levels and take into consideration different time spans. Starting with a general overview of the forest fires issue at the European level in the last 20 years, the quantitative impact of fire crimes will be estimated both at the country-level for a selected number of European countries and at the regional-level for Italy. In particular, the European level analysis will be focused only on the environmental impact of fire crime while the regional level analysis on Italy will also take into account the assessment of other impacts arising from fire crime (health and material).

The monetary impact of fire crimes will be measured only at the Italian level. Such a methodological choice was due not only to the lack of suitable information at a wider geographical level, but also because, given the event-specific nature and the broad heterogeneity of this type of crime (i.e., protected areas, national parks, wooded/non-wooded area, etc.), the degree of accuracy in the estimate of the monetary damage is lower the larger the geographical area considered. The analysis was conducted using two different sources of data: i) the EFFIS database and (ii) the NIAB (Nucleo Investigativo Antincendi Boschivi) database. The European

Forest Fire Information System (EFFIS) database provides detailed yearly information on the number of forest fires, the size of the burnt area, the average fire size, and the different types of illegal activity (i.e. arsons and negligent fires). Data were collected for Mediterranean countries since 1985 and for the entire EFFIS network since 2005. The EFFIS network includes 24 EU Member States, 10 European non-EU countries and 4 MENA (Middle East and North Africa countries. The EFFIS data are also available at country, region and province level (NUTS1, NUTS2, NUTS3). Information on the causes of wildfires is crucially important to understand whether a fire is or is not a crime, to support the environmental and civil protection policies and to design appropriate prevention measures.

The Nucleo Investigativo Antincendi Boschivi (NIAB) is the investigative body of the Italian Forest Corp responsible for the national coordination and investigation activities related to forest fires. The NIAB database provides detailed information on each forest fire event that occurred over the period 2000-2012. In particular, the most useful information for our analysis are: the size of the burnt area, the environmental characteristics of the burnt area (i.e., national parks, protected areas, etc.) the number of people who died or were injured from fire causes, the type of material damage (i.e., material assets, personal properties, etc.) the number of offenders detected, the number of offenders arrested, the duration and the number of people who were employed in the extinction operations and the cost of the equipment used.

2 Description of methodology

2.1 Methodology of the quantitative impacts of fire crimes

Forest fires can be defined as uncontrolled fire caused naturally or by humans with susceptibility to expand into forested areas, arboreal or bushy, including any structures and infrastructures placed within such areas, or on cultivated or uncultivated land and pastures adjacent to these areas (National Wildfire Coordinating Group, 2012). However, naturally occurring forest fires are far less frequent than man-made ones. The most frequent causes of natural wildfires are volcanoes, lightning, spontaneous combustion, and sparks caused by rockslides. Usually, forest fires "*are started by human negligence. Intentional arson, equipment sparks, discarded cigarettes, unattended campfires, controlled agricultural burns and power lines have all caused a number of wildfires in the past*" (Green Conduct Blog)[3].

[3] http://greenconduct.com/blog/2013/07/25/the-environmental-consequences-of-forest-fires/ (accessed on 27th February, 2015).

With an increase in deforestation across the globe, highly flammable grasslands have taken their place. Statistically speaking, six times as many wildfires are started by humans than any natural causes (JRC, 2009). For this reason, knowing the causes of fires is crucial to a) understanding the extent of the fire crime, and b) supporting the environmental and civil protection policies by designing appropriate prevention measures.

At the European level a joint scheme with four causes (deliberate, accident/negligence, natural and unknown) of fires has been implemented to collect detailed information since 1992. European countries use national schemes that are often much more comprehensive than the simple four common causes, but they are not coherent with other country schemes and comparisons may be arduous (San Miguel and Camia, 2010). The need for a new EU scheme, more accurate than the four simple categories and harmonized across European countries, has been developed in order to improve the level of information and knowledge on the origins of forest fires in Europe. This new scheme has been designed to be pertinent with few changes to the previous country settings, preserving as much as possible the historical data series of each country while also utilizing as much as possible the level of detail of the information available. A key feature of the new scheme is that common fire causes categories have to be widely agreed/recognized; in addition, clear and unambiguous definitions are an integral part of the new classification scheme. The new classification of fires, similar to the former "Common Core" EU scheme, considers the following categories:

–*Unknown*: wildfire with no cause found;

–*Natural*: any wildfire caused by natural origin, with no human involvement in any way;

–*Accident*: wildfire unintentionally and indirectly caused by human without use of fire, connected neither to will nor to negligence, rather to fatality;

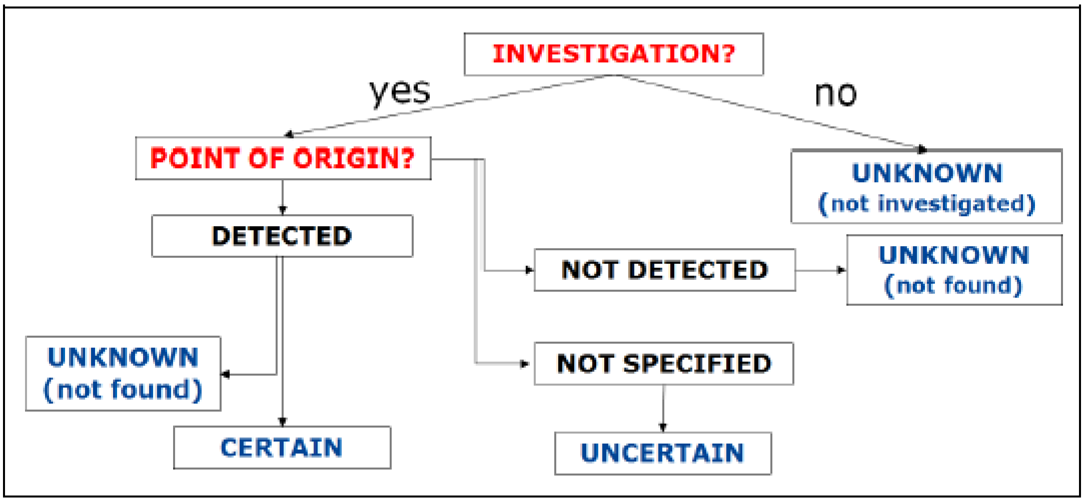
–*Negligence*: Wildfire unintentionally caused by human using fire or glowing object, not connected to fatality;

–*Deliberate*: Wildfire intentionally caused by human with the use of fire;

–*Rekindle*: Wildfire caused by re-ignition of a previous fire, due to latent heat or embers.

As emphasized by San Miguel and Camia (2010) in their executive report on European Fire Causes Classifications (EFCC), the new scheme is based on the objective evaluation of the fire cause through accurate investigations that aim to identify the point of origin of the fire (ignition point). If the point of ignition is known (and possibly also the ignition device), the fire cause might be recognized and reported in the database as certain (however, in some cases, even with known point of ignition, the cause cannot be identified and remains unknown). If not, the fire cause is to be reported as either uncertain or unknown. If the point of ignition cannot be precisely specified, remaining vaguely identified while different elements from the investigation support specific assumptions about the fire cause, then the fire cause can be reported as uncertain. If the point of ignition is not detected, the fire cause is included in the database as unknown. Figure 1 below shows a binary key to assessing the level of certainty in the identification of the fire causes in order to be recorded in the fire database.

Figure 1. Determination of the level of certainty of fire causes identification



Source: San Miguel and Camia, 2010.

Box 1. Kernel density maps and the Delphi method

Kernel maps allow for an immediate cartographic visualization of both two- and three-dimensional images of the ignition points of the events in the geographic area taken into account; thanks to this, it is possible to plan prevention activities. Such activities are crafted on the areas where specific types of wildfire frequently occur and sometimes with impressive regularity: they are almost always unintentionally caused and the consequence of supposed agriculture-related or cleaning-up needs.

The Delphi method is a technique to obtain solutions to an issue. It relies on a panel of anonymous experts, who propose independent solutions, refining them in two or three rounds. At the end of each round, a facilitator drafts a summary of the different answers proposed by the experts and their supporting arguments and sends the summary back to all the experts, so that they can reconsider their own answers on the basis of their colleagues'. Finally, a sort of arithmetic mean of the answers is determined.

The person who coordinates this process is called a *facilitator:* as a matter of fact s/he arranges the answers of the group of experts. The experts are chosen on the basis of their expertise and are asked to contribute opinions and points of view.

The Delphi method is therefore a systematic and interactive review method based on independent reasoning of selected international experts; it values experts' opinions, experiences and intuitions and allows using these unique resources as valuable information when complete and exhaustive scientific knowledge is not available.

The extent of environmental, social, and economic impacts of fire crimes is therefore, a function of several factors such as the size, intensity, location and causes (deliberate and negligent) of the event. In order to investigate the wide range of impacts arising from fire crimes, based on the availability of data and consistently with the objectives of the present report, some environmental, material and health indicators are taken into account.

From an environmental point of view, the occurrence of a forest fire can change the physical and structural characteristics of the vegetated landscape, thus producing significant variations of its vegetation, soil and fauna. Ecosystems affected by fires lose, totally or partially, the plant cover for a period of time lasting up to several months or even years. During this time, the soil is exposed to erosion by wind and water that can contribute to the degradation of the nutrient cycling (carbon, nitrogen, phosphorus) essential for vegetation, and thus, for herbivorous animal species. The environmental impact of forest fires on ecosystems is a result of the combined effects of warming produced during the fire combustion (fire severity), and the loss of vegetation (fire size). Therefore, the most critical fires are those reaching very high temperatures at surface level giving rise to the loss of nitrogen and organic materials from the soil. In order to estimate the environmental impact of forest fire crime, at least three main fire components should be considered, namely fire cause, fire size, and fire severity (fires whose burnt area > 50 ha). While fire cause allows us to select only fire crimes, fire size helps to quantify the burnt area that could have been potentially damaged. Moreover, knowing fire severity is useful to assess the average damage level. Hence, we estimate the environmental impact using the following indicator:

$$Environmental\ Impact\ (ha) = Average\ Burnt\ Area\ per\ fire\ (ha) * \ Number\ of\ fire\ crimes \quad (1)$$

With regard to our regional level analysis within Italy, to give a more accurate estimate of the environmental impact, we combine the indicator (1) with the land use type (for example, forest vs. non-forest) and for forest characteristics (i.e. protected areas, national parks and so on). In particular, the NIAB database provided us detailed information on each forest fire event that occurred over the time period 2010-2012. Therefore, we are able to exactly quantify the magnitude of the environmental impact that occurred in protected areas, with regard to the ordinary statute regions served by the Italian Forest Corp, by summing the burnt area (in ha) due to forest fire crime that occurred at the provincial level.

Moreover, wildfire provides an ample field to put human life in danger. Research on the human health impacts of forest fires takes place in a variety of disciplines including forestry, epidemiology, anthropology and so on. Particularly, epidemiologic studies (see Barnaba et al. 2011) focus on health consequences of air pollution created by the burning of biomass, while there are only few example of research (Viegas et al., 2009) aimed at investigating accidents involving fire fighters and the population affected by forest fires. The loss of human lives is the worst outcome arising from forest fires and this has repeatedly occurred in the last years. However, other important issues concerning human health are related to injury and pathologies affecting fire fighters and people with respiratory problems. Due to data limitations, we estimate the health impact of fire crime using the following as an indicator of the number of dead and injured people as a result of fire crime in Italy during the decade (2003-2012).

$$Health\ Impact\ = Number\ of\ deaths\ and\ Number\ of\ injured \quad (2)$$

Finally, another important wildfire impact to focus on is the material impact. Namely, the loss of income due to the destruction of human physical assets that takes place in forest fires affected/surrounding areas. Occasionally, forest fires burn houses, factories, farms and infrastructures, in a zone commonly called the *wildland-urban interface*[4]. In recent years, as living standards have evolved, rural areas have been populated by secondary homes, which have enlarged the wildland urban interface. Therefore, human populations and related physical assets are exposed at a higher risk of forest fires than ever. As stated in a CRS Report (2012), it seems that one or more fires annually have burned down several to a few hundred homes and outbuildings (sheds, garages, etc.) in Europe, and these structures generally have ignited in one of two ways: through direct contact with fire or through radiation (heating from exposure to flames). Given data availability, in order to quantify the material impact of fire crimes in Italy over the decade 2003-2012 we look at four main categories of assets: (i) *vehicles* including wagons, bicycles, motor vehicles (motorcycles, cars, trucks, buses), railed vehicles (trains, trams); (ii) *material assets* (any item of movable personal property, such as furniture, domestic animals, etc.); (iii) *real estate* (i.e. immovable property, buildings or housing in general but also farmland); and (iv) *farm animals* (e.g. cows, horses, pigs, chickens, ducks, rabbits, goats, sheep).

$$Material\ Impact\ = Vehicles, Material\ assets, Real\ estate\ and\ Farm\ animal \quad (3)$$

2.2 Methodology of the monetary impacts of fire crimes

Generally, economic costs range from direct costs associated with fire fighting to loss of income from the land following wildfire incidents and damage to property. Restoring damaged habitats is also becoming an important component of post-wildfire recovery in sensitive environments, which is typically a very costly and time-consuming process. In this respect, monetization of specific impacts may be difficult to achieve since not all impacts may be subjected to monetary analysis.

The components of the damage resulting from wildfires have been the subject of extensive analysis by various authors (see, for example, Barbosa et al., 2004 and Bovio et al., 2005), both for the monitoring and reconstitution aspects and for environmental and social impacts. From this perspective, the Italian Academy of Forest Fire and the Italian Forest Corp (2007) provided

[4] The wildland-urban interface is an area where structures (homes, factories, farms) are in or near wildlands (forests or rangelands). For more information, see CRS Report RS21880, Wildfire Protection in the Wildland-Urban Interface, by Ross W. Gorte and Kelsi Bracmort (2009).

a methodological framework for the economic monetization of specific impacts arising from wildfire. In particular, according to this study, the total impact is divided into three components:

- *Fire extinguishing costs*: these costs can be estimated by knowing the average personnel cost per hour, the number of people employed, the duration of the extinction operations and the cost of the equipment used. Usually, all this information is collected for each fire event (e.g. Italy);

- *Environmental damage*: a conventional approach for an analytical estimation of the environmental damage can refer to the cost of reconstruction or restoration. The estimation criterion is based on the assumption that an asset is worth (at least) what it costs;

- *External damage* to physical assets (e.g. infrastructures, building, etc.) in terms of reconstruction costs: damage to human capital, or disability (temporary or permanent) and, in extreme cases, loss of human lives.

$$\textit{Total Damage} = \textit{extinguishing costs} + \textit{environmental damages} + \textit{external damages} \qquad (4)$$

It is worth noting that, considering the existing methodology applied in literature (Ciancio et al., 2007) and the good availability of data, we can proceed in providing new insights for the assessment of different impacts arising from fire crimes and for the economic monetization of some of them. To this end, we rely on some in depth case studies (Maracallo and Favale di Monfasso-Italy) that could offer a springboard to advance theoretical arguments with much broader policy implications.

3 The quantitative impact of forest crime

Whether caused by human negligence or naturally occurring, wildfires can generate significant impacts on the economy, on the environment, on the affected communities and on human health. An appropriate analysis on different impacts of wildfires could be crucial in terms of an effective, appropriate prevention to mitigate the loss of biodiversity and an efficient protection action can slow the ecological degradation and contribute to the development of an effective environmental policy at the European Level.

3.1 General overview of the phenomenon at the European level

Forest fires represent a persisting problem for most Mediterranean countries showing a rather upward trend during the last twenty years, even though these countries have been investing more funds in methods to prevent and mainly suppress them (Tampakis at al., 2005). Therefore, fire trends in Europe show a high concentration of fire events and, more importantly, fire impacts in the Mediterranean regions. Most of the total burnt area in Europe concentrates in this geographical area. The average area affected by fires annually across Europe reaches 500,000 ha, and 95% occur in the Mediterranean countries, with approximately 35,000 events per year. Assuming that such a phenomenon is regularly distributed over time, then this averages about 100 fires a day throughout the year (European Commission, JRC technical Report, 2012).

The long time series of wildfire events available from the EFFIS database for these five southern countries (Portugal, Spain, France, Italy, and Greece) validates a separate analysis, as mentioned earlier. The latest data reported by the EFFIS system on the EU are those for the year 2012, which have been elaborated and published in 2013. Table 2 below summarizes the extent of wildfires that occurred in Mediterranean countries during the last two decades, taking into account the number of fires, the burnt area and the fire severity. At this stage of the analysis, we disregard fire causes, which will be the object of a more detailed analysis in the following sections.

Table 2. Number of fires, burnt area and fire severity in the five southern Member States (1992-2012)

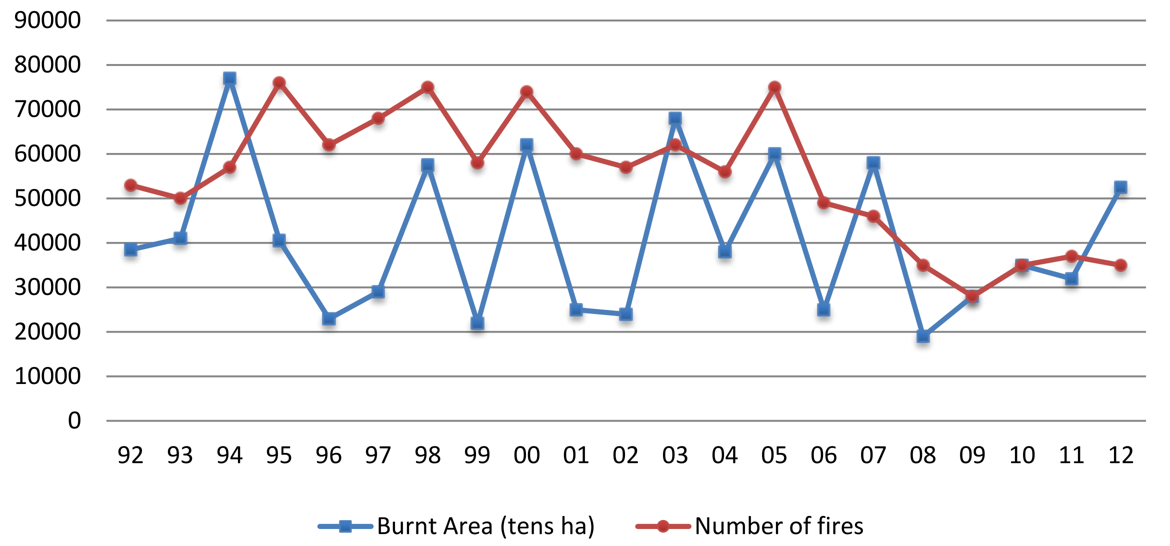
	ITALY	SPAIN	PORTUGAL	FRANCE	GREECE	TOTAL
Number of Fires						
2012	8,252	15,902	21,176	4,105	1,559	38,994
% Total in 2012	16%	31%	42%	8%	3%	100%
Average 1992-2002	11,078	18,152	22,250	5,538	1,748	58,851
Average 2002-2012	9,736	18,337	24,949	4,406	1,695	50,645
Total 1992-2012	203,294	335,793	409,485	103,357	34,188	1,086,117
Burnt areas (ha)						
2012	130,814	209,855	110,231	8,600	59,924	519,424
% total in 2012	25%	40%	21%	2%	12%	100%
Average 1992-2002	147,150	161,319	102,203	22,342	44,108	448,938
Average 2002-2012	83,878	125,239	150,101	9,433	49,238	430,798

Total 1992-2012	2,442,234	3,775,048	2,383,346	580,421	1037,106	1,0132,344
Fire Severity						
Fires 2012	8,252	15,902	21,176	4,105	1,559	38,994
Burnt area > 50 ha	68%	69%	62%	58%	60%	64%
Average 1992-2002	11,078	18,152	22,250	5,538	1,748	58,851
Burnt area > 50 ha	52%	51%	47%	34%	38%	48%
Average 2002-2012	9,736	18,337	24,949	4,406	1,695	56,645
Burnt area > 50 ha	55%	52%	51%	48%	46%	52%

Source: authors' elaborations on EFFIS Database

In 2012, in these five most affected countries, forest fires burned a total area of 519,424 ha: such an impact has been well above the average value of the last 20 years (about 400,000 ha) and among the highest since 2000. On the other hand, the number of fire events has been below the average and among the lowest over the last two decades (see Table 2). Figure 2 clearly shows this unusual trend by combining the number of fires per year with the total burnt area in the five southern Member States from 1992 to 2012. In particular, 2012 displays a local maximum for burned area despite the decrease in the number of fires that occurred, which is well below the average for the period. This evidence shows that wildfires during this period were clearly much more destructive than in past years; notably, in 2012, 64% of the total area affected by fires was burned by high-intensity wildfires making vegetation communities highly vulnerable. Therefore, as already emphasized by Montealegre et al. (2014), fire severity plays a key role in determining the magnitude of wildfire impacts on the environment in terms of total tree mortality, loss of biodiversity (i.e., in the form of animal species, vegetation) giving rise to a deep ecological landscape change.

Figure 2. Burnt area and number of fires in the southern EU Member States (1992-2012)



Source: authors' elaborations on EFFIS Database

3.1.1 Some estimates on fire crimes in Europe

As we already emphasized previously, forest fires are often not a natural disaster or fatal to human life, but rather an anthropogenic phenomenon. Hence, knowing fire causes is crucial to understanding the extent of such an environmental crime and to support the environmental and civil protection authorities in designing appropriate prevention policies.

By definition, an environmental impact is a change that directly affects an ecosystem through damages caused to the vegetation, peat and soils, which result in loss of valuable habitat for autochthonous wildlife. Therefore, a possible way to quantify the environmental impact is to focus on the countries' areas affected by forest fires and assess how much of this area has been burnt because of fire crimes. To this end, we need to determine *a priori*, for each Member State, how much area has been burnt on average, by a single forest fire during the last decade. Then, by combining such information with the forest fire causes, the environmental impact at European level for the decade 2003-2012 is estimated by the following indicator:

*Environmental Impact = Average Burnt Area per fire (ha) * Number of fire crimes*

Using the EFFIS database, which provides information on fire causes (natural, accidental, responsible and deliberate), table 3 shows the number of fire crimes, the related burnt area (environmental impact) and the percentage rate of fire crime for the 21 Member States over the decade 2003-2012.

Table 3: Assessing the environmental impact in the 21 Member States from 2003 to 2012.

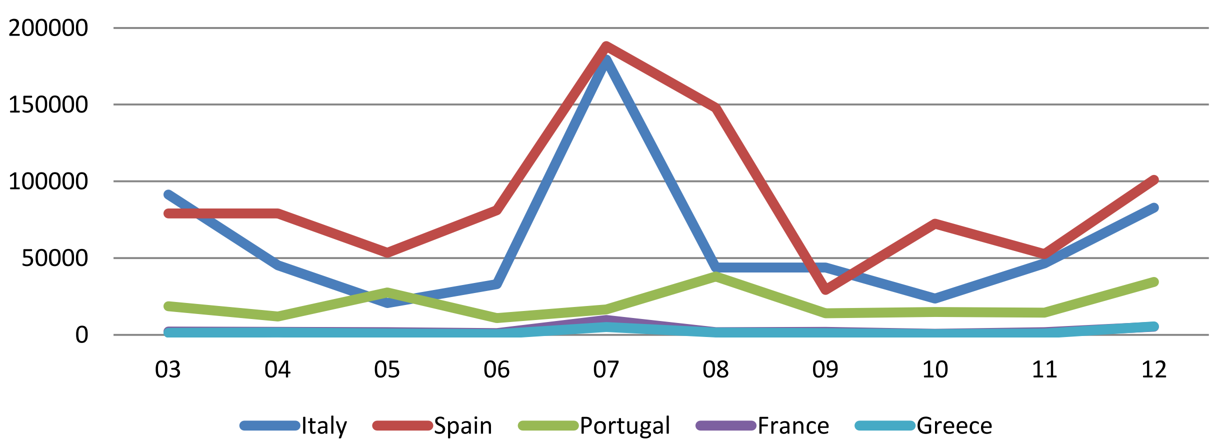
Country	Number of fire crimes	Average burnt Area (ha)	Environmental Impact (ha)	Crime %
Belgium	270	0.19	52.7	8.3
Switzerland	154	1.28	197.3	47.3
Czech Republic	559	0.29	163.7	21.3
Cyprus	280	5.02	1,198	19.8
Germany	450	0.54	244	15.9
Spain	89,282	9.5	848,241.4	63.7
Finland	229	0.06	14	2.0
France	6,033	4.06	24,494.4	28.5
Greece	3,819	4.09	15,619.71	45.6
Croatia	3,160	0.54	1,724.4	20.2
Hungary	206	1.98	408.2	11.5
Italy	48,105	11.613	558,643.4	65.6
Lithuania	397	0.27	108.7	12.4
Latvia	784	0.22	170.5	15.9
Poland	1,288	4.5	5,9035	57.2
Portugal	17,870	3.8	72,838.3	31.6
Romania	478	1.69	807.9	30.4
Serbia	822	2.11	1,734.4	33.1
Slovenia	135	1.61	218.4	22.7
Slovakia	294	0.33	98.5	12.5
Turkey	1,413	1.90	2,691	20.4
TOTAL			**1,535,572.41**	

Source: authors' elaborations on EFFIS database

During the last decade, in the 21 Member States, forest fires due to human causes burned a total area of 1,535,572.41 ha. Environmental impact varies notably from one country to another according to the meteorological conditions related to the geographical location of the countries. The extent of the environmental impact due to fire crime is particularly significant for the southern Member States. Looking at table 3, we see that Spain was the most affected country in terms of burnt area due to fire crimes, with 40% of the whole burnt area in Europe (848,241.4 ha), followed by Italy (558,643.4 ha) and Portugal (72,838.3 ha). Conversely, less affected countries were Finland (14 ha), Belgium (52.7 ha) and Slovakia (98.5 ha). Another important aspect to consider is the extent of fire crime. In this framework, taking into account only known forest fire causes, we are able to assess the magnitude of such crime by considering the percentage ratio between fire crime causes (deliberate and negligent) and the total number of wildfires. Forest fire events due to deliberate actions and negligent behaviors account for 66% of the total number of forest fires in Italy, followed by Spain (64%) and Poland (57%) (see Annex 1 for more detailed information). Despite a quite low environmental impact of fire crime in Poland (5,903.5 ha), the fire crime rate is fairly significant. This is mainly due to the cultivation techniques involving the burning of stubble of the large lands (50% of the country land is cultivated). Therefore, among forest fire causes in Poland, the consequences of fire transfer from the meadows and plant residuals subjected to burning has been growing in recent years, this is also due to the insufficient national legislation for tackling fire crimes (Ubysz, 2003).

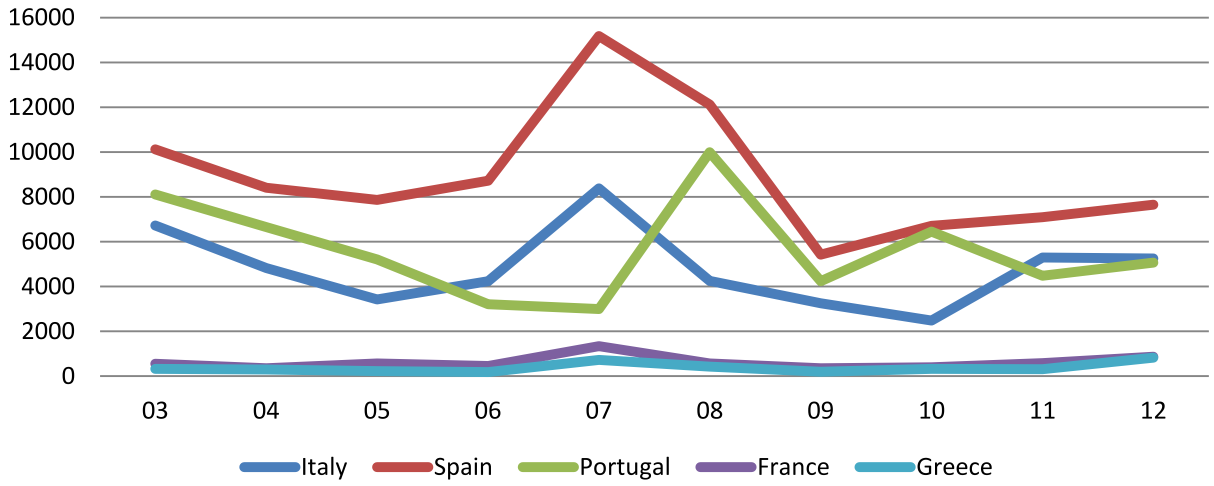
As already emphasized in the previous section, the long time series of fire crime available from the EFFIS database for some southern countries (Portugal, Spain, France, Italy, and Greece) should deserve an *ad hoc* extensive analysis to investigate the trend over time of the environmental impact caused by fire crime. Figures 3 and 4 show the evolution of the number of fire crimes and related total burnt area per year in the five southern Member States over the last decade (2003-2012). The statistics vary considerably from one year to the next but they mostly follow the same trend for the different countries, which clearly denote that the total burnt area could depend mostly on seasonal meteorological conditions. For instance, during the "dramatic" summer of 2007, Europe experienced extremely hot and dry weather conditions that led to a number of extraordinarily large forest fires over the Iberian Peninsula and the Mediterranean coast. Overall, taking into account the five countries all together, we can see that 2012 exhibits a local maximum for burned area; the 2006, 2007 and 2008 fire seasons were the worst years followed by three consecutive relatively "positive" years.

Figure 3. Trend of the environmental impact (ha) in the five most affected European countries (2003-2012)



Source: authors' elaborations on EFFIS database

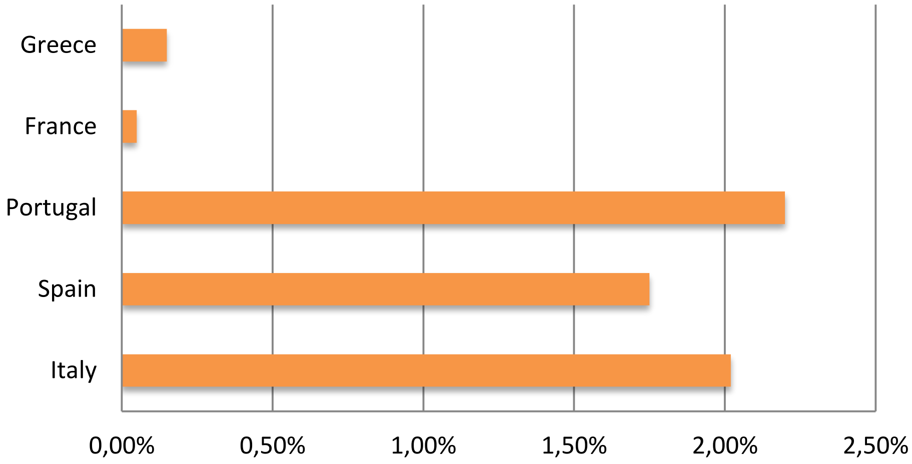
Figure 4. Number of fire crimes in the five most affected European countries (2003-2012)



Source: authors' elaboration from European Fire Database

Since the area of each country is different, and the area at risk within each country is also different, the comparisons of the magnitude of the environmental impact among countries cannot be absolute. Therefore, it could be convenient to consider the burnt area for each country and, thus, the environmental impact with regard to the individual country's surface area. Figure 5 shows how much of each country's total surface has been burnt due to fire crime during the last decade. Overall, Portugal has been the most affected country considering the ratio (2.02%) between burnt area (201,210.9 ha) and total surface area (9,209,000 ha), followed by Italy (2.02%), Spain (1.75%) and, to a lesser extent, by Greece (0.15%) and France (0.05%).

Figure 5. The evironmental impact by country total surface



	Italy	Spain	Portugal	France	Greece
Burnt Area/Total Surface	2,02%	1,75%	2,20%	0,05%	0,15%

Source: authors' elaboration from European Fire Database

3.2 Forest fire crime in Italy

3.2.1 Normative framework

In Italy, the legislation regarding activities to contrast forest fires was derived from the following constitutional principles:

–Italian Constitution - Article 9. "The Republic […] safeguards the landscape and the historical and artistic heritage of the Nation";

–Italian Constitution - Article 32. "The Republic safeguards health as a fundamental right of the individual and as a collective interest […]";

–Law 121/1981, Article 16, paragraph 3 assigns to the State executive bodies the capacity of public protection for the safeguard of public safety and the maintenance of public order in those areas where forest fires occur;

–Law 225/1992 and subsequent amendments and additions (Law 353/2000), establishes the National Service of Civil Protection to which functions of civil protection: prediction, prevention and proactive fight (monitoring; surveillance; sighting, alarm, and fire extinguishing through vehicles both by land and air; management, coordination and suppression of forest fires) are assigned;

–VI Title, Penal Code - crimes against public safety and special law - Law 353/2000, on public security to safeguard public safety and activities to contrast forest fire crimes.

The legislation regarding activities to contrast forest fire crimes has been enriched over time with the following norms:

- Decree August 4th, 2000 "Amendments to the Penal Code" converted into Law 275/2000;
- Establishment of the Investigative Unit Against Forest Fires (Nucleo Investigativo Antincendi Boschivi, N.I.A.B.) within the former General Directorate of Forest, Mountain, and Water Resources, on August 10th, 2000 now reorganized and strengthened by Decree of the Head of the Forest Body in February 2013;
- Law 353/2000 "Framework law concerning forest fires"; Law 353/2000 contains very important elements to address the issue, including:
- The legal definition of "forest fire"
- The assignment of crucial tasks to the regions for fire prevention and fighting.
- The provision of a census of all the burned areas;
- The introduction, in Title VI of the Penal Code, of a specific crime for forest fire (Article

423-bis, Penal Code);

- The temporary prohibition of new constructions on the areas touched by the fire;

Under Article 2, Law 353/2000 "The term *forest fire* indicates a fire susceptible to expand on wooded, bushy, or arboreal areas, including any anthropic facilities and infrastructure located within the aforementioned areas, or on cultivated, uncultivated, and grazing lands neighboring such areas."

For the purposes of the application of the penal norm, it is fundamental to properly define the concept of "forest" as the object to safeguard.

At the national level, Article 2 Legislative Decree no. 227/2001, amended by Article 26 Law 35/2012, contains useful references for the legal definition of *forest*:

• It specifies that the terms *wood* and *forest* are equivalent, that is, in legal terms they have exactly the same meaning;

• It does not provide an unambiguous definition of *forest,* which could be applied as it is throughout the national territory; in fact, Regions are entrusted with the exact identification of the concept;

• It equates to forest under any circumstance and therefore throughout the country: a) the lands encumbered by the obligation of reforestation for the purposes of defense of the territory and protection of the landscape and of the environment in general; b) forest areas which, for various reasons, are temporarily lacking trees and shrubs; c) the glades and all other limited areas that interrupt the continuity of the forest;

• It introduces the so-called "state" definition of forest, which is valid until the enactment of regional laws and, unless otherwise already defined by the Regions themselves, the territories referred to in paragraph 6 are considered to be a "forest";

Constitutionally, Article 117, as amended by the Constitutional Law 3/2001 is important, as it assigns;

• Exclusive competence to the State in the field of environmental safeguard policies (paragraph 2); and

• Concurrent competence to State and Regions regarding the promotion of environmental and cultural assets (paragraph 3):

Consequently, the definition of *forest,* which is functional to the safeguard of the landscape, the environment, and the ecosystems, is in the sole responsibility of the state that manages it according to the mentioned national legislation. In certain cases, it may be the Regions' duty to establish a different concept of *forest* for the territories under their jurisdiction and only for different purposes connected to regional powers, such as the development of agriculture and forests, the management of wood production, the reforestation activities, and the management of the chestnut trees. Regional powers are therefore limited to productive and managerial aspects of the forest. Consequently, in the case of criminal proceedings that have the forest as their object, the forest must be identified as the common good. In particular, when curbing forest offenses or crimes committed within the forests, law enforcement agents and the judiciary authority have to identify the *forest* as the common good to be protected according to the unvarying criteria established at the national level, which are precisely those indicated in Article 2, paragraph 6. As a matter of fact, the common good that is protected by the criminal law, can only be the same throughout the national territory and the application of the criminal law cannot allow differences in the ways the common good is regarded across regions. Therefore:

–Only a few regions have legislated in compliance with the provisions of Article 2, paragraph 2 of the 2001 Legislative Decree n. 227.

–Regional laws establish a concept of *forest* that goes essentially back to the State's definition, with the introduction of some specific features that must adapt the definition of *forest* to the forest areas under their jurisdiction and in accordance with regional policies for the management of such areas.

–In defining the *forest,* some Regions report typological and cultivation-related dimensions, which in some cases coincide with the State's definition, but in other cases are even more restrictive. Moreover, in some Regions the size parameters of the forest are defined by regulation, not by law. In other Regions, instead, the definition of forest is contained in legislative measures that regulate different matters.

Finally, several regions, in addition to defining the concept of *forest*, have also established what is not considered a forest. In principle, in such regions, the following are not considered to be forests: city parks, gardens and picnic areas, cultivations of Christmas trees of less than 30 years of average age, the rows of plants and the avenues of trees, the botanical gardens, the orchards, the woody crops when on lands that are not objected to hydro-geological constraints, the abandoned lands, the truffle-cultivated areas of artificial origin, the meadows, and the wooded pastures.

3.2.2 Characteristics of forest fire crime

The forest fire crime under the Article 423-bis of the penal code, is an offense of presumed danger, which means that the threat to the common good is sufficient reason for an allegation: the existence of an actual offence is not necessary. The danger is already in the action, in the potentially dangerous behavior. The Article stipulates that:

• The regulation is intended to protect public safety and national forests and forestry;

• The forest fire crime occurs when there is a fire that is susceptible to expanding to wooded areas;

• The attempted forest fire (performing suitable and unequivocal actions that cause damage to the common good and/or are presumed dangerous - threatening to the forest, even when the action is not fulfilled or the event does not occur) occurs when the fire ignition caused by the criminal has not occurred yet.

The characterizing features of the new article of the Penal Code regarding forest fires can be summarized as follows:

1. A specific common good is subject to legal protection: the woods or forests;
2. The sentence is increased in case of deliberate cause:
3. The sentence is increased also for the typical cases of crime against the environment;
4. The rule contributes to defining the legal concept of forest fire.

3.2.3 Prohibitions and requirements on the matter of forest fires

The law prohibits:

• Reforestation activities and environmental engineering supported by public funds for 5 years after a fire;

• The construction of infrastructures and buildings aimed at civilian settlements or productive activities or pastoral and hunting activities in the 10 years following a fire;

• The zone change of the land in the 15 years following a fire. This restriction must be reported, under penalty of nullity, in any contract, stipulated within 15 years from the fire, of sale of land and properties that fall within those areas.

For the purposes of the effective application of such a system of constraints and requirements, the Framework Law requires that every year the municipalities survey the areas affected by fires through a dedicated land registry. This provision has been disregarded by the majority of the municipalities, which resulted in government measures in 2007, intended to promote compliance to the regulations by the defaulting administrations. In any case, this type of prohibition appears to have worked as an effective preventive measure on the actions of destruction of the forest through fire for the purpose of property speculation, since the criminal project is in any case going to be nullified on account of the constraints mentioned above.

In case of violation of the prohibitions and non-compliance with the requirements, in addition to criminal sanctions, fixed or proportional administrative sanctions will be applied as well.

3.2.4 Administrative sanctions

Article 10 of the Framework Law also prescribes the following administrative sanctions:

• The violation of the prohibition of grazing on stands of wooded areas touched by fire is punished with a administrative penalty, from 30 to 60 euros per head (of cattle);

• The administrative penalty to be applied in case of violation of the hunting ban oscillates between 206 and 413 euros.

• The violation of the prohibition of construction of buildings and/or facilities and infrastructures aimed at civilian settlements and/or production activities on stands affected by fire is punished by the penalty provided for in Article 20, paragraph 1, letter. C) Law 47/1985 (imprisonment up to two years and a monetary fine that varies from a minimum of 15,493 to a maximum of 51,645 euros);

• Finally, a civil fine ranging from 1,032 to 10,329 euros is applied to those who violate the prohibitions established in the annual plans issued by the regions. Such plans identify, in the areas and periods at risk of forest fire, all the prohibited actions that may, even potentially, determine the onset of a fire.

Such penalties are doubled in the case that the offender belongs to CNVVF - National Corp of Firemen, the CFS - Italian Forest Corp, the Armed Forces, other police forces of the State, the Regional Forestry Service, the Regional Civil Protection Service, a volunteer organization committed in the fight against forest fire.

3.2.5 Administrative penalties and responsibilities for environmental damage

The following provisions are also relevant:

• The **General Prescriptions and Forestry Police** (PMPF), contained in the Forestry Law – Royal Decree no. 3267/1923, which enacts the prohibitions and requirements, whose violation entails the application of administrative penalties;

• The **Consolidated Laws of Public Safety** (T.U.L.P.S.) - RD no. 773/1931, Chapter V "Provisions on the prevention of accidents and disasters" whose Article 59 defines the violations subjected to monetary administrative fine;

Art. 18 Law 349/1986 - Establishment of the Ministry of Environment and rules regarding environmental damage, on the right to compensation for environmental damage, the determination of which includes the costs incurred for the active fight and the estimated damages to the topsoil and the soil.

3.3 An evaluation of the impact of forest fire in Italy

Among the Mediterranean countries, Italy represents one of the countries most affected by wildfires. Fire events cover the entire Italian peninsula, however, the larger wildfire events normally occur in the south. In 2012, throughout the country there were 8,252 forest fires, which burnt 130,814 ha in total, of which 74,543 were wooded. Compared to the previous year 2011, the total number of forest fires increased by just 1%, while the total areas burnt by fire increased by over 80% (in particular the wooded areas, which increased by 94% compared with an increase in non-wooded areas of 67%). The complete archive of the data relating to forest fires collected by the Italian Forest Corp is available from 1970 to the present. Figures 6, 7 and 8 show in detail the historic trend of forest fires in Italy (1970-2012) in terms of number of forest fires, burnt area (wooded and non-wooded) and average fire size.

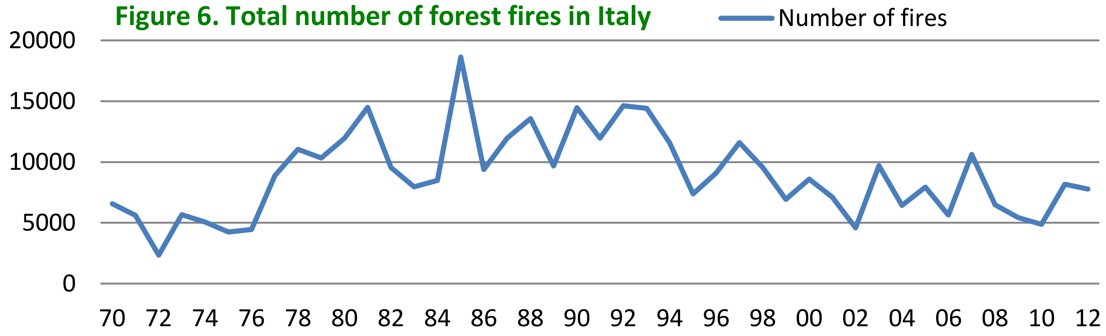


Figure 6. Total number of forest fires in Italy

Source: authors' elaboration from Italian Forest Corps data

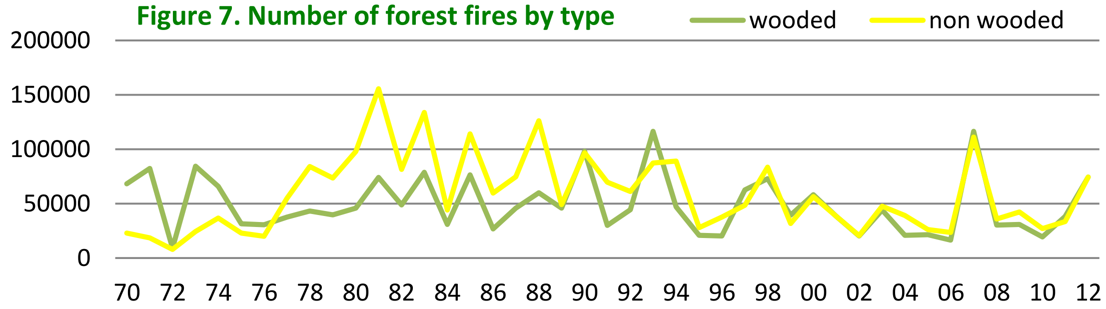
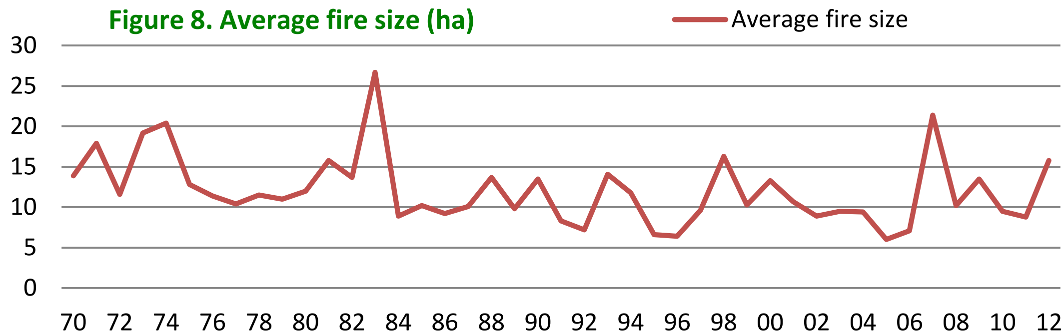


Figure 7. Number of forest fires by type

Source: authors' elaboration from Italian Forest Corps data



Source: authors' elaboration from Italian Forest Corps data

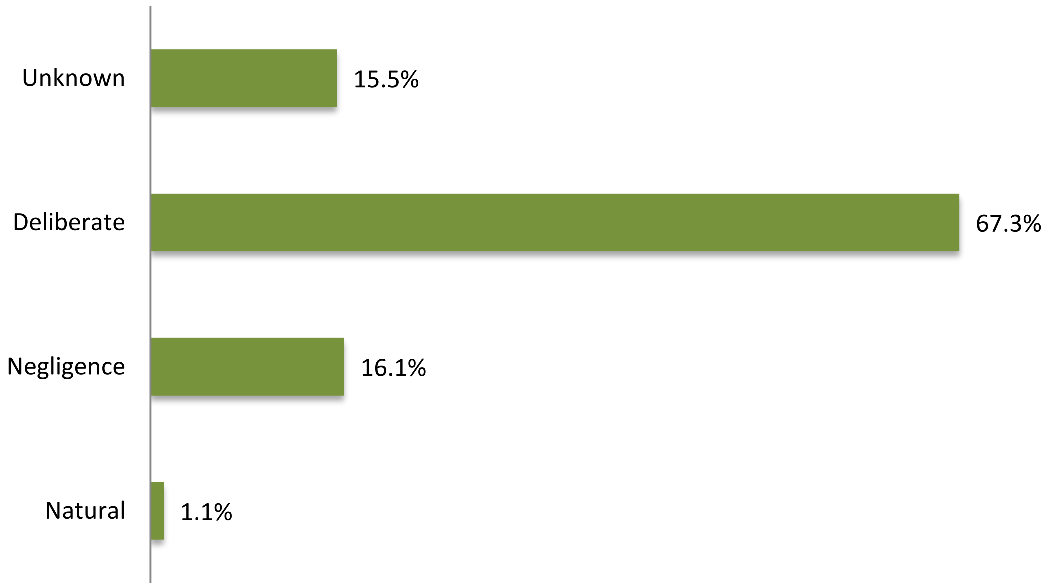
The number of wildfire events rose in the 1970s, subsequently stayed less than 10,000 per year until 1978, when there were more than 11,000 fire events, and persisted constantly high during the 1980s and 1990s. From 2000 to 2007, the average number of events decreased by one-third in comparison with the previous 20 years. The affected wooded area has been significant from the early '70s and has continued to be above 50,000 ha on average for over the last 30 years, decreasing to about 40,000 in the last 8 years. The non-wooded burnt areas were moderately low during the 1970s, with an average of 36,000 ha per year, reaching the highest value in the period 1980-89 with more than 93,000 ha per year and then dropping in the 1990s, with an average value of about 65,000 ha, further decreased to 45,000 in the last years. The average burnt area per fire has reduced gradually over the decades, from 13.5 to 12.7 ha during the 1970-1980s, to 10.6 in the 1990s, with a slight increase in the years 2000-2007 to 10.8 ha. The year 2012 shows a local maximum for burnt area (51,9424 ha), but at the same time the number of fires occurred decreased to well below the average recorded in the last two decades. As often emphasized by the Italian Forest Corp, the most critical seasons were recorded in 1985, for number of fires (18,664), in 2007 for forest area affected by fire (116,602 ha) and in 1981 for total area (229,850 ha).

3.3.1 Assessing forest fire crime

The Italian Forest Corp, in fighting forest fire crimes, has given impetus to both the centralized and decentralized organizations through the NIAB. This was established in 2000 by the General Inspectorate, which operates on the national territory, with the exception of the regions with special statute and the autonomous provinces (Sicilia, Sardegna, Valle d'Aosta, Trentino-Alto Adige, Friuli Venezia Giulia). The NIAB is responsible for coordination and direction of information investigations and analysis in relation to forest fires and provides operational, investigative and logistical support to the territorial offices of the Italian Forest Corp, also through the research of evidence collected at the scene of fires and the analysis of residues of explosives and triggers.

Knowing forest fire causes is crucial in understanding the extent of the crime. The cause analysis performed in this section is based on the causes attributed by the Italian Forestry Corp for statistical surveys of forest fires. This statistic, therefore, does not include details of the events, which occurred in the autonomous regions and provinces. With the exclusion of the most obvious cases, attributing causes of fire is a very complex process that is the outcome of a series of analyses carried out on the site of the event and subsequently, through investigative activities of the judicial police. In this context, figure 9 below shows forest fires and related causes that occurred in Italy in 2012. Particularly, forest fire events due to deliberate actions account for 67.3%, those due to negligent behaviors are about 16%, while naturally occurring forest fires are quite marginal (1%).

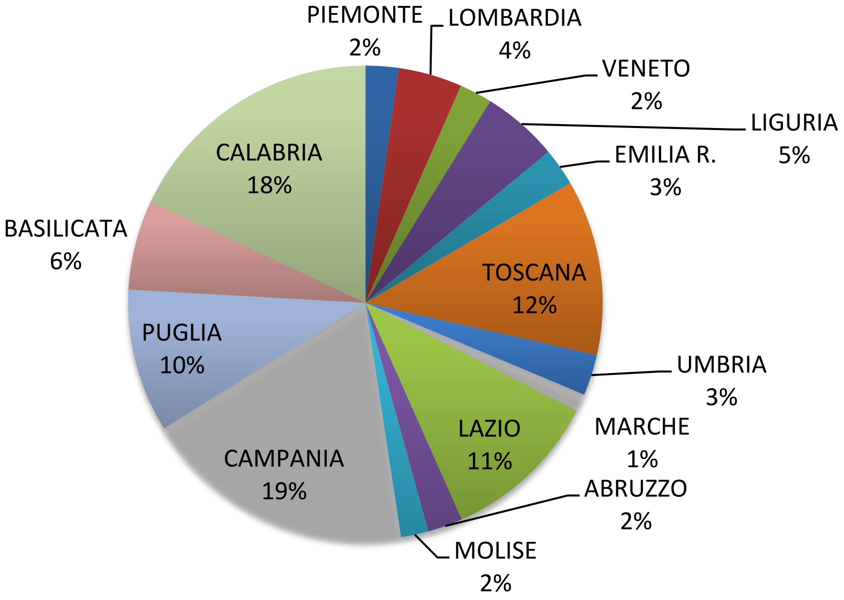
Figure 9. Forest fire by causes in Italy (2012).



Source: authors' elaboration from Italian Forest Corp data

The analysis of the number of forest fire crimes by region (Figure 10), in 2012, clearly shows that the most affected regions were Campania with 972 fire events (19% of the national amount), followed by Calabria (18%), Toscana (12%), Lazio (11%) and Puglia (10%): these regions concentrate almost 60% of the total forest fire crimes at national level, amounting to 3,622 events.

Figure 10. Forest fire crime in Italy (2012) by regions



Source: authors' elaboration from Italian Forest Corp data

3.4 Environmental impact

As already emphasized, a possible way to quantify the environmental impact of fire crime is to assess how much area has been burnt because of it. To this end, we are able to determine, on average per year, how much area has been burnt owing to forest fires during the last decade. By combining such information with forest fire causes (i.e. deliberate or negligent) we can try to quantify the environmental impact. Hence,

*Environmental Impact = Average Burnt Area per fire (ha) * Number of fire crimes*

Table 4 below displays in detail the number of fire crimes that occurred in Italy over the decade 2003-2012. Overall, the total environmental impact recorded in the last decade amounts to 610,485.52 ha due to 48,105 events of arson.

Table 4. Number of forest fire crimes, average burnt area and environmental impact in Italy over the last decade

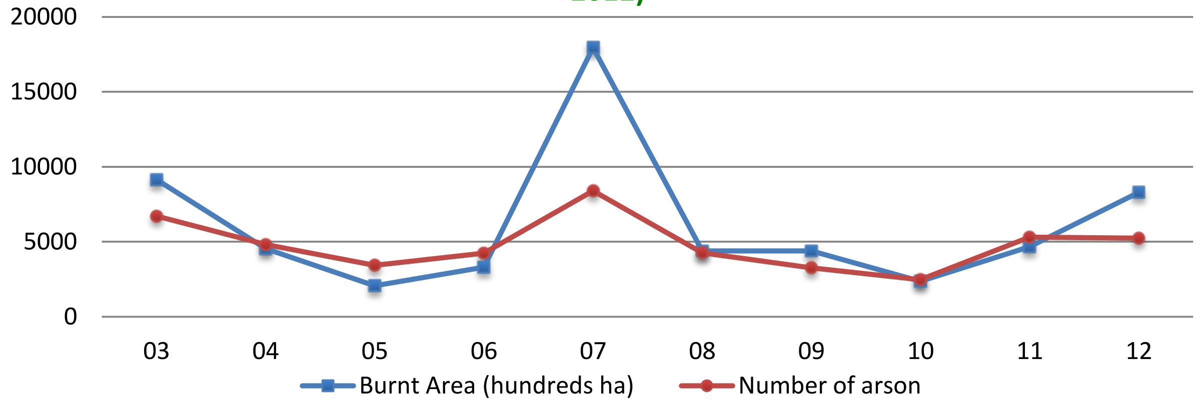
Years	Number of fire crimes	Average Burnt Area (ha)	Environmental Impact (ha)	Fire Severity
2003	6,720	13.6	91,392	46%
2004	4,823	9.4	45,336.2	52%
2005	3,422	6.01	20,566.22	53%
2006	4,238	7.8	33,056.4	54%

2007	8,384	21.4	1,794,17.6	61%
2008	4,250	10.3	43,775	54%
2009	3,251	13.5	43,888.5	52%
2010	2,475	9.52	23,562	51%
2011	5,296	8.8	46,604.8	61%
2012	5,246	15.8	82,886.8	68%
Average	4,810.5	11.6	61,058.55	55%
Total	48,105		610,485.52	

Source: authors' elaboration from Italian Forest Corp data

Figure 11 below depicts the progression of the number of fire crimes and related total burnt area per year in Italy over the last decade (2003-2012). In particular, 2012 exhibits a local maximum for the environmental impact, being overpassed in previous years only by the critical 2007 fire seasons. In line with other Mediterranean countries, in 2012 the number of fire crimes decreased but, at the same time, the overall burnt area increased. According to Montealegre (2014), in this framework, fire events resulted to be clearly much more destructive than in the past years since 68% of them represent large fire (>50 ha).

Figure 11. Environmental impact and number of fire crimes Italy (2003-2012)



Source: authors' elaboration from Italian Forest Corp data

In Italy, forest fire events are widespread throughout the country, however they assume different configurations moving from one region to another given the broad geographical heterogeneity. The latest data from NIAB reporting system are those for the year 2012, which were elaborated and published in 2013. Table 5 summarizes the extent of the forest fire crime that occurred in each Italian region in 2012 (excluding autonomous regions and provinces) taking into account the number of forest fire crimes, average burnt area and environmental impact. Moreover, in order to have a comparable indicator we link environmental impact to each region's total surface.

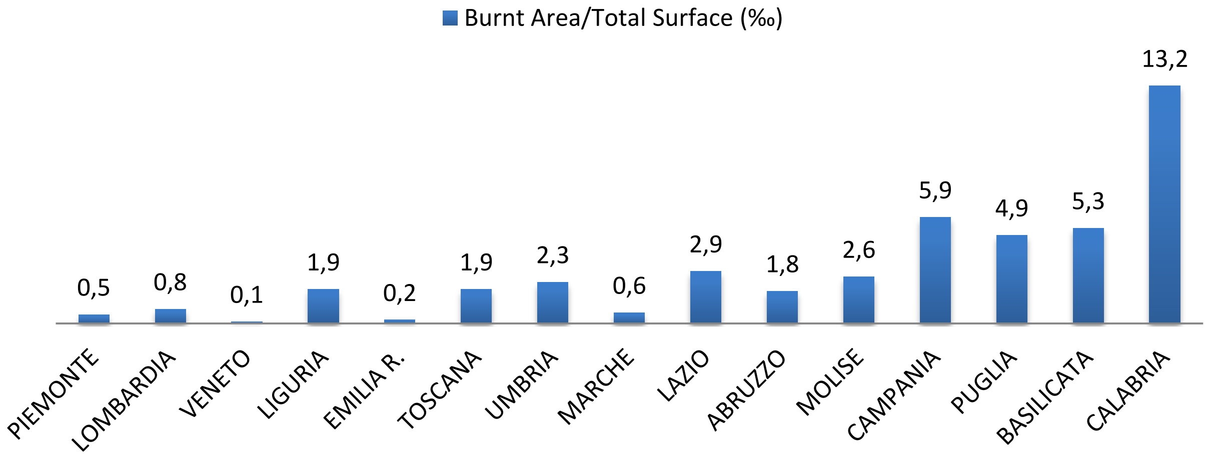
Table 5. Number of forest fire crimes, average burnt area and environmental impact in Italy in 2012

Region	Number of fire crime	Average Burnt Area (ha)	Environmental Impact (ha)	E. Impact/Total Surface (‰)
PIEMONTE	118	8.3	979.4	0.5
LOMBARDIA	228	5.1	1,162.8	0.8
VENETO	117	0.9	105.3	0.1
LIGURIA	272	3.7	1,006.4	1.9
EMILIA R.	135	3.0	405	0.2
TOSCANA	627	3.7	2,319.9	1.9
UMBRIA	147	13.2	1,940.4	2.3
MARCHE	62	3.9	241.8	0.6
LAZIO	568	11.3	6,418.4	2.9
ABRUZZO	125	9.9	1,237.5	1.8
MOLISE	98	6.6	646.8	2.6
CAMPANIA	972	6.8	6,609.6	5.9
PUGLIA	512	14.8	7,557.6	4.9
BASILICATA	322	17.9	5,763.8	5.3
CALABRIA	943	21.1	19,897.3	13.2

Source: authors' elaboration from Italian Forest Corp data

In Table 6 it is possible to observe that the regions with the highest number of fire crimes were Campania (27% of the national amount, corresponding to 972 fires) whose burnt area is about 6,600 ha, Calabria (21%) with 943 fires and an environmental impact amounting to 19,897 ha and Puglia (15% of the total, equal to 512 fires) with an affected area equivalent to 7,557 ha. It is worth noting that the environmental impact arising from fire crimes seems to be quite negligible in some north Italian regions. In particular, during 2012, intentional fires burnt 0,1% (Veneto) and 0,2% (Emilia Romagna) of the total regional surface (see figure 12).

Figure 12. Extent of the evironmental impact considering regional total surface



Source: authors' elaboration from Italian Forest Corp data

The environmental impact of forest fire crime seems to be extremely concerning in those areas designed for the protection and maintenance of the ecological diversity through legal or other effective measures. Protected areas, including any state forest, national park and protected public land, are increasingly threatened by forest fire in Italy. Fortunately, from the estimates derived so far about the environmental impact of forest fire crimes, latest data form NIAB allow us to exactly quantify the magnitude of the environmental impact on protected areas, with regard to the ordinary statute regions served by Italian Forest Corp, by summing the burnt area (ha) arising from every single fire crime event. In particular, in 2012, the occurrence of forest fires in protected areas such as national parks and public protected land totaled 793 fire crimes that covered about 8,864 hectares (Table 6).

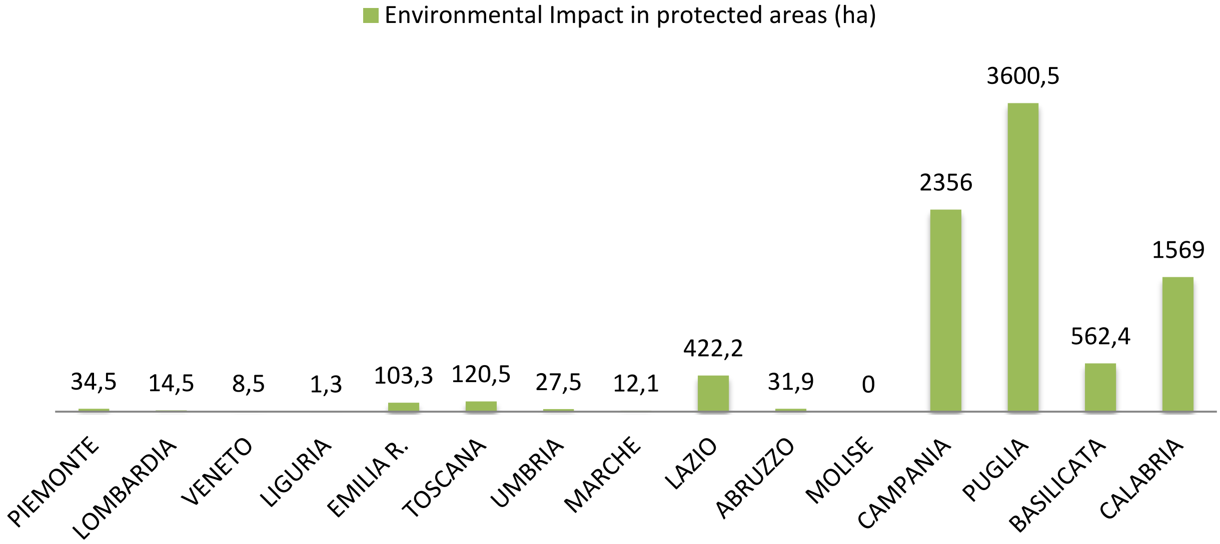
Table 6. Number of forest fire crimes and environmental impacts in protected areas in Italy in 2012

Region	Number of fire crimes	National Parks	Other public protected land	Environmental Impact
PIEMONTE	6	-	6	34.5
LOMBARDIA	18	1	17	14.5
VENETO	18	4	14	8.5
LIGURIA	6	3	3	1.3
EMILIA R.	18	2	16	103.3
TOSCANA	28	3	25	120.5
UMBRIA	8	2	6	27.5
MARCHE	6	2	4	12.1
LAZIO	55	4	51	422.2
ABRUZZO	8	5	3	31.9
MOLISE	2	1	1	0
CAMPANIA	303	171	132	2,356
PUGLIA	186	99	87	3,600.5
BASILICATA	51	42	9	562.4
CALABRIA	80	70	10	1,569
TOTALE	793	409	384	8,864.2

Source: authors' elaboration from Italian Forest Corp data

During 2012, the most affected regions in terms of protected area burnt by fire crimes were Campania and Puglia. The number of fires were 303 in Campania with a total environmental impact of 2,356 ha, and 186 in Puglia, where the environmental impact amounts to 3,600.5 ha, 2,022.2 of which were in national parks. In figure 13, it is worth noting the huge difference between northern and southern regions: Campania, Puglia, Basilicata and Calabria total about 95% of the whole national environmental impact in protected area due to fire crime.

Figure 13. Evironmental impact in protected areas (2012)



Source: authors' elaboration from Italian Forest Corp data

3.5 Health impact

To estimate the health impacts of forest fires in Italy, two variables are taken into account; namely number of deaths and injured people. Table 7 below displays in detail the number of victims of forest fires in Italy over the decade 2003-2012, disregarding fire causes. Overall, the loss of human lives amounts to 55 people with 442 injured. The most dramatic season was recorded in 2007 with 23 deaths and 26 injured.

Table 7: Forest fire victims in Italy (2003-2012)

Years	Injured	Deaths
2003	75	7
2004	35	2
2005	43	3
2006	17	1
2007	26	23
2008	30	4
2009	12	4
2010	55	3
2011	92	4
2012	57	4
Total	442	55

Source: authors' elaboration from Italian Forest Corp data

Table 8 below indicates number of forest fire crime victims (injured and deaths) recorded in Italy during 2012 with regard to the ordinary regions.

Table 8: Fire crime victims in Italy by region (2012)

Region	Number of fire crimes	Injured	Deaths
PIEMONTE	118	2	1
LOMBARDIA	228	3	-
VENETO	117	-	-
LIGURIA	272	2	-
EMILIA R.	135	2	1
TOSCANA	627	5	-
UMBRIA	147	4	-
MARCHE	62	1	-
LAZIO	568	2	-
ABRUZZO	125	12	-
MOLISE	98	1	-
CAMPANIA	972	9	1
PUGLIA	512	2	-
BASILICATA	322	5	-
CALABRIA	943	3	1
Total		53	4

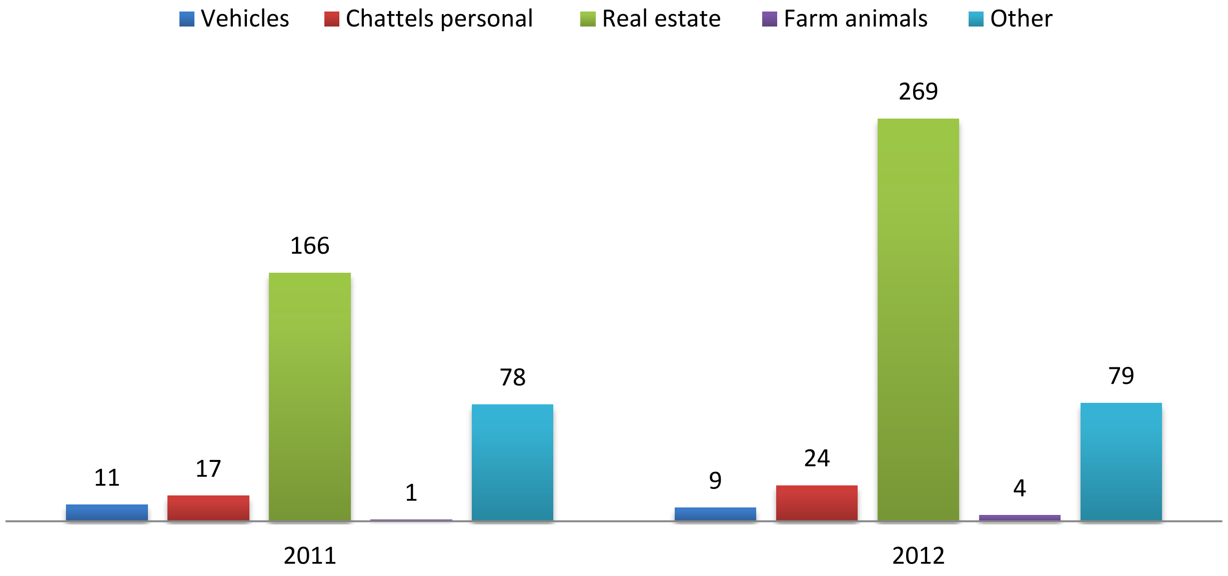
Source: authors' elaboration from Italian Forest Corp data

The number of victims of forest fire crime, in 2012, concerning only the ordinary statute regions served by the Italian Forest Corp, was slightly lower compared to other years: there were 4 deaths and 53 injured, the latter probably underestimated, since it refers only to cases reported in the official survey of the Italian Forest Corp. The deaths occurred in Piemonte, Emilia Romagna, Campania and Calabria, in the provinces of Torino, Modena, Napoli and Cosenza. While, the regions that have a greater number of injured are in order: Campania (14), Toscana (8), Umbria (7), Liguria and Basilicata (5), Calabria (3).

3.6 Material impact

To assess the material impact of forest fire crime, relying on Italian Forest Corp data, we consider five main categories of possible damages arising from fire events: (i) *vehicles*; (ii) *chattels personal*; (iii) *real estate*; (iv) *farm animals* or other damages. The available data allow us, in contrast to other investigated impacts, to quantify material impacts, both at country and region level, only for the years 2011-2012. Figure 14 below shows how many forest fires gave rise to such impacts.

Figure 14. Material Impacts (2011-2012)



Source: authors' elaboration from Italian Forest Corp data

Although the number of fire crimes that occurred in the country in 2012 (5,246) decreased with respect to 2011 (5,296), the total number of fire events that caused material impacts on the aforementioned categories significantly increased from 273 to 385 fires. This is particularly evident with regard to real estate, which was much more affected by fire crimes (166 in 2011 vs. 269 in 2012).

Table 9 below displays in detail the extent of the economic and material impact in 2012, looking at the different categories of affected goods for each ordinary statute region.

Table 9. Number of fire crimes causing material impact by regions (2012)

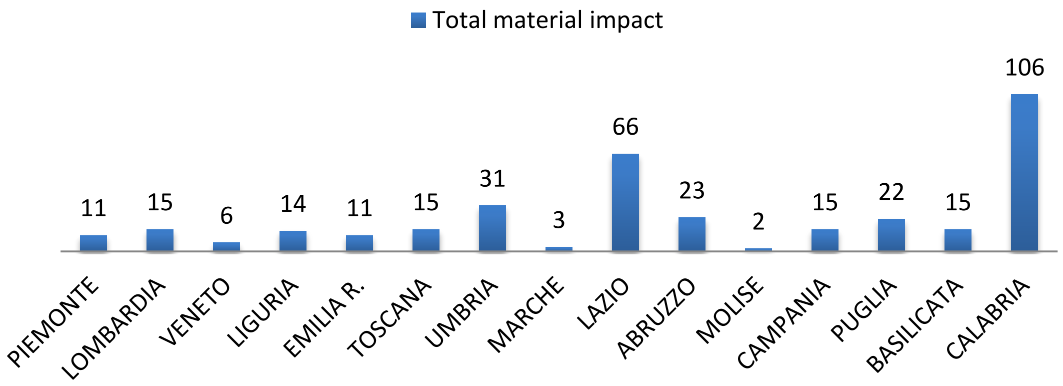
Region	Vehicles	Chattels Personal	Real Estate	Farm Animals	Other
PIEMONTE	7	2	2	-	-
LOMBARDIA	1	1	12	16	1
VENETO	-	3	3	-	1
LIGURIA	1	3	5	-	5
EMILIA R.	-	2	9	12	-
TOSCANA	1	2	3	2	7
UMBRIA	-	1	27	1	3
MARCHE	-	-	1	1	2
LAZIO	-	11	55	-	1
ABRUZZO	-	-	15	-	8
MOLISE	1	1	-	-	-

CAMPANIA	2	1	27	12	4
PUGLIA	1	4	19	1	5
BASILICATA	-	2	6	1	6
CALABRIA	1	2	88	7	15

Source: authors' elaboration from Italian Forest Corp data

As it is possible to see in figure 15, the regions particularly affected are Calabria and, surprisingly, Lazio: in Calabria as many as 106 fire crimes led to economic damages, mainly on real estates (88); in Lazio fire crime events totaled 66, 11 of which destroyed personal chattels. Conversely, less affected regions were Marche and Molise, with respectively 3 and 2 fires that gave rise to material consequences during 2012.

Figure 15. Number of fire crimes with material impacts



Source: authors' elaboration from Italian Forest Corp data

3.7 Investigation and enforcement activities in Italy

In the fight against fire, which every year also devastates parks and natural reserves, the Italian Forest Corp has started an operational activity that could soon be effective. In fact, the solution to the issue does not lie in dispensing punishment, which in 2000 was punishable with 10 years prison, rather, it is "a matter of civic-mindedness, of sharing the will to protect the common good: the forest."

Fires are very costly, both for the environment and for the State. Every year, each Italian taxpayer unwittingly invests about € 60-70 of his/her taxes in the protection system for the active fight against forest fires, which is a significant social cost. For a year now, the Italian Forest Corp has undertaken an initiative to charge those who, negligently or willfully, set fire to our beautiful forests. The charge would correspond to the cost of the environmental damage, which could be really expensive.

Evaluating the environmental damage, a fire of 40-50 acres of forest can reach figures greater than € 250,000. The exact amount depends on the quality of the forest. Moreover, there is also a cost related to the mission to contain the fire. The Canadair and the helicopter have a considerable per hour cost and the fire tanker truck is expensive as well. A fire extinguished with the involvement of an airplane, can easily cost more than € 30,000 which is only the expense for the extinction.

Quantifying the value of the forest patrimony in the country is not easy. As a matter of fact, the protection of forests and the safeguard of environmental integrity coincide with the protection of human life on the planet and to attribute an economic value to this is difficult and complex. In order to deal effectively with forest fire crimes it is necessary to adopt an integrated investigative approach, which entails various types of analysis, including: (i) analysis of the causes of the fires, (ii) analysis and investigative psychology, (iii) criminological analysis (profiling), and (iv) socio-economic analysis, but also an interdepartmental approach involving the Italian Forest Corp, the intelligence services, local authorities, the National Body of Fire Brigades, the Police, and the Civil Protection services.

The Italian Forest Corp, in fighting against fire crimes, has given impetus to both the central organization and outstations, through the NIAB. This was established in 2000 by the Inspectorate general and operates throughout the national territory, with the exception of the regions with special statutes and the autonomous provinces. The NIAB is responsible for coordination and direction of information investigation and analysis in relation to forest fires and provides operational, investigative and logistical support to the territorial offices of the Italian Forest Corp, also through its research of evidence collected at the scene of fires and the analysis of residues of explosives and triggers (JRC Technical Report, 2012). Article 423-bis of Italian Penal Code (Incendio boschivo), introduced by Law 353/2000, has in recent years improved the assessment of the reasons underlying the wildfires in order to understand and analyze in depth the phenomenon of forest fires and, consequently, to introduce effective legal instruments for carrying out the investigations.

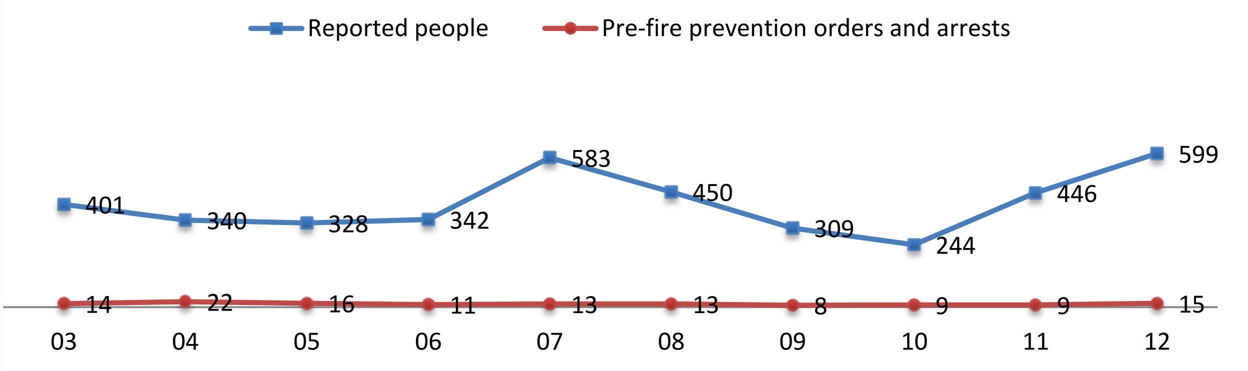
Actions against forest fire offences undertaken by the territorial Italian Forest Corp in 2012, made it possible to report 614 people to the judicial authority, of which 563 were for negligent fires and 51 for arson (table 10). Of these, 15 people were arrested, pursuant to custodial measures for arson, while 579 were released on caution (table below).

Table 10: Number of forest fire crime complaints in 2012

Cause	Numbers	Percentage
Negligence	563	91.7
Deliberate	51	8.3
TOTAL	**614**	**100**

Source: authors' elaboration from Italian Forest Corp data

Figure 16. Number of reported people and pre-fire prevention orders in Italy (2003-2012)

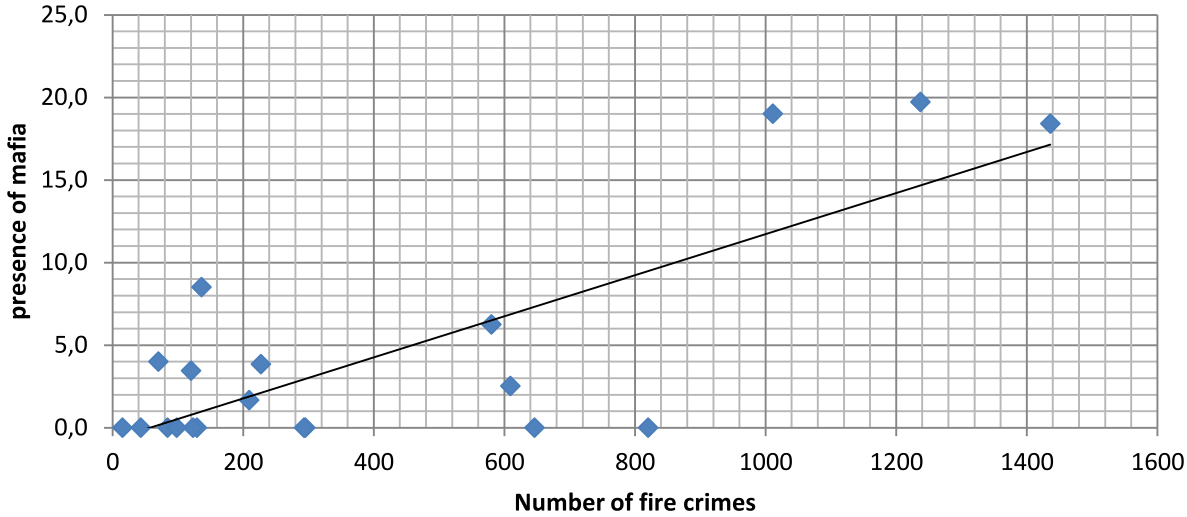


Source: authors' elaboration from Italian Forest Corp data

In total, over the period 2000-2012, over 5,000 people were reported to the judicial authority for forest fire offences, of which 164 were arrested in the act or were subjected to pre-fire detention orders. Analyzing the judgments issued by the judicial authority and collected by NIAB since 2000, it is revealed that the judicial process ended with the conviction of responsible in 45% of the reported cases, while the remaining with absolution (13%) or archiving (42%).

The Italian environmental group Legambiente (2010) believes that more than half of all Italy's fires are started deliberately, whether by organized crime, building speculators or farmers seeking more land to cultivate. It is interesting to note, from a more in depth analysis based on our estimation using 2012 data, how evidence has emerged of a positive correlation between organized crime (i.e., mafia-like organizations) and number of fire crimes (Figure 17). The grip of organized crime seems to be stronger in several of Italy's southern regions, right where the government's ability to enforce the law there is correspondingly weaker.

Figure 17. Correlation between organized crime and number of fire crimes



Source: authors' elaboration and estimation.

However, according to the Italian Forestry Corp (Forest Fires Report – various years) mafia-like organizations have nothing or very little to do with forest fire crimes. Organized criminals responsible for deliberate fires can represent isolated and sporadic actions. The most important motives behind deliberate fires in Italy (Tedim et al., 2014) are: (i) profit activities to obtain goods, jobs or even money (e.g. hunting products in areas scorched by fire passage such as mushrooms and wild asparagus; vegetation burning to earn agricultural land; fire caused with the intent of being included in fire-fighting crews), (ii) social and interpersonal tensions (e.g. hunting conflicts, ownership controversies), (iii) negligent behaviors (e.g. fire-crackers and bottle-rockets), and (iv) fire as a means of protest against public powers (e.g. retaliation against public administration and protest of seasonal fire-fighters).

In fact, the marginal role of mafia-like organizations in the context of forest fire crimes has been investigated by the Italian Forestry Corp, which conducted, on behalf of the government, an exploratory investigation on the causes of fires in Italy since 2001. This allowed, for the first time, studying the causes of forest fires and working on their classification in an organic and systematic manner, nationwide. To this end, it has been used as a scientific method for the detection of the causes of the onset of the forest fire (called Method of Physical Evidence)[5]. The

[5] This method shall be understood as a procedure, a technique, a sequence of steps that allows us to reconstruct the evolution of a fire through the study of its behavior and the traces it left up to the determination of the point of origin, and therefore to the identification of the cause of the fire, of its initiators and its reasons. The method is characterized by a complex procedure, which unfolds in several stages: i) determination of the geometry of the fire; ii) reconstruction of the evolution of the fire; iii) the definition of the path of propagation and detection of the

result is that, compared to a negligible percentage of natural and accidental causes (around 1.6%), there is a significant figure relative to fires caused by people: among these fires, the large majority (almost 60% of the total) seemed to be deliberate. In particular:

- In 70.4 % of cases the motivation for the arsonist to start a fire was the pursuit of financial gain;
- In 25% of cases it is *resentment* towards measures enacted by the authority that manages the burnt areas (e.g. protected areas);
- The remaining 4.6% of undefined deliberate causes are definitely related to voluntary acts, but the aim pursued by the author cannot be classified with certainty, for the lack of precise and objective evidence.

4 The monetary impact of fire crime

Forest fires represent a calamity that significantly affects wild-lands worldwide with its related impacts. Nevertheless, there is relatively little information on the actual monetary damage resulting from forest fires. An appropriate monetary analysis concerning forest fires could represent an effective instrument to prevent and mitigate the loss of biodiversity. It is crucial to justify and to select the most effective forest fire management measure in order to minimize the different impacts of wildfires as much as possible.

In the literature, the analysis on the estimation of the damage is mainly focused on private goods. However, recently, some authors have paid specific attention to damages from fires in wooded areas, bringing forth some case studies, which hardly account for the complexity of the issue. Very few are the studies that focus on the public profile of the damage (see among others Marangon and Gottardo, 2001 and Valese et al., 2011), looking at estimating the environmental damage of this delicate issue.

As previously emphasized in the methodological section, the Italian Academy of Forest Fire and the Italian Forest Corp (2007) provided a methodological framework for the economic monetization of specific impacts arising from wildfire. In particular, according to this study (Ciancio et al., 2007), the total impact is divided into three components: *extinction cost*, *environmental damage*, and *external damage.* The estimate of such components is based on the calculation of the Total Economic Value (TEV) of the forest areas in question employing

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ignition area; iv) search for the ignition point; v) search for the evidence relating to the cause; vi) comparison between physical evidence and witnesses' statements; and vii) classification of the cause of the fire.

different approaches, analytics or synthetics, based on the characteristics of the forest fire (i.e. fire size, type of fire - wooded or not wooded, etc.). By definition, the damage arising form forest fires is peculiar because of its mixed nature of forest as goods, of which the concurrence of both public and private damage occurs. In addition, there are also other peculiar aspects connected with the presence of multiple and joint productions, the wide time span between the initiation and the use of the stand, the different structure of the stands (on the base of the age, articulated, etc.) as well as the possible existence of complementary relations between the areas affected and those not affected by the fire, and so on and so forth.

For the sake of clarity, in the following sections, we present three different case studies, concerning the estimation of the monetary impact of forest fires, as examples of different methodological approaches that have been considered reliable by the Italian courts in criminal proceedings.

4.1 The case of the Morfasso forest fire in Favale

The forest fire considered here took place on the 22nd and 23rd July 2010. It destroyed over 8.5 ha of woods above Mignano dyke (Piacenza). It was a large-scale fire and required the intervention of fire fighter helicopters and Canadair airplanes for its suppression. The Piacenza Provincial Command of the Italian Forest Corp reported, as the alleged responsible authority for the fire, two workers from Romania who were employed in an agricultural cooperative. They were performing forest-cleaning operations on behalf of the Mountain Community Valnure and Valdarda.

During the investigations, which were carried out immediately after the fire through the use of the Physical Evidence Method (M.E.F.), the Italian Forest Corp found traces of several fires along the path that ran along the forest. One of these that had been lit on July 22nd, when a strong wind was blowing, was not adequately supervised. It was determined that the flames started from there and in no time invaded the slopes expanding toward the village of Favale, which fortunately was not reached by the flames.

The essential elements that characterized the fire are summarized in table 11 below:

Table 11. Favale of Morfasso forest fire

Region	Emilia Romagna
Province	Piacenza
Municipality	Morfasso

AIBFN	N°3/2010/3775
Affected Area	8.5 ha
Fire cause	Negligence
Type of affected forest	oaks, ash trees, manna ashes and black pine forest
Beginning of fire	22rd July 2010
End of fire	23rd July 2010
Utilized vehicles	2 aircrafts, 6 extinguishing trucks
Material damages	None
Injured and / or deceased	None

Source: authors' elaboration from Italian Forest Corp data

4.1.1 Estimation method

The estimate is based on the calculation of the TEV of the forest areas under investigation following the analytical approach and looking at the three components of the monetary impact: *fire extinction costs, environmental damage, and external damage.*

- **Fire extinction costs** (or suppression costs) are costs relating to machines and personnel's equipment used during the operation of active fire fighting. According to the *AIBFN* report N°3/2010/3775 of the Italian Forest Corp, we are able to quantify the total cost of intervention. Particularly, two Canadair airplanes were used in the fire fighting activities on the 22nd and 23rd July 2010, in Favale of Morfasso forest fire. One CAN 20 for eight hours and dropping of fire suppressing foam eleven times for a duration of one minute and a one time use of fire retardant; one CAN 2 for seven hours and forty six minutes, dropping fire suppressing foam eleven times and fire retardant one time. This totals fifteen hours and forty-seven minutes, twenty-two fire suppressing foam drops and two fire retardant drops. Taking into account the duration of the extinction operations and the cost of the equipment used the total cost of the intervention is € **100,504.54** (Table 12)**.**

Table 12. Favale of Morfasso forest fire extinction costs

Authority working to suppress the fire	Overtime hours	Cost of the missions	Fuel/ Extinguishing or retardant agents	Hourly cost for flight/ rent/ labor
Civil Protection Department (COAU[6])			€ 5,233.00	€ 68,445.58
Italian Forest Corp (P.C.[7] Provincial Command)	€ 1,393.34	€ 244.64	€ 142.38	
Piacenza's Fire fighters	€ 383.33		€ 196.96	
Civil Protection AIB Volunteers (P.C.)		€ 561.00	€ 632.31	
Morfasso's Municipality				€ 7,272.00
VV.FF.[8] Bologna Flight Department				€ 16,000.00
TOTAL	**€ 1,776.67**	**€ 805.64**	**€ 6,204.65**	**€ 100,504.54**[9]

Source: authors' elaboration from Italian Forest Corp data

- ***Environmental damage***: a conventional approach for an analytical estimation of the environmental damage can be made by determining the cost of reconstruction or restoration. The estimation criterion is based on the assumption that an asset is worth (at least) what it costs. The damage was evaluated through an analytical approach based on two functions of the forest: (i) *Wood production loss* (PPL) and (ii) *Decreased hunting activity* (DAven). The estimate of the PPL equals the sum of the

[6] COAU = Unified Aircraft Operative Center

[7] P.C.= Piacenza

[8] VV.FF. = Fire fighters

[9] The ordinary cost of the Institutional Staff is not included.

economic damages (DE) suffered by the coppice forest (c), the tree trunk forest (a), and the mass of the harvested timber, therefore

$$PPL = DEc + DEa + DEle$$

The values are given by the following functions:

For the oak coppice:

$$DEc = Sup * Vol * \left(\frac{Pimp - Cte}{(1+r)^n}\right) = 6.7425 * 13.82 * \frac{(85 - 20)}{(1 + 0.03)^{15}} = € \mathbf{6056.79}$$

where *DEc* is the Economic damage derived from the loss of firewood of the coppice forest, *Sup* is the coppice area affected by the fire (in hectares), *Vol* represents the volume of marketable firewood lost due to the fire (in m^3/ha), *Pimp* represents the average price of the pallet, *Cte* are the cutting and skidding costs, *r* is the discount rate and n represents the remaining years to reach the age of customary round.

For the black pine tree trunk

$$DEc = Sup * Vol * \left(\frac{Pimp - Cte}{(1+r)^n}\right) = 1.4635 * 48.32 * \frac{(33{,}60 - 30{,}00)}{(1 + 0.03)^{15}} = € \mathbf{396.60}$$

For the oak firewood

The damage loss is quantified in:

$$DEle = (Mt * Vimp) + (Mp * Vmac) = (117.86 * 7{,}5) + (23.17 * 6.5) = € \mathbf{1034.55}$$

where *Mt* represents the mass in quintals of lost wood; *Vimp* is the value of the piled wood in the accessible area on the forest track; *Mp* represents the mass in quintals of the root sprouts compromised by the fire and *Vmac* is the stumpage value, that is cutting and skidding costs before taxes.

The total environmental damage from the loss of timber production amounts to:

$$\boldsymbol{PPL = DEc + DEa + DEle == 6056.79 + 396.60 + 1034.55 = € 7487.94}$$

The second forest function to evaluate the environmental damage is the *decreased hunting activity* (DAven) (ungulates and local fauna). Given that the Law 353/2000 article 10 paragraph 1 prohibits hunting for 10 years in forest areas where topsoil was crossed by fire, the damage can be estimated as the sum of ten yearly values of the annual hunting function, at the discount rate of 3%.

The calculation function is therefore:

$$DAven = Sup * R * \frac{(1 * r)^{10} - 1}{r * (1 * r)^{10}} = 8.2814 * 7.45 * 8.5302 = € \mathbf{526.28}$$

where *Sup* is the area crossed by the fire and excluded from the hunting activity (in hectars); *R* represents yield per unit of the hunting area for the ATC PC 7 (Piacenza's hunting zone) (euro/hectare).

- **Extraordinary external damages.** The estimate of extraordinary external damages only takes into account the cost of reconstruction of the destroyed and damaged topsoil. As a matter of fact, there are no infrastructures or buildings in the area, nor has damage occurred to people or mechanical means. The current price list for the regional forest works' public initiative (Resolution 2085 of 20th December 2007) was taken as a reference for the typologies of forest works to be designed and implemented. Unit prices (before taxes) are applied in relation to the damaged areas and to the level of damage calculated according to the Method of the Observable Effects. The payment of damages for the cost of forest regeneration is quantifiable as follows:

  $$Dcr = (CR * Sc * LD) + (Cte + Sa) + (Ctab * S) = (2702.90 * 6.7425 * 0.09) + (4157.32 * 1.4635) + (102.36 * 8.2814) = € \mathbf{8572.10}$$

  Where *CR* is the cost of regeneration of the oak coppice forest through selective thinning by elimination of damaged, cut off, decaying or dried root sprouts, including cutting off branches, splitting wood, and piling of resulting material, cleaning up the adjacent outer and inner tracks (item 41 regional price list) (euro/hectare); *Sc* represents the Coppice Area; *LD* is the level of damage, estimated by the ratio between mass loss (13.82 m^3/ha) and existing medium mass (149 m^3/ha), by applying the Method of the Observable Effects similarly to the estimate of the economic damage for loss of firewood (DEc); *Cte* is the cost for pine cutting and skidding, 20-30 cm in diameter, including branch pruning, wood sawing, piling, transportation of the wood to the pallets and removal of the branches by using tractors (items 45 + 47 + 49 of the regional price list). In relation to the mass loss of 48.32 m^3 in tree trunk, and to the average volume - m^3 0.327/plant, the dead standing or cut plants to be removed are estimated at €148/hectare; *Ctab* is the cost of supply and installation of warning signs (40x35 cm) on wooden poles (estimated 3/hectare) (item 76 of the regional price list) (in euro/hectare); *S* is the total area of the fire to be posted with no hunting signs (hectares).

After having analytically estimated the three components of the forest fire monetary impact we are able to exactly quantify it. To this end, the table below (13) shows a summary of the monetary estimated impact for the Morfasso forest fire that occurred on July 22nd and 23rd, 2010:

Table 13. Summary of the monetary impacts

Costs for fire suppression	€ 100,504.54
Environmental damage (services and goods)	€ 8,012.22
Extraordinary external damage (cost for forest regeneration)	€ 8,572.10
Total monetary impact	**€ 117,088.86**

Source: authors' elaboration from Italian Forest Corp data

4.2 The case of the Maracallo forest fire

For a better understanding of the proposed methodology in assessing the monetary damage from forest fire (Ciancio et al., 2007) we present a second case study concerning the fire event that occurred in Monte della Croce, municipality of Maracallo (Va) on April 22nd, 2011. Such a case study relies on the approach of standard costs (personal and related equipment) for the assessment on the extinction costs and the environmental damage. Moreover, the extraordinary external damages were not considered because the forest fire did not affect physical assets or people's health. The main points of information about the investigated fire are listed in table 14 below:

Table 14. Maracallo forest fire

Region	Lombardia
Province	Varese
Municipality	Maracallo
AIBFN	N°4/2011/7757
Affected area	1.49 ha
Fire cause	Negligence
Type of forest affected	Chestnut and pine
Beginning of fire (time)	11:40

End of fire (time)	18:00
Personnel intervened	2 CFS, 5 VV.FF.
Voluntaries intervened	30 voluntary AIB
Utilized vehicles	2 helicopters, 6 extinguishing trucks
Material damages	None
Injured and / or deceased	None

4.2.1 Estimation method

Contrary to the previous case, given the limited fire size of the event, we considered it appropriate to employ parameterized models for synthetic quantifications, with reference to standard costs reported in the Italian Forest Corp technical document and/or from other official sources (D.s. 10-06-2011 n. 5256 Rural Development Program 2007- 2013 (Reg. CE 1968/2007) "*Approval standard costs measure 226 Restoring forestry potential and introducing prevention actions*").

- **Fire extinction costs** can be estimated by knowing the average personnel cost per hour, the number of people employed, possibly divided into two categories (voluntary and contract fire-fighters) the duration of the extinction operations and the cost of the equipment used. Clearly, in the case of intervention of voluntary fire fighters, they should not be considered for the calculation of personnel costs, but must be included for the quantification of the cost of equipment. Usually, the Italian Forest Corp collects all this information for each fire event in Italy. Specifically, two helicopters Eurocopter Ecureuil AS 350 B3 and six extinction trucks were used in the fire fighting activities on 22nd April 2010 in the Maracallo forest fire. The table below shows in detail the equipment used and the related amount of time. Taking into account the duration of the extinction operations, the cost of the equipment used and the fire-fighters employed, the total cost of the intervention is **€ 21,307.98.**

Table 15. Maracallo forest fire extinction costs (standard costs)

Equipment and fire-fighters	Number	Hours	Hourly cost (€)	Adaptation rate (ISTAT)	Total monetary impact (€)
Extinction trucks	6	6	100	1.117	4,021.20

Helicopters	2	4	2,000	1.000	16,000
CFS Fire fighters	2	6	18	1.117	402.12
VVFF Fire fighters	5	2	18	1.117	482.54
TOTAL					**€ 21,307.98**

Source: authors' elaboration from Italian Forest Corp data

- **Environmental damage** is estimated using the conventional approach that focuses on the cost of reconstruction or restoration. Basically, the economic estimation of the environmental damage corresponds to the product of the following factors: *reconstruction cost*, *affected area* and *level of damage* due to the forest fire. The cost of reconstruction was determined with reference to the standard costs identified by costs measure *226 "Restoring forestry potential and introducing prevention actions*" implemented by the Lombardy Region. In particular the initiative A.1.4.6 - *Recovery and reconstitution of forests damaged by natural disasters and fire*: Class 1 of operational difficulty, density from 701 to 900 plants/ha, damaged plants from 26% up to 50%; €11,903.33 per hectare (value refers to the year 2011). The forest fire affected area, determined through GPS, was exactly 1.49 ha. The level of damage, assessed on site by Italian Forest Fire experts, has been recognized equal to 50%. Finally, in order to take into account the time needed for the forest to reach pre-fire conditions we postponed the reconstruction cost of 20 years with a discount rate of 2%. The total amount of environmental damage amounts to €27,143.72, as shown in table 16 below.

Table 16. Maracallo forest fire environmental damage (standard costs)

Measure 226	Area (ha)	Reconstruction cost	Adaptation rate (ISTAT)	Total monetary impact (€)
A.1.4.6	1.49	6	1.30	18,267.53
		Discount rate	Years	
Forest regeneration cost	18,267	0.02	20	**27,143.72**

Source: authors' elaboration from Italian Forest Corp data

Sometimes even a fire of modest size with limited environmental damage can result in significant costs associated with the partial or total destruction of tangible assets (civil infrastructures, settlements, agricultural crops, etc.). However, in the forest fire area, no damage to infrastructure, buildings, people or vehicles occurred.

The table 17 below shows a summary of the monetary estimated impact for the Maracallo forest fire that occurred on April 21st, 2010:

Table 17. Summary of the monetary impacts

Costs for fire suppression	€ 21,307.98
Environmental damage (services and goods)	€ 27,143.72
Extraordinary external damage (cost for forest regeneration)	€ 0.00
Total monetary impact	**€ 48,451.71**

4.3 The case of the Rocca Romana (Trevignano) forest fire

Finally, we present a further analytical approach based on the economic assessment of forest fire damage relating to the loss or reduction of the different utility functions (i.e. economic, social and environmental) provided by a forest area. To this end, we take into consideration the forest fire crime that occurred in Rocca Romana, municipality of Trevignano Romano (RM) from August 7th to 10th 2003. Table 18 shows the main information that characterized the fire:

Table 18. Rocca Romana (Trevignano) forest fire

Region	Lazio
Province	Rome
Municipality	Trevignano Romano
AIBFN	N°2/2003/1245
Affected Area	22 ha
Fire cause	Negligence
Type of affected forest	Oaks, chestnuts, hornbeams
Beginning of fire	7th August 2003

End of fire	10th August 2003
Utilized vehicles	2 Canadair CL-415, 1 Helicopter AIB412
Material damages	None
Injured and / or deceased	None

Source: authors' elaboration from Italian Forest Corp data

4.3.1 Estimation method

In contrast to previous forest fires, given the large size of the event and the heterogeneity of the affected area, we considered it appropriate to employ the estimate of the different forest functions for the calculation of the TEV. This approach proposed by Ciancio et al. represents the most accurate and articulate method for the evaluation of the monetary impact of forest fire. In particular, the environmental damage rests on the appraisal of seven forest functions: *(i) wood production loss; (ii) non-wood production loss; (iii) tourism-recreation loss; (iv) hunting activity loss; (v) soil protection; (vi) protection from climate change; and (vii) biodiversity protection.*

The total value of environmental damage results from the sum of the aforementioned seven functions. However, the identification of the seven components of the damage does not imply their contextual involvement in each forest fire event. It is, in fact, unusual that a wildfire produces, for example, both significant damages to the hunting activities of the forest and biodiversity. It is worth noting that, in this case, the hunting function has to be ignored since the Italian law forbids hunting activities in protected areas. Therefore, appraisal of the environmental damage, in this specific case, will take into account the following forest functions: i) the wood production, ii) the non-wood production (in this case it refers to, among others, the collecting of mushrooms), iii) the tourism-recreation and iv) the protection from climate change.

For the estimation of ***wood production*** **(WP)**, we recognize that**:**

$$WP = \frac{M}{n} * Pz$$

where M is the average forest mass per hectare; n represents the average age of the forest and Pz the average price of stumpage.

The average forest mass per hectare (M) was estimated by the weighted average (i.e. number and size of trees) of the data obtained for each hectare of the affected area and by linking them with the 'volume tables' developed by the Italian Forest Corp for forests comparable to that of the

Rocca Romana. The average forest mass is about 230 m^3/ha. The average annual increase, given by M/n (where n, average age of the population, is equal to 35 years), is 6.6 m^3/ha.

Pz, which is the basic element of the whole economy of forestry production, represents the unit value of the mass in the raw state. It is obtained by subtracting from the end product market value, the necessary processing costs for the management of the raw material (i.e. cutting costs, transportation cost, cost of extraction, cost of insurance and so on). It was calculated by applying data derived from previous experiences in similar conditions and geographical location. Table 19 below shows in detail the different costs of stumpage:

Table 19. Different costs of stumpage for Rocca Romana forest fire

Operation	Hours	Cost (€)	Unit cost (€/m^3)
Cutting			
– Specialized worker	1	13,00	13,00
– Chainsaw	0.66	4.00	2.64
Collection and transport			
– Tractor	0.33	17.00	5.61
– Specialized worker	0.8	13.00	10.40
Indirect Costs			12.66
Total cost of transformation			**€ 44.31**

Source: authors' elaboration from Italian Forest Corp data

The price of stumpage, equal to the difference between the market value of the product and the cost of processing, is € 15.69/m^3 (€ 60.00/m^3 - € 44.31/m^3).

Therefore, the *wood production* (WP) amounts to:

$$WP = \frac{230}{35} * 15.69 = \text{ €}\,\mathbf{103.10}$$

For the estimation of **non-w*ood production* (NWP)**, it is interesting to verify the existence of *ad hoc* data on NWP (i.e. chestnuts, mushrooms, truffles, acorns) identified by ISTAT for the investigated area. Looking at mushrooms, it is estimated that the yearly average produced quantity is 25 kg per hectare, divided into 15 kg of mushrooms of the genus Boletus and 10 kg of the genus *Cantarellus, Russula, Lattarius*. The local average market price of the Boletus mushrooms sits at 14 €/kg and the value attributable to mushrooms belonging to other genres, which have no local market, equals 3 €/kg, the annual value of the benefit produced by mushrooms, in the forest under evaluation, is **€ 240** per hectare.

Another possible economic aspect of forests is tourism. The economic growth in the past 30 years and the increase in leisure time, combined with the degradation of the urban areas, have resulted in an increase in visits within the protected areas. The tourism function of forests is typically offered through walks, picnics, guided tours, and educational trips for school groups. The monetary appraisal of the recreational value of forests (tourism-recreation function) that generates positive effect on the economy of the local population (which here, however, we do not consider) is carried out using a plurality of methodologies: the cost of an individual trip, contingent valuation and the willingness to pay for a visit. The approach we consider in our study refers to the willingness to pay for a visit in the affected area and can be summarized by the following equation:

$$TR = \frac{v * dp}{Sup}$$

where v is the recorded number of trips per year in the area under evaluation and dp represents the estimate of willingness to pay per visit;

Data from the Touristic Information Point of the Trevignano municipality recorded 4,570 visitors in the whole forest area (200 ha). With regard to the value of willingness to pay for a visit in the forest, a sample survey has been carried out. The interviews show that, on average, a person is willing to pay about 4 € for a day in Rocca Romana forest, which would be the cost of a hypothetical entrance fee. Therefore, the tourism recreation function amounts to:

$$TR = \frac{4570 * 4}{200} = € \mathbf{91}$$

One of the most important functions of a forest is carbon sequestration. Over the past decades, forests have moderated climate change by absorbing most of the carbon released by human activities such as the burning of fossil fuels and the changing of land uses. Carbon uptake by forests reduces the rate at which carbon accumulates in the atmosphere and thus reduces the rate at which climate change occurs. The *protection from climate change* function (PCC) is estimated by looking at the economic value of the carbon immobilized by the forest ecosystems. This estimation is very complex and involves several methodological approaches. Our case study relies on the assumption that a forest represents a natural storage for the emissions of carbon. Therefore, we introduce the concept of *carbon tax* for a forest as the "shadow price" of its associated absorption benefit. This seems quite reasonable since, if we assume that the emission of carbon dioxide could be taxed, then the activities that have an opposite effect represent a social benefit. Hence, the PCC function is given by:

$$PCC = \frac{M}{n} * Xn * Xc * C$$

where M is the average forest mass per hectare, n represents the average age of the forest, Xn is the ratio Total biomass/above ground biomass (equal to 1.8), Xc is the biomass conversion factor m^3/t carbon (equal to 0.65) and C is the economic value of 1 ton of carbon based on the carbon tax estimation (average carbon tax 10 €/t). The PCC function is equal to:

$$PCC = \frac{60}{35} * 1.8 * 0.65 * 10 = € \mathbf{22}$$

Since the fire caused different degrees of damage in two different areas we decided to weight the considered forest functions with the level of destruction of the forest. In particular, data shows that 2 hectares were totally destroyed while 20 hectares were partially touched giving rise to a loss of about 25% of the biomass. Therefore, monetization of the environmental damage for each single forest function is:

- For WP function on 2 hectares: € 103 * 1 = € 103
- For WP function on 20 hectares: € 103 * 0.25 = € 25.75
- For NWP on 22 acres: € 245 * 1 = € 245
- For TR function of 22 hectares: € 91 * 1 = € 91
- For PCC on 2 hectares: € 22 * 1 = € 22
- For PCC on 2 hectares: € 22 * 0.25 = € 5

The total amount of environmental damage is equal to: [(103/0.03)* 2 +(25.75/0.03)* 20 + (240/0.03)* 22 + (91/0.03) * 22 + (22/0.03) * 2 + (5.5/0.03) * 20] = **€ 113.633**

Moreover, fire extinction costs can be quantified by knowing the average personnel cost per hour, the number of people employed, the duration of the extinction operations, and the cost of the equipment used. Clearly, in the case of intervention by voluntary fire fighters, they should not be incorporated into the calculation of personnel costs, but must be included in the quantification of the cost of equipment. Specifically, one helicopter AIB412 and two Canadair CL-415 were used in the fire fighting activities in the Rocca Romana forest fire. Table 20 below shows in detail the equipment used and the related amount of time and money.

Table 20. Rocca Romana forest fire extinction costs

Equipment and fire-fighters	Number	Hours	Hourly cost (€)	Total monetary impact (€)
Canadair	2	8	9,000	72,000

Helicopters	1	2	5000	10,000
CFS Fire fighters	7	12	80	6,720
TOTAL				**€ 88,720**

Source: authors' elaboration from Italian Forest Corp data

Taking into account the duration of the extinction operations, the cost of the equipment used and fire-fighters employed, the total cost of the intervention is **€ 88,720.**

Table 21 below shows a summary of the monetary estimated impact for the Rocca Romana forest fire that occurred from August 7th to 10th, 2003:

Table 21. Summary of the monetary impacts

Environmental damage (forest functions)	€ 113,633
Costs for fire suppression	€ 88,720
Extraordinary external damage (cost for forest regeneration)	€ 0.00
Total monetary impact	**€ 202,353**

Source: authors' elaboration from Italian Forest Corp data

5 Conclusion

5.1 Summary of the extent of the impacts

This report summarizes the current status of wildfire impacts, in both quantitative and monetary terms and, to some extent, whether the geographical impact of fire crimes is correlated to the presence of organized crime across Italian regions. It could represent the beginning of a dialogue on what data and knowledge are still needed to inform policy makers in order to better prevent and reduce negative economic, social and environmental impacts of forest fire crimes.

Due to data limitations, the analysis was carried out at two different geographical levels (European and Italian) and took into account different time spans. In particular, the European level analysis focused only on the environmental impact of fire crime while the regional level analysis (Italy) took into account the assessment of other impacts arising from fire crime (health, material and monetary) and the possible correlation between organized crime and fire crimes.

During the last decade, in the 21 Member States, forest fires due to human causes burned a total area of 1,535,572.41 ha. The magnitude of such devastation is particularly significant for the southern Member States. Spain was the most affected country in terms of burnt area with 55% of the whole burnt area in Europe (848,241.4 ha), followed by Italy (558,643.4 ha) and Portugal (72,838.3 ha). Conversely, less affected countries were Finland (14 ha), Belgium (52.7 ha) and Slovakia (98.5 ha). Given the lack of suitable information at a wider geographical level, the event-specific nature and the broad heterogeneity of this type of crime (i.e., protected areas, national parks, wooded/non-wooded area, etc.) the health, material and monetary impact of fire crimes were measured only at the Italian level. Data from NIAB reporting system show that over the decade 2003-2012, the number of human casualties due to forest fires in Italy amounted to 55, while 442 people suffered an injury. The most dramatic season was recorded in 2007 with 23 deaths and 26 injured. Moreover, although the number of fire crimes that occurred in the country in 2012 decreased (5,246) with respect to 2011 (5,296), the total number of fire events that caused material impacts on *vehicles*, *chattels personal*, *real estate* and *farm animals* significantly increased from 273 to 385 fires.

The article 423-bis Italian Penal Code introduced by Law 353/2000 has introduced more effective legal instruments for carrying out the investigations and analyzes the phenomenon of forest fire crime in depth. In this context, the actions against forest fire offences undertaken by the territorial Italian Forest Corp in 2012, made it possible to report 614 people to the judicial authority, of which 563 were charged with negligent fires and 51 for arson. Although, from a more in depth analysis, evidence has emerged of a positive correlation between organized crime (i.e., mafia-like organizations) and the number of fire crimes, looking at the judgments issued by the judicial authority, mafia-like organizations have a very marginal role in forest fire crimes. Organized criminals responsible for deliberate fires represent, actually, only isolated and sporadic actions.

The monetization of damages resulting from wildfires has been the subject of extensive analysis by various authors from the monitoring and reconstitutions aspects as well as the environmental and social impacts. In this respect, following the methodologies proposed by Ciancio et al. (2007), we focused on three different forest fire crimes that occurred in Italy, (Morfasso (VA) 22nd - 23rd July, 2010, Maracallo (PC) 22nd April, 2011, and Trevignano (RM) from 7th - 10th August, 2003) employing three distinct approaches (analytical, standard costs and forest utility approach) to determine the three components of the damage (i.e. *extinction* *cost*, *environmental damage*, *external damage*) owing to the heterogeneity of the investigated fire crimes.

5.2 Research needs and recommendations

The analysis of the impact of fire crimes in both quantitative and monetary terms presents several difficulties which relate to both methodological and data availability issues. From the methodological point of view, the impact assessment of forest fire crime has to take into consideration the fact that each fire produces several impacts (environmental, health, economic and social) that are very specific to the particular area where it occurs. Moreover, it is worth noting that fires directly impact benefits and resources that people receive from the environment, but only a minor part of them exhibit a market price that can be used as a possible proxy for assessing their value, while the majority of goods and services is not marketed. However, although EFFIS represents a useful and effective effort to collect data in a harmonized way among the Member States, it lacks adequate indicators to measure the economic, social and health impacts of forest fire, making results incomparable among different countries.

Our conclusions on data availability and methodological issues raise two important questions. First, would more harmonized data on wildfire impacts be a valuable tool for policy makers, and hence worth the costs of data collection? And, if so, how could we effectively carry out data collection, which information do we really need, and how can this knowledge be exploited to provide policies and practices to prevent and reduce negative economic, social and environmental impacts of wildfires?

Given the broad heterogeneity of forest fire impacts, the long time horizon for ecosystem recovery, and due to the trans-boundary nature of the events, the policy planning for their prevention should be seen from a regional perspective in order to discuss preventive measures, improve the international cooperation and identify opportunities for further collaboration among European Member States. This report highlights the availability of information that characterizes different forest fire impacts, but it does not consider the role of policy over time, for instance fire prevention or other management decisions that could affect the likelihood of forest fire and the extent of different impacts. Therefore, our findings may open the way for further investigations on the cost, effectiveness, and impacts of preventive management actions on forest fire crime, and contribute towards determining best practices in forest fire management.

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Annex A

Table 3: Assessing the environmental impact of forest fires in the 21 Member States from 2003 to 2012.

Nuts Code	Country	Number of fires 2003 - 2012	Burnt area 2003-2012 (ha)	Average burnt area per fire (ha)	Unknown causes	Known causes	Deliberate total	Crime % Deliberate/Known	Known causes %	Environmental Impact (ha)
BG	Belgium	4,742	925	0.19	1,527	3,215	270	8.3	67.8	52.7
CH	Switzer.	679	870	1.28	325	354	154	47.3	52.2	197.3
CZ	Czech Rep.	3,586	1,050	0.29	962	2,624	559	21.3	73.1	163.7
CY	Cyprus	1,617	6,123	5.02	208	1,409	280	19.8	87.2	1,198
DE	Germany	5,637	3,057	0.54	2,819	2,818	450	15.9	50.9	244
ES	Spain	163,818	1,556,386	9.5	23,752	140,066	89,282	63.7	85.3	848,241.4
FI	Finland	13,340	898	0.06	2,970	11,400	229	2.0	79.3	14
FR	France	42,501	172,557	4.06	21,379	21,122	6,033	28.5	49.7	24,494.4
GR	Greece	5,672	23,222	4.09	8,523	8,357	3,819	45.6	49.8	15,619.71
HR	Croatia	22,630	12,349	0.54	7,043	15,587	3,160	20.2	68.8	1,724.4
HU	Hungary	7,222	14,322	1.98	5,438	1,784	206	11.5	24.7	408.2
IT	Italy	72,597	798,305	11.613	19,250	73,347	48,105	65.6	73.4	558,643.4
LT	Lithuania	3,451	945	0.27	273	3,178	397	12.4	92.1	108.7
LV	Latvia	5,665	1,232	0.22	751	4,914	784	15.9	86.7	170.5
PL	Poland	6,328	29,004	4.5	4,564	1,764	1,288	57.2	84.8	5,903.5
PT	Portugal	245,334	999,984	3.8	188,931	56,403	17,870	31.6	22.9	72,838.3
RO	Romania	2,262	3,823	1.69	690	1,572	478	30.4	69.4	807.9
SE	Serbia	2,670	5,648	2.11	1,064	1,606	822	33.1	59.7	1734.4
SI	Slovenia	919	1,487	1.61	326	593	135	22.7	64.5	218.4
SK	Slovakia	2,612	875	0.33	264	2,348	294	12.5	89.8	98.5
TR	Turkey	12,731	24,246	1.90	5,809	6,922	1,413	20.4	54.3	2,691
TOTAL										**1,535,572.41**

Source: authors' elaborations on EFFIS database

Annex B

Table 4. Number of fire crimes, average burnt area and environmental impact in the five southern Member States over the last decade

	ITALY	SPAIN	PORTUGAL	FRANCE	GREECE	TOTAL
Number of Fire crimes						
2003	6,720	10,123	8,101	554		25,818
2004	4,823	8,402	6,657	355	290	20,527
2005	3,422	7,867	5,210	562	231	17,292
2006	4,238	8,723	3,212	455	180	16,808
2007	8,384	15,168	2,997	1,345	724	28,618
2008	4,250	12,123	9,990	567	423	27,353
2009	3,251	5,423	4,234	352	196	13,456
2010	2,475	6,702	6,455	398	320	16,350
2011	5,296	7,093	4,478	575	312	17,754
2012	5,246	7,656	5,069	870	823	19,664
Average	4,810.5	8,928.2	5,640.3	603.3	381.9	20,364.2
Total	48,105	89,282	56,403	6,033	3,819	**203,642**
Aver. Burnt Area (ha)						
2003	13.6	7.8	2.3	3.7	3.7	6.22
2004	9.4	9.4	1.8	5.1	4.8	6.1
2005	6.01	6.8	5.3	3.1	3.6	4.96
2006	7.8	9.3	3.4	2.5	2.4	5.08
2007	21.4	12.4	5.5	7.1	6.8	10.64
2008	10.3	12.2	3.8	3.1	3.6	6.6
2009	13.5	5.4	3.3	5.2	5.2	6.52
2010	9.52	10.8	2.3	1.8	1.6	5.2
2011	8.8	7.4	3.2	2.8	2.55	4.95
2012	15.8	13.2	6.8	6.1	67	9.72
Average	11.6	9.5	3.8	4.05	4.09	6.61
Environmental Impacts (ha)						
2003	91392	78,959.4	18,632.3	2,049.8	1,184	192,217.5
2004	45,336.2	78,978.8	11,982.6	1,810.5	1,392	139,500.1
2005	20,566.22	53,495.6	27,613	1,742.2	831.6	104,248.62
2006	33,056.4	81,123.9	10,920	1,137.5	432	126,669.8
2007	179,417.6	188,083.2	16,483.5	9,549.5	4,923.2	398,457
2008	43,775	147,900.6	37,962	1,757.7	1,522.8	232,918.1
2009	43,888.5	29,284.2	13,972.2	1,830.4	1,019.2	89,994.5
2010	23,562	72,381.6	14,846.5	7,16.4	512	112,018.5
2011	46,604.8	52,488.2	14,329.6	1,610	795.6	115,828.2
2012	82,886.8	101,059.2	34,469.2	5,307	5,514.1	229,236.3
Average	61,058.55	88,374.97	20,121.09	2,751.1	1,812.65	174,118.36
Total	610,485.52	883,749.7	201,210.9	27,511	18,126.5	**1,741,083.62**

Source: authors' elaborations on EFFIS Database

European Union Action to Fight Environmental Crime

The Costs of Illegal Wildlife Trade: Elephant and Rhino

WP3 Quantitative Analysis

Deliverable No. 3.2c

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LIST OF ABBREVIATIONS

AfESG	IUCN/SSC African Elephant Specialist Group
AfRSG	IUCN/SCC African Rhino Specialist Group
CAR	Central African Republic
CITES	Convention on International Trade in Endangered Species
DRC	Democratic Republic of Congo
ETIS	Elephant Trade Information System
GPTF	Game Products Trust Fund
IRDNC	Integrated Rural Development and Nature Conservation
IUCN	International Union for the Conservation of Nature
LRA	Lord's Resistance Army
MIKE	Monitoring the Illegal Killing of Elephants
PIKE	Proportion of Illegally Killed Elephants
SSC	Species Survival Commission
UNEP	United Nations Environmental Program
UNWTO	United Nations World Tourism Organisation
WTTC	World Travel and Tourism Council

1 Background: Summary of Available Information

1.1 Introduction: Illegal trade of Elephant and Rhino

African elephants and rhino are facing an uncertain future, placed at risk in the short term by increasing demands for ivory and rhino horn and in the long term by habitat loss, degradation and fragmentation from expanding human settlements. It is the short term threat of poaching, however, that puts elephant and rhino at immediate risk of extinction in the wild. Since 2007, illegal poaching has risen precipitously year after year to meet the insatiable market demand of mainly Asian consumers. The high value of ivory, and particularly rhino horn, have established these items as lucrative black market commodities, which has led to the trade becoming highly organised and professionalised.

Historically, both elephants and rhino became nearly extinct because of unsustainable hunting. The mass slaughter and near elimination of both species during the twentieth century led to concerted rehabilitation missions.[1] In the 1960's an international coordinated effort called "operation rhino" involved the repopulation of white rhino to southern Africa using just a few individuals. For elephants, unregulated poaching and hunting led to the introduction of important protection measures in the late 1980s with a 1989 Convention on International Trade in Endangered Species (CITES) ban on commercial trade of ivory and the inclusion of elephants on the International Union for the Conservation of Nature (IUCN) Red List as "vulnerable."[2] Elephant poaching levels are also the highest in over 25 years.[3] Both elephants and rhino are currently listed in Appendix I of CITES meaning that commercial trade in wild-caught specimens is illegal.[4]

As a result of these efforts, elephant and rhino populations experienced a decade of low levels of poaching (from 1995 to 2007) that saw their populations begin to rehabilitate. This brief period, however, came to an end in 2007 when poaching levels escalated dramatically and continued to increase on an annual basis.[5] For rhino in particular, the poaching rate over time exhibits the magnitude of growing demand with an average of only fourteen poached rhino individuals per year between 1990 and 2007 increasing to over a thousand in 2014.[6]

Poaching for ivory and particularly rhino horn is driven by the high value these products have on the black market. According to the wildlife trade monitoring network TRAFFIC, the street price of Rhino horn is $100,000/kg compared to the price in 1990 which at the time was estimated at $250-500/kg, with a

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[1] Tom Milliken and Jo Shaw, *The South Africa-Vietnam Rhino Horn Trade Nexus: A Deadly Combination of Institutional Lapses, Corrupt Wildlife Industry Professionals and Asian Crime Syndicates*, A TRAFFIC Report, (2012), http://www.traffic.org/species-reports/traffic_species_mammals67.pdf.

[2] IUCN Red List, "Loxodonta Africana," 2015, http://www.iucnredlist.org/details/12392/0.

[3] UNEP, *Elephants in the Dust: The African Elephant Crisis*, A Rapid Response Assessment (United Nations Environmental Programme, 2013), http://www.unep.org/pdf/RRAivory_draft7.pdf.

[4] Hunting only takes place with a limited number of permits/licenses dictated by CITES to private owners.

[5] Richard Emslie, Tom Milliken, and Bibhab Talukdar, *African and Asian Rhinoceroses - Status, Conservation and Trade.*, A report from the IUCN Species Survival Commission (IUCN/SCC) African and Asian Rhino Specialist Groups and TRAFFIC to the CITES Secretariat pursuant to Resolution Conf. 9. 14 (Rev. CoP16) (Bangkok, 2013), http://www.rhinos.org/professional-resources/iucn-african-rhino-specialist-group.

[6] Milliken and Shaw, *The South Africa-Vietnam Rhino Horn Trade Nexus: A Deadly Combination of Institutional Lapses, Corrupt Wildlife Industry Professionals and Asian Crime Syndicates*.

single horn weighing between 1-3kg, depending on the age and species.[7] Thus, the poached value of a rhino individual ranges between $100,000 - $300,000. The price of ivory has tripled in the last three years in China.[8] Uncarved ivory is worth $2,100 per kilo and an elephant on average has 10 kilos per tusk, thus the black market revenue of one poached elephant is approximately $21,000.[9] Demand from consumers is not abating and parallels the purchasing power of Asia's rising middle class which finds rhino and ivory to be symbolic of prestige and wealth.[10] However, what is fundamentally new is the surge in demand of rhino horn from Vietnam stemming from a rumour around 2008 when a Vietnamese politician claimed to be treated for cancer with rhino horn.[11] Contrary to popular belief, rhino horn is not a common ingredient of traditional Chinese medicine and its use now is distinctly a new trend tied to increased wealth and its perceived medicinal qualities.[12] The number of multimillionaires in Vietnam has grown 150% in the last five years. [13] At the same time, cancer rates in Vietnam are increasing 20-30 % annually with an estimated 150,000 new cases each year making for a long waiting list for radiation therapy and lack of capacity to deal with cancer in conventional facilities.[14] Scientifically rhino horn is composed of carotene and is the same chemical composition as a human finger nail, thus making the trade not only unsustainable but scientifically misguided. On the other hand, ivory has been traded throughout history, but demand from Asian countries particularly Vietnam (for rhino horn) over the last decade has led to a resurgence in poaching pushing many African elephant populations towards extinction.[15]

Range or source countries for ivory are widespread across Africa and include but are not limited to Sudan, Central African Republic, Democratic Republic of Congo, Chad, Kenya, Tanzania, Zimbabwe, Zambia, Malawi, Mozambique, Nigeria, Cameroon and Mali.[16] The main source countries for rhino horn are South Africa, Namibia, Zimbabwe and Kenya where 98% of all black and white rhinos live.[17]

Range and source countries have varying levels of experience, capacity and political will to stop the poaching and illegal trade. A 2014 study by Chatham House conducted an extensive literature review to

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[7] Gwynn Guilford, "China's Obsession with Rhino Horns Is Sending South African Rhino Deaths through the Roof," *Quartz*, December 19, 2013, http://qz.com/159902/chinas-obsession-with-rhino-horns-is-sending-south-african-rhino-deaths-through-the-roof/.

[8] "Price of Ivory in China Triples," *The Guardian*, July 3, 2014, http://www.theguardian.com/environment/2014/jul/03/price-ivory-china-triples-elephant.

[9] Rob Brandford, *Dead or Alive? Valuing an Elephant.* (United Kingdom: The David Sheldrick Wildlife Trust, 2014), http://iworry.org/wp-content/uploads/2013/09/Dead-or-Alive-Final-LR.pdf.

[10] UNEP, CITES, IUCN, TRAFFIC (2013). Elephants in the Dust – The African Elephant Crisis. A Rapid Response Assessment. United Nations Environment Programme, GRID-Arendal.

[11] Milliken and Shaw, *The South Africa-Vietnam Rhino Horn Trade Nexus: A Deadly Combination of Institutional Lapses, Corrupt Wildlife Industry Professionals and Asian Crime Syndicates*.

[12] Ibid.

[13] Gwynn Guilford, "Why Does a Rhino Horn Cost $300,000? Because Vietnam Thinks It Cures Cancer and Hangovers," *The Atlantic*, May 15, 2013, http://www.theatlantic.com/business/archive/2013/05/why-does-a-rhino-horn-cost-300-000-because-vietnam-thinks-it-cures-cancer-and-hangovers/275881/.

[14] "Rhinos Face Extinction by 2020, Wildlife Experts Warn | Al Jazeera America," accessed March 12, 2015, http://america.aljazeera.com/articles/2014/4/14/rhinos-face-extinctionby2020.html.

[15] Environmental Investigation Agency, *Made in China: How China's Illegal Ivory Trade Is Causing a 21st Century African Elephant Disaster* (Washington D.C., 2007), http://eia-global.org/images/uploads/Made_in_China_Report.pdf.

[16] George Wittemyer et al., "Illegal Killing for Ivory Drives Global Decline in African Elephants" 111, no. 36 (2014), http://www.pnas.org/content/111/36/13117.full.pdf+html.

[17] Emslie, Milliken, and Talukdar, *African and Asian Rhinoceroses - Status, Conservation and Trade.*

illustrate the consensus among authors that poaching and illicit trade flows thrive in regions with weak governance and conflict and is perpetuated by such conditions.[18] However, even countries such as South Africa that have a long standing history and experience in conservation as well as governmental and financial capacity, are also unable to adequately end illegal poaching and trade, which has become highly professionalized. For example, in South Africa, the number of prosecutions against poachers has risen, but poaching continues to accelerate annually.[19]

At the crux of the crisis is the fact that poaching levels are increasing annually and in some regions actually outpacing the natural reproductive capacities of the species. Possible extinction in some ranges for both elephants and rhino is a reality, in Central Africa the regional elephant population has been reduced by 64% in the last decade, decimated by large-scale poaching incidents killing as many as 300 individuals at once.[20] Extinction is also a real threat for some rhino populations, with the western black rhino declared extinct in 2013.[21]

As the largest land mammals on earth, elephant and rhino are particularly vulnerable because of their time-intensive reproductive habits that make their conservation status in the wild especially sensitive to consecutive years of accelerated poaching. Despite numerous reports citing known poaching statistics of elephant and rhino, there are few studies that attempt to gauge the rate of extinction from over harvesting, thereby coming to a predictive estimate of when a species will begin a steady population decline. Developing tentative extinction rates can inform effective conservation management and action.

1.2 Summary of Available Information

Both the African elephant and African black and white rhino have been included in Appendix I of the CITES Convention for over a decade and they or their parts are not allowed to be traded for commercial purposes. The most comprehensive source of population data for elephants is the African Elephant Database, which is maintained by the IUCN/ Species Survival Commission (SSC). The African Elephant Specialist Group (AfESG) and has produced five reports to date (i.e. 1995, 1998, 2002, 2007 and provisionally in 2015). The AfESG cooperates with the two CITES-mandated elephant monitoring systems (the programme for Monitoring the Illegal Killing of Elephants - MIKE - and the Elephant Trade Information System - ETIS) in an effort to integrate all available information on populations, poaching and illegal ivory trade. For rhino, the IUCN/SSC African Rhino Specialist Group (AfRSG) is the authority on population and poaching. The relatively low number of rhino and their earlier near extinction has led to meticulous monitoring and statistics.

The CITES trade database (Table 1) includes data on the trade of CITES-listed species using self-reported data by member states on import, export and re-exports. The database allows users to identify where trade of a particular endangered species is occurring at the national level, it allows searching for species, animal parts, and finished products (e.g. chess sets made from ivory) and also includes information on its intended purpose (circus, educational, hunting). The CITES trade database can give an indication of overall trade volumes of *legal* wildlife commodities or derivatives thereof. However, since elephant and

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[18] Katherine Lawson and Alex Vines, *Global Impacts of the Illegal Wildlife Trade: The Costs of Crime, Insecurity and Institutional Erosion.* (London: Chatham House, 2014), http://www.chathamhouse.org/sites/files/chathamhouse/public/Research/Africa/0214Wildlife.pdf.

[19] Milliken and Shaw, *The South Africa-Vietnam Rhino Horn Trade Nexus: A Deadly Combination of Institutional Lapses, Corrupt Wildlife Industry Professionals and Asian Crime Syndicates*.

[20] Wittemyer et al., “Illegal Killing for Ivory Drives Global Decline in African Elephants.”

[21] Platt, John R. “How the Western Black Rhino Went Extinct | Extinction Countdown, Scientific American Blog Network,” accessed March 4, 2015, http://blogs.scientificamerican.com/extinction-countdown/2013/11/13/western-black-rhino-extinct/.

rhino are forbidden to be commercially traded, the CITES database is not useful in gauging the level of *illegal* trade.

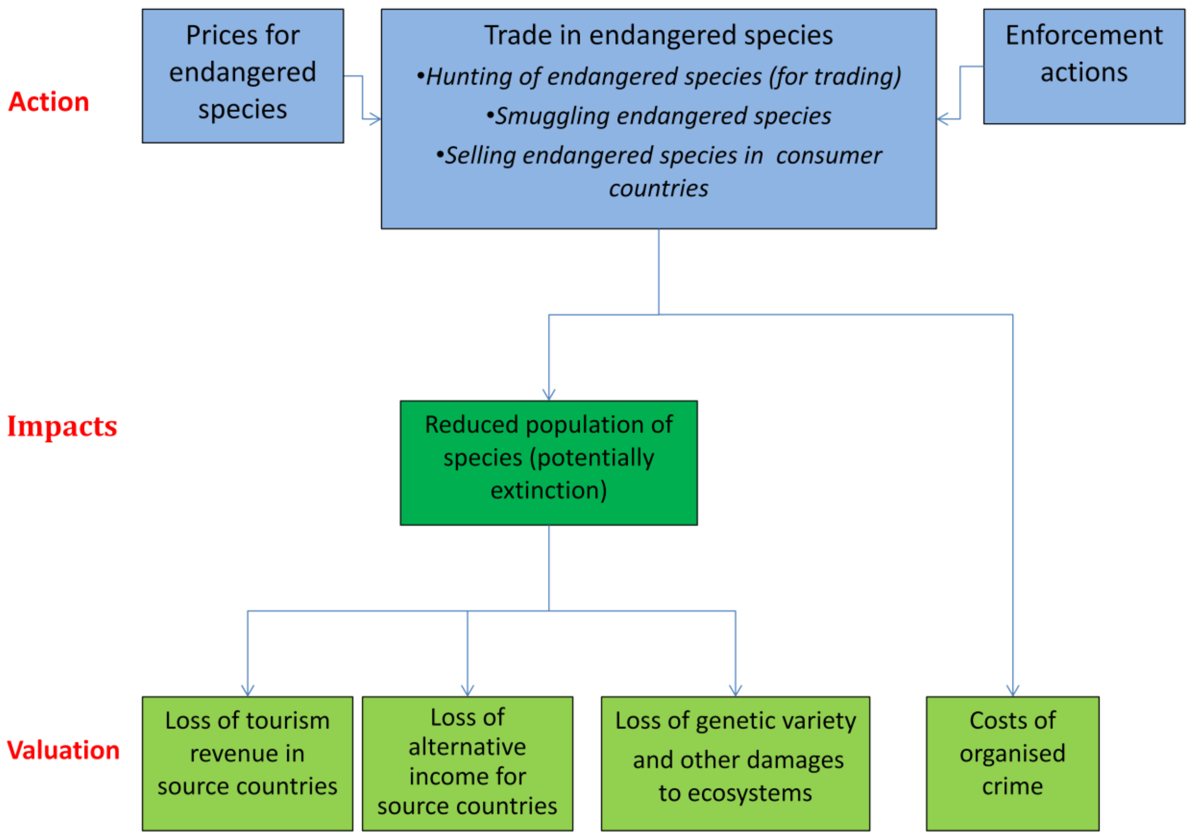
Information on the financial value of horn and ivory is constantly changing and for this reason online sources were used that collected information from newspapers and journalists to gauge the street value in real-time. Several internet search tools were also used that aim to tap into and evaluate the black market directly, collecting open-source information on the black market.[22] These websites collect information from involved individuals, consumers and whistleblowers. Black market prices of illegal goods were also compared with recent reports produced by TRAFFIC.

[22] Two websites that collect information black market activity are: https://wildleaks.org/ http://www.havocscope.com/

2 Methodology: what methods were used for quantitative analysis and for the monetary analysis

The following methodology sets out to explain how the impacts of illegal trade in certain CITES protected species could be assessed and which data sources and methods will be deployed to achieve thorough estimates for those impacts. The methodology is thereby based on the impact chain illustrated in Figure 1, which was developed in task one of WP3.

Figure 1: Impacts of trade in endangered species



The methodology has two distinct steps:

1. In the first step the impact of illegal trade on population numbers of the species protected by CITES is assessed based on existing statistics on poaching and population.
2. In the second step the impact of such population losses will be gauged taking into consideration environmental, social, political and economic impacts. For some of the monetary impact valuations will be undertaken.

2.1.1 Estimating the impact of illegal trade on population numbers

The most immediate impact of illegal trade in endangered species is the effect it has on populations, global biodiversity and ecosystem services. For many different species, data can be obtained on the conservation status of certain species' populations. Some databases[23] also keep track of the number of seizures of a certain species or parts thereof, however seizure data only gives an indication of illegal activity that is discovered by the authorities, and information on the "real" number of illegally traded species is difficult to discern, and thus in some cases the understanding of the direct impact of the trade on population numbers is limited.

For our analysis we therefore have chosen two species, where it is possible to divulge particularly accurate information on the real amount illegal activity.

- Elephants: A long history of conservation efforts combined with an upsurge in illegal poaching in recent years has led to improved monitoring efforts and data collection. The Elephant Database tracks the conservation status of elephants, with the last continental population census undertaken in 2012. The large geographic range of elephant populations which span some 37 countries, mean that available figures are estimates.[24] CITES and ETIS contain the most extensive database and record of ivory seizures, while MIKE contains "kill data", in other words, records resulting from identified poached animal carcasses in elephant range states.
- Rhinos: The IUCN Red List database tracks the conservation status of Rhino (Black and White). The near extinction of African Rhino in the 1980s led to close monitoring and conservation efforts resulting from an upsurge of poaching in 2007. The conservation history and low numbers of individuals in the wild has resulted in comprehensive and extensive surveillance and population data. Therefore, data exists on populations, the number of individuals poached by year and on seizures.

For each of the two species the available information on population and poaching is assessed to identify the causality between estimated illegal trade numbers and population figures. It is important to emphasize the fact that different datasets will have certain biases and are not complete. Therefore, we do not suggest to undertake a sophisticated statistical analysis, as the basic biases. The statistics can be more easily and more transparently identified using the following steps:

- **Trend analysis:** For elephant and rhino, the first phase of data collection and analysis will compare population data with the available data on kills for trade (bodies found without tusks or horns). The analysis should provide a first intuition of whether the data correlates to population declines, and if so in which magnitude and with which time delay.
- **Causal relationships:** These first intuitions are then tested by reviewing literature on the reproductive behaviour (average life span) of the species and other potential influences on population. It is tested whether the first intuitive trends of the impact are consistent with the best knowledge on the drivers of population figures.

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[23] There are also several databases that record data on seizures related to specific species such as elephants (ETIS) and tigers (Tiger Tracker Wildlife Trade Tracker). These databases give information on the number of seizures, the geographical location and volume or quantity. They are useful for monitoring the economic value and the environmental impact of trade in a specific species. EU TWIX is a database containing centralised data on seizures and offences reported by all 28 EU Member States, it is only accessible by national law enforcement officers and CITES authorities.

[24] Elephant Database, "2012 Continental Totals," 2013, http://www.elephantdatabase.org/preview_report/2013_africa/Loxodonta_africana/2012/Africa.

It is obvious that the impact of specific individuals (rhino and elephant) on a population differs according to the age of the poached animal, the average age of the population as a whole and the local distribution of the animals. Thus, aggregated values for the impact are always imprecise. But the main focus of this analysis is to produce estimates for the total economic and other loss caused by the crime and for that average figures of population loss will be calculated.

This will result in estimates on where poaching at current levels is reducing the population and where it is "only" reducing the increase of the population.

2.1.2 Estimating and valuing impacts of decreasing population numbers

Decreasing species numbers can have important impacts on host countries. Firstly, there is the danger that an existing ecosystem is destabilized and changed by the decreasing or even disappearing species. So the first step of the analysis needs to assess the role of the species in the ecosystem.

The second step of the analysis will undertake to value the loss or damage to the ecosystem and host countries. These are based on two different sources of income provided by the ecosystem with elephants and rhinos:

- If the poaching does not lead to reduced numbers of the species, the societal loss is valued by estimating the alternative legal income that the host society could reap from the animals, if they would not be poached. The assumption is here that without poaching, the host countries could use rising animal population numbers for to support the wildlife industry as it relates to tourism, trading of hunting rights, and game ranching. Earned income could then be reinvested in nature conservation or other purposes.

- If the poaching reaches a level that leads to a reduction of the population, the loss is valued as a loss of natural capital. The wildlife is the wealth of the source countries on which basis they can attract wildlife tourism and the associated annual income from it. It is assumed that a reduction in the numbers of rhinos and elephants leads to a reduction in wildlife tourism. Objectively a poaching rate of 1% above replacement level would probably lead to lower reduction in population numbers in the first years while if sustained the losses will be bigger in the later years, when populations become too small and scattered for reproductive purposes. But for lack of better data, the assumption is that this relationship is linear.

Other impacts mentioned above like the impacts of organised crime will not be quantified.

2.1.3 Limitations of the Above Methods of Estimation

The estimates provided have several limitations which need to be taken into account using the figures:

- For lack of better data we estimated that an extinction of rhinos and elephants would each diminish the total wildlife watching trade by 20%. As both rhinos and elephants have a very high status for wildlife tourists this may be an underestimate but no data was found to prove or disprove this assumption.
- As population figures are not always precise and up to date we had to calculate the population loss sometimes on the basis of the estimates for poaching. We estimated that 1% poaching above the natural growth of 5% would lead to a 1% population loss. This is a simplification as it is more likely that the impact will change in the course of a process leading to extinction. While poaching more than the natural growth will probably first have a lower impact (less than 1%) on population numbers as the average age of the population decreases, but if sustained, the impact will then become even higher than 1% as sex ratios become skewed and behaviour dynamics from the trauma of poaching affect animal groups.

3 Impact of Illegal Poaching on Population Numbers

The main environmental impact and threat posed by illegal harvest of horn and ivory is the extinction of African elephant and rhino in the wild. Taking current estimates of total populations and measuring the average population growth rate compared to the poaching rate can help predict the time period in which it would take for the species to become extinct.

3.1 Elephants

3.1.1 Geographic Range and Population Estimates

The extensive geographic range of African elephants makes it difficult to get exact figures on population and poaching and the quality and availability of specific population data varies greatly across the continent. Regionally, Southern Africa is home to the largest portion of known elephants (almost 55%) and the population is currently expanding under strong protection measures.[25] Eastern Africa contains 28% of the total population, with most in Tanzania and Kenya. Central Africa has 16% and West Africa less than 2%.[26] The impact of poaching differs region to region, with Central African elephant populations being hit especially hard. A study by Milliken (2014) estimates that Central African forest elephant populations have undergone a 76% decline since 2002.[27] The 2014 study by Wittemyer confirms such estimates, concluding that 75% of elephant populations are declining in Central Africa.[28]

Elephants are slow to mature and reproduce and for this reason are particularly susceptible to exploitative harvesting. A single calf is typically born every 2.5-9 years, after a gestation period of 22 months, and lives up to 70 years. Females are typically fertile between the ages of 25 and 45, and males begin successfully competing for mating after the age of 20.[29] Mature individuals are also the most heavily targeted because their bigger sized tusks make them more valuable, thus poaching often targets those elephants that are most important to maintaining reproductive potential of elephant communities and range populations. Poaching also has the effect of destabilizing herd dynamics and skews populations often leaving strongly skewed sex ratios and social damage from destroyed families and orphans, which affects the recovery of populations and conservation efforts.

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[25] The greatest share in this region and on the continent are located in Botswana; Mozambique, Namibia, South Africa, Zambia and Zimbabwe also boast significant populations.

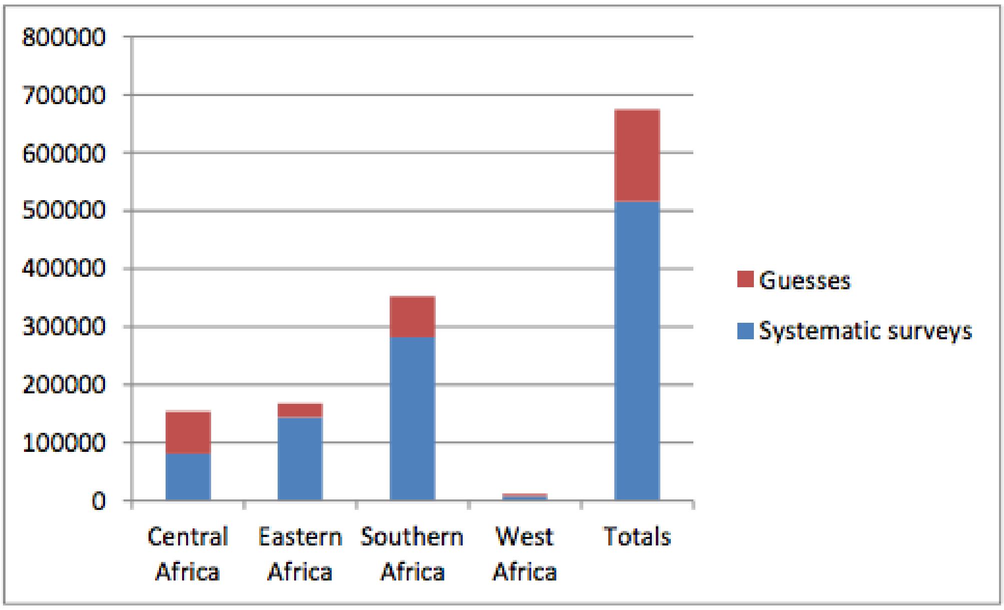
[26] Elephant Database, "2012 Continental Totals."

[27] Tom Milliken, *Illegal Trade in Ivory and Rhino Horn: An Assessment Report to Improve Law Enforcment under Wildlife TRAPS Project* (USAID and TRAFFIC, 2014).

[28] Wittemyer et al., "Illegal Killing for Ivory Drives Global Decline in African Elephants."

[29] WWF, "African Elephants," *African Elephants*, 2014, http://wwf.panda.org/what_we_do/endangered_species/elephants/african_elephants/.

Figure 2: Elephant Population and Range



Source: http://www.cites.org/sites/default/files/eng/com/sc/65/E-SC65-42-01

3.1.2 Impact of Poaching on Population Growth Rates

To monitor poaching, CITES created the MIKE database (Monitoring the Illegal Killing of Elephants) in 2002, a data collection network that collects information on the number of illegally killed elephants in African range states. MIKE aims to establish relative poaching levels by calculating the proportion of illegally killed elephants (PIKE) from the total number of identified carcasses.

Using PIKE data it is estimated that 20,000 to 25,000 elephants are illegally killed each year out of total population estimated to be between 420,000 and 650,000.[30] Poaching rates reached an all time high during the period from 2010 to 2012 where it is estimated that over 100,000 elephants were killed.[31] Illegal poaching became unsustainable in 2010 when illegal killing rates were ~6.8% between 2010 and 2012 meaning an average of ~33,630 elephants were killed per year based on current estimates of the entire elephant population. Between 2010 and 2013, approximately 7% of the entire elephant population was lost each year.[32]

In 2011, 40,000 elephants were illegally killed marking a kill rate of ~8% which correlates to a species reduction of ~3% that year. In 2012, the killing rate was 7.4% compared to an average annual population growth for elephants of 5% (in the absence of illegal killing), which means that more animals are being killed than are being born.[33] Thus, the poaching rate continues to outpace the natural reproduction rate thereby already indicating consecutive years of a downward impact on overall populations.

[30] IUCN African Elephant Specialist Group, *2013 Provisional African elephant status report.*

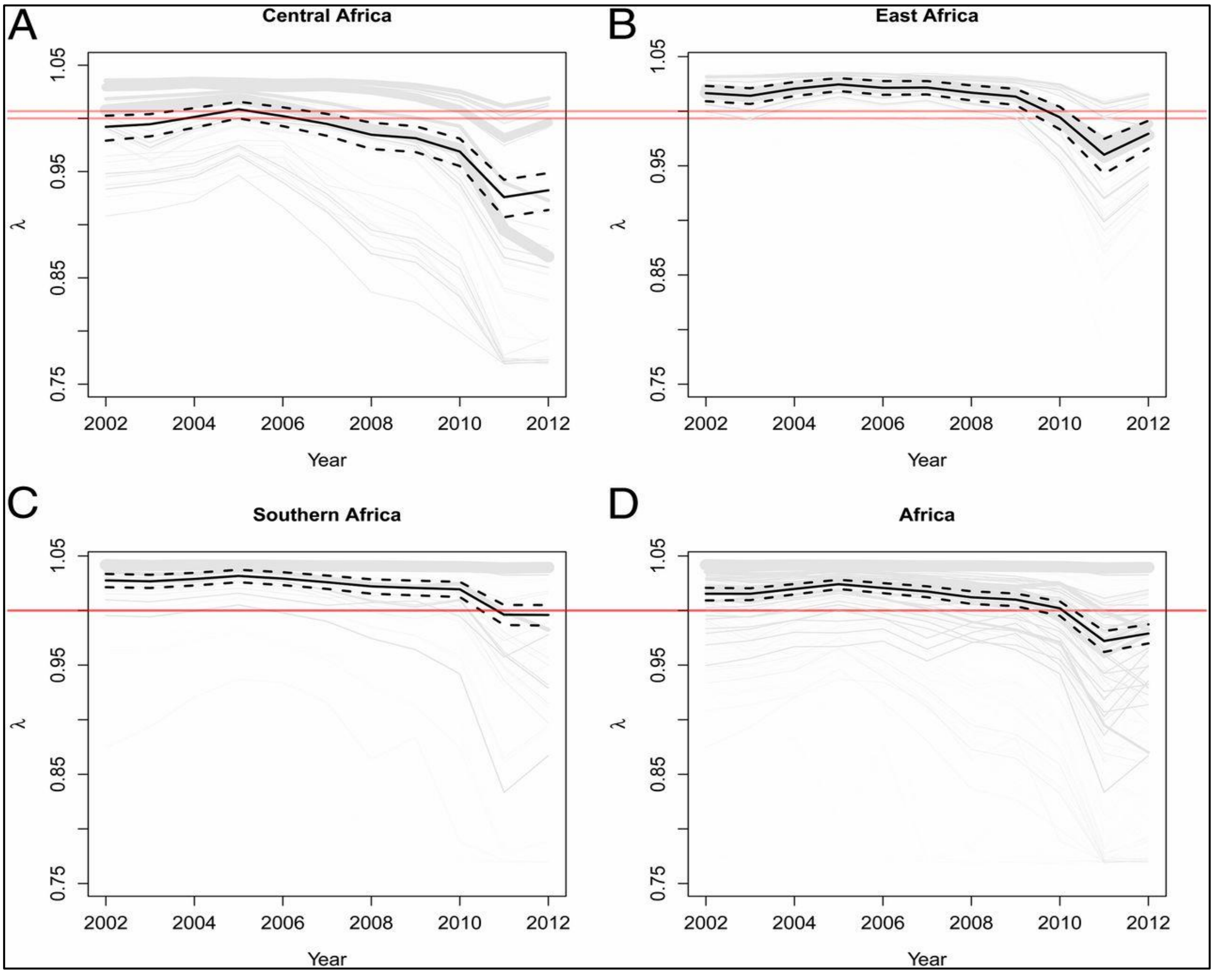
[31] "Organized Crime Threat to Wild Species on the Increase, Says UN on Wildlife Day - UNEP," accessed March 4, 2015, http://www.unep.org/newscentre/Default.aspx?DocumentID=26788&ArticleID=34775&l=en.

[32] Wittemyer et al., "Illegal Killing for Ivory Drives Global Decline in African Elephants."

[33] Ibid.

Figure 3: Modelled trends in annual population changes between 2002 and 2012 for 306 elephant populations across Africa

Modelled trends in annual population changes between 2002 and 2012 for 306 elephant populations across Africa presented by region: (A) Central, (B) East, and (C) Southern Africa regions and (D) all combined. Gray lines represent the site-specific annual population changes, where the thickness represents relative population size. Black lines represent the aggregate trends. Dashed lines represent the 95% confidence interval of aggregate trends.)



Source: George Wittemyer et al., "Illegal Killing for Ivory Drives Global Decline in African Elephants" 111, no. 36 (2014), http://www.pnas.org/content/111/36/13117.full.pdf+html. P13119

3.1.3 Conclusions for the Economic Assessment

For the economic assessment of the impacts of poaching we aim to distinguish between poaching that leads to a slower growth rate of the population and poaching that actually reduces the number of animals leading to extinction. For that purpose we regard a poaching rate of 5% of the population or less as reducing only the growth of population while a poaching rate of more than that reduces the population towards extinction. Using the numbers indicated above the following conclusions can be made:

- Africa lost a total of 100,000 elephants to poaching, which could have provided legal income for African countries.[34]
- As without poaching the population would have increased by about 75,000 animals (5% per year) in the three years, this poaching led to an estimated population loss of 25,000 elephants or about 5% of the estimated population of 500,000.

As figure 2 shows the trends were different in different regions of Africa. While in South Africa nearly no population loss occurred, the population losses were substantial in Central Africa.[35]

3.2 Rhino

3.2.1 Geographic Range and Population Estimates

There are currently an estimated 5,055 black rhinoceros and 20,405 white rhinoceros in Africa.[36] Today, globally, there are 29,000 rhinos in the world (including Asian species), compared to 70,000 in 1970 and 500,000 in 1900.[37] There has been a decline in population of 97% in the last century and rhinos have disappeared entirely from some regions of their natural range.

African rhino populations exist in four key range states: Kenya, South Africa, Namibia and Zimbabwe. South Africa, however, is home to the largest populations rhino accounting for 85% of the world's white rhino population and 73% of the black rhino population.[38]

The rate of poaching is gaining on that of rhinoceros reproduction with several authors predicting an overall downward trend in population growth beginning between 2014 and 2016.[39] Poaching threats, therefore, are compounded by the slow reproductive rates of rhino. A single calf is typically born every 2.5-4 years, after a gestation period of 16 months. Calves stay with their mother for 2-4 years which significantly slows down their reproductive potential. Females typically become fertile between the ages of 4 and 7 years, and males begin successfully competing for mating between 7 and 10 years. They live between 40 and 50 years of age.[40]

Comparing the poaching rate to the reproductive rate of rhino, it is possible to gauge the possible rate of extinction. The total population of white and black rhino in Africa has increased by 17.5% between 2007 and 2012 marking an increase in total population from 21,705 in 2007 to 25,510 in 2012.[41] The average rate of population growth during this time period was 4.9% per year between 2007 and 2010. This growth

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[34] Wittemyer et al., "Illegal Killing for Ivory Drives Global Decline in African Elephants."

[35] Ibid.

[36] Save the Rhino, "Rhino Population Figures," *Save The Rhino*, 2012, http://www.savetherhino.org/rhino_info/rhino_population_figures.

[37] Ibid.

[38] Guilford, "China's Obsession with Rhino Horns Is Sending South African Rhino Deaths through the Roof."

[39] Emslie, Milliken, and Talukdar, *African and Asian Rhinoceroses - Status, Conservation and Trade.*

[40] WWF, "African Rhinos," 2014, http://wwf.panda.org/what_we_do/endangered_species/rhinoceros/african_rhinos/.

[41] Sarah Standley and Richard Emslie, *Population and Poaching of African Rhinos across African Range States* (Climate, Environment, Infrastructure and Livelihoods Professional Evidence and Applied Knowledge Services (CEIL PEAKS), 2013).

rate decreased from 2010 to 2012 to 0.9% per annum.[42] While still growing, this significant decrease in rate indicates the pressures of poaching.

3.2.2 Impact of Poaching on Rhino Populations

From the early 1990s to 2007, rhino poaching was at a relatively low rate, which allowed for the recovery and growth of rhino populations across the continent. From 2002 to 2005, an 56 rhinos were illegally killed on the African continent. And from 1990 to 2006 an average of 15 rhinos were killed annually.[43] In some countries with targeted repopulation programmes, maximum growth rates were achieved. After 2007, the annual number of rhinos illegally poached for their horn soared, in particular, rhinos in Zimbabwe were hit hard as the country entered a state of economic and political turmoil. Annual rhino poaching figures have continued to increase year after year jumping from 201 in 2009 to 745 in 2012 and 1090 in 2013.[44] In preparation for the CITES COP15, the IUCN's Species Survival Commission (SSC) African Specialist Group estimated that between 2010 and 2011 rhino poaching increased 43% representing a loss of 3% of the African rhino population.[45]

Table 1 Poaching Statistics from the IUCN African Rhino Specialist Group

Region	2006	2007	2008	2009	2010	2011	2012	2013	2014
African Continent	60	62	262	201	426	520	745	1090	
South Africa	36	13	83	122	133	448	668	1004	1215

While poaching is occurring throughout the continent, few countries other than South Africa report statistics on illegally killed rhinos. South Africa has sustained the largest number of poaching incidents and is home to the overwhelming majority (85%) rhino population.[46] In 2014, 1,215 rhinos were illegally poached representing an increase of 21% from the previous year where 1,004 rhinos were poached.[47]

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[42] Sarah Standley and Richard Emslie, *Population and Poaching of African Rhinos across African Range States* Ibid.

[43] Milliken, *Illegal Trade in Ivory and Rhino Horn: An Assessment Report to Improve Law Enforcment under Wildlife TRAPS Project*.

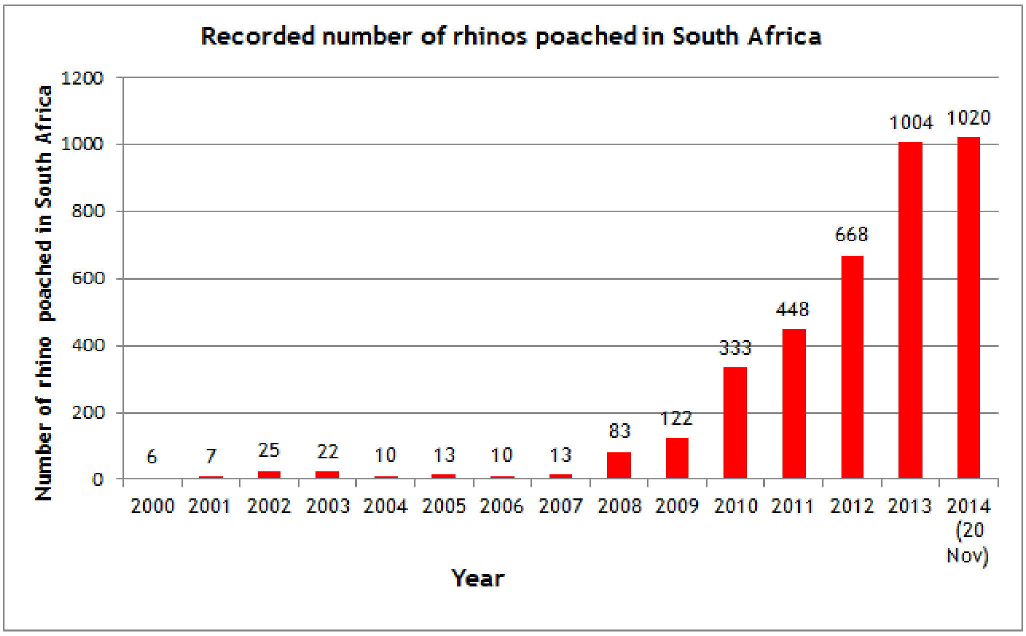
[44] Paul De Hert, Pieter Paepe, and Henk Griffioen, "Europees milieustrafrechtarrest minder voor nationale strafbevoegdheden," *144*, June 14, 2006.

[45] IUCN African Rhino Specialist Group, *CITES Rhino Report for Bangkok COP16* (COP16, 2013), http://www.rhinos.org/professional-resources/iucn-african-rhino-specialist-group.

[46] Save the Rhino, "Poaching Crisis in South Africa," 2015, http://www.savetherhino.org/rhino_info/thorny_issues/poaching_crisis_in_south_africa.

[47] Ibid.

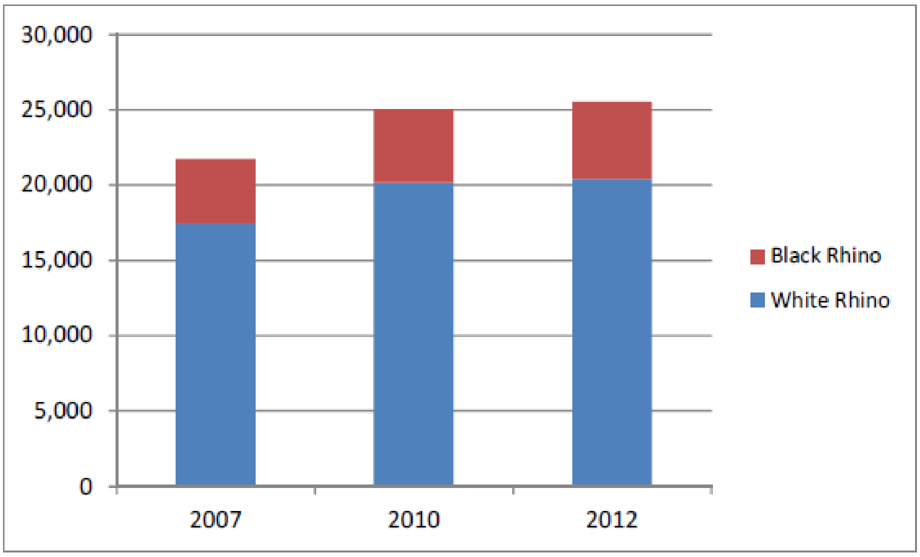
Figure 4 Recorded number of rhinos poached annually in South Africa



Graph 1, data published by South African Department of Environmental Affairs (2014)

Source: South Africa Department of Environmental Affairs. (2015) Available online: http://www.savetherhino.org/rhino_info/poaching_statistics

Figure 5 Rhino population increase for Africa between 2007 and 2012 (Data from IUCN SSC AfRSG)



Source: Sarah Standley and Richard Emslie (2013), *Population and Poaching of African Rhinos across African Range States.* Climate, Environment, Infrastructure and Livelihoods Professional Evidence and Applied Knowledge Services (CEIL PEAKS).

Reported poaching rates (% of poached rhinos per year compared to total population) of rhino between 2006-2010 are as follows: South Africa: 3.99%, Kenya 6.47%, in Namibia 0.18%, Zimbabwe 44.52%. During this period death rates exceeded birth rates in Kenya and, dramatically, in Zimbabwe. In 2010, in South Africa the poaching rate was 1.61%; in Kenya 2.29%, in Namibia 1.61 and in Zimbabwe 6.24%. Populations of rhino therefore seem to still be increasing in South Africa, Namibia, and Kenya despite 2010 poaching levels. However, current poaching rates in South Africa and Kenya are approaching levels that would lead to population losses. If poaching levels increase at the level they have in consecutive years of +34% to +46% a year, deaths will exceed births as early as 2015 and 2016.

Box 1: Zimbabwe: Case Example of Overharvesting and Population Collapse 2006-2009

Between 2006 and 2009, rhino poaching in Zimbabwe reached unsustainable levels fueled by political and economic instability in the country. As of 2007, the number of illegally poached rhino overtook the annual birth rate leading to a precipitous decline in the overall population of both black and white rhino. From 2006 to 2011, the reported number of poached rhino as a percentage of the population (2010) reached 44.5%. Concerted conservation efforts and the abatement of civil unrest within the country led to a decrease in poaching levels after 2010; however, rhino poaching as a percentage of the population remains very high at 6.46%.

Source: Sarah Standley and Richard Emslie (2013), *Population and Poaching of African Rhinos across African Range States* (Climate, Environment, Infrastructure and Livelihoods Professional Evidence and Applied Knowledge Services (CEIL PEAKS).

3.2.3 Conclusions for the Economic Assessment

Again we distinguish between poaching levels that reduce the population and poaching levels that only reduce the increase in population which could have been expected without poaching. We use the period 2006-2014, due to the availability of data.

- South Africa: For 2006-2014 3,827 Rhinos were poached which reduced the overall population growth, but did not lead to a reduced population. Only in 2013 and 2014 (more than a 1,000 animals per year) was the overall poaching close to the level where a population decrease could be expected.
- Namibia: Only five animals were poached from 2006-2012, which did not lead to any reduction in population.
- Kenya: 101 animals were poached between 2006-2012, which reduced the increase in population but did not lead to a population decrease.
- Zimbabwe: As mentioned above due to internal strife between 2006-2012, 378 animals were poached. The population decreased during that time by 67 animals or 8% of the population.

4 Qualitative and Quantitative Impacts of Illegal Rhino and Elephant Poaching

As outlined in chapter 2 the quantification of impacts is done on the basis of the economic value of the wildlife tourism and game preserve business for the African economies and on an estimate for potential economic gains of rhinos and elephants without poaching.

4.1 Quantifiable Social and Economic Impacts

We have calculated in chapter 3 the amount of poaching of rhinos and elephants that happened in the past. The following chapter undertakes to estimate an economic value for that amount of poaching based on methodologies:

1. Unsustainable poaching levels that lead to a decrease in population endanger the population as a whole and with that the whole income stream of wildlife tourism and game preserves. Maintaining that level of poaching in the long term will lead to a loss of that income stream. We estimate the loss of capital that is caused by this unsustainable poaching level.
2. But even levels of poaching that do not reduce population levels rob the African economy of a potential source of legal income. In a world without illegal poaching, African countries could use the population surplus to fund their spending on wildlife preservation, when the level of desired population is reached.

4.1.1 Wildlife Tourism and Game preserves – An Estimate of the Value of the Natural Wealth Loss

The most direct and straightforward loss incurred from illegal wildlife trade is that exploitation and elimination of countries' natural resource base on which many natural processes, human populations and industries depend.

Africa's unique ecology has made nature and its resources of particular value. Recognizing this fact, the United Nations World Tourism Organisation (UNWTO) conducted a study in 2014 to evaluate the economic importance of the wildlife watching sector of the tourism sector in African countries. The study used surveys and gathered information from 31 African governments and 148 tour operators from 31 different countries of which 49% were European and 51% were Africa based.[48] 93% of African governments confirm that poaching is a problem in the protected areas of their countries and 70% of tour operators stated that poaching is negatively affecting wildlife tourism. The survey study confirms that stakeholders are concerned about poaching and the impact it has on the future viability of the wildlife watching sector.

The wildlife watching sector can be understood as the tourist attraction of observing wildlife in its natural and non-captive habitat; the study did not include trophy hunting. Wildlife tourism is a motivation for foreigners to visit Africa, particularly to view the iconic "Big Five" (lion, leopard, water buffalo, rhino and elephant) in their natural habitat. The survey results from the 145 tour operators interviewed for the UNWTO study, indicate that approximately 80% of total annual sales trips to Africa are for wildlife watching and the average price per person per day of a standard wildlife watching tour is US $243 and for a luxury tour US $753.[49] Compared to culture, wellness, adventure, and a host of other entertainment sectors, wildlife is considered the most important tourism assets for incoming visitors (See Figure 6) and the important sight-seeing and ecological role of keystone species amplifies the negative impacts of overexploitation and extinction. Foremost, the potential extinction of one or two of the "Big Five" would affect the attractiveness of wildlife tourism. As poaching decreases wildlife populations, it also negatively affects the touristic experience as it changes animal behaviour (animals become shyer and more difficult to locate) and instils fear and safety concerns among visitors and gives countries a bad reputation. High levels of poaching are a deterrent for visitors particularly when park shootings take place, no-go areas are roped off and armed poachers are confronted which make tourists feel in actual danger.[50] Finally, a problem that

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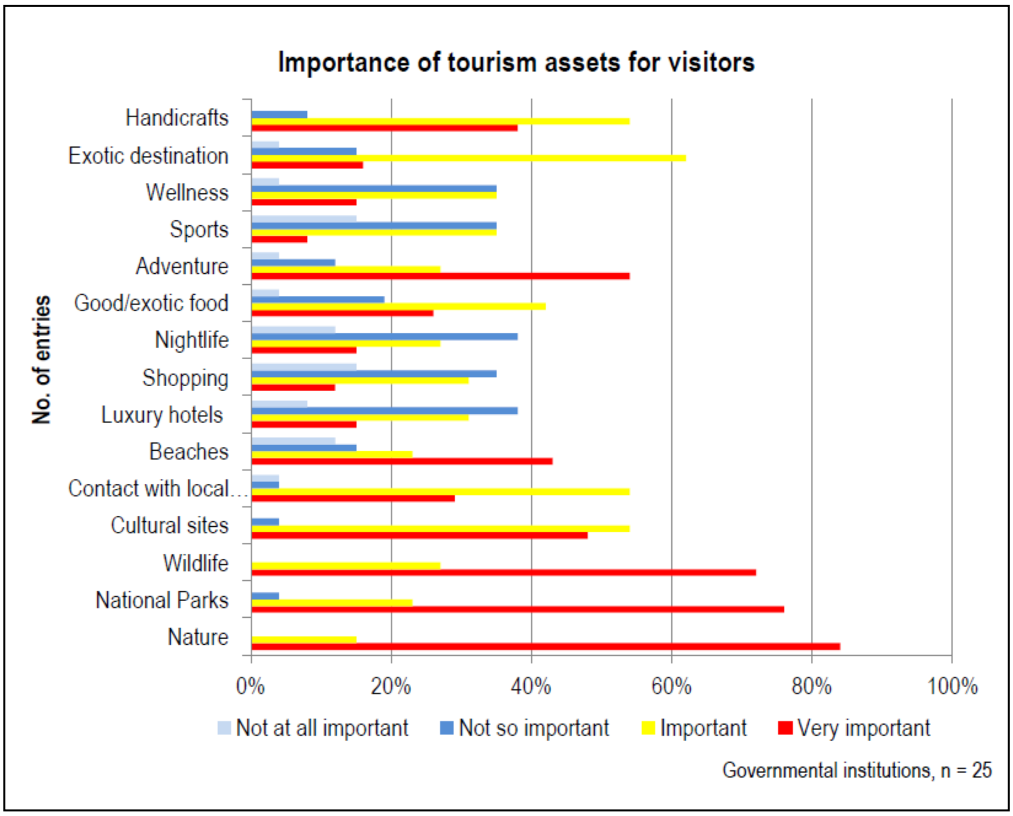
[48] UNWTO, *Towards Measuring the Economic Value of Wildlife Watching Tourism in Africa*, Briefing Paper (Madrid: World Tourism Organization (UNWTO) - A Specialized Agency of the United Nations, 2014), http://apta.biz/wp-content/uploads/2014/10/UNWTO-Wildlife-Study_Report.pdf.

[49] UNWTO, *Towards Measuring the Economic Value of Wildlife Watching Tourism in Africa*.

[50] Ibid.

this study has not valuated but that is extremely important is the fact that carrying out anti-poaching measures in protected areas is a huge financial burden. Efforts to thwart poaching rely on extensive monitoring by rangers and veterinarians, tagging, observation and security of animals, relocation and rehabilitation in cases of injury and death.

Figure 6 Importance of Wildlife and Wildlife Related Sectors for Visitors



Source: UNWTO, *Towards Measuring the Economic Value of Wildlife Watching Tourism in Africa*.

The economic impacts incurred from a destabilised wildlife tourism sector would also have substantial negative effects on national GDP and employment in source countries. The Word Travel and Tourism Council (WTTC) has provided data on the direct and indirect contribution of tourism to GDP and employment. Indirectly, wildlife tourism provides many employment opportunities, particularly for poor and rural populations who are the most vulnerable to marginalisation, food insecurity and extreme poverty. Employment from tourism spurs the development of restaurants, hotels/guesthouses, guides and also indirect financial benefits from the redistribution of protected area fees and community funds.

Figure 7 Direct and Indirect Contribution of Tourism on GDP and Employment in Source Countries

South Africa[51]

- GDP: Direct Contribution: 3.0% of GDP (ZAR103.2bn) in 2013 and is forecast to rise by 3.9% per annum from 2014-2023
- GDP: Indirect Contribution: 9.5% of GDP (ZAR323bn) in 2013, forecast to rise 9.8% of GDP

[51] World Travel and Tourism Council, *Travel and Tourism: Economic Impact 2014: South Africa* (WTTC, 2014), http://www.wttc.org/-/media/files/reports/economic%20impact%20research/country%20reports/south_africa2014.pdf.

- Employment (Direct Contribution):645,500 jobs (4.6 % of total employment)
- Employment (Indirect Contribution): 1, 404,000 jobs (10.1% of total employment)

Kenya[52]

- GDP: Direct Contribution: 4.8 of GDP (KES183.4bn) in 2013 and is forecast to rise by 5.2% per annum from 2014-2023
- GDP: Indirect Contribution: 12.1% of GDP (KES462.8bn) in 2013, forecast to rise 5.2% of GDP
- Employment (Direct Contribution): 226,500 (4.1 % of total employment)
- Employment (Indirect Contribution): 589,500 jobs (10.6% of total employment)

Zimbabwe[53]

- GDP: Direct Contribution: 5.6 of GDP (USD 420.1mn) in 2013 and is forecast to rise by 6.1% per annum from 2014-2023
- GDP: Indirect Contribution: 11.4% of GDP (USD857.9mn) in 2013, forecast to rise 6.0% of GDP
- Employment (Direct Contribution): 42,500 (3.7 % of total employment)
- Employment (Indirect Contribution): 98,500 jobs (8.2% of total employment)

Namibia[54]

- GDP: Direct Contribution: 3.0 of GDP (NAD3.126-5mn) in 2013 and is forecast to rise by 9.2% per annum from 2014-2023
- GDP: Indirect Contribution: 14.8% of GDP (NAD15,302.6mn) in 2013, forecast to rise 7.6% of GDP
- Employment (Direct Contribution): 24,000 (4.5 % of total employment)
- Employment (Total Contribution): 103,500 jobs (19.4% of total employment)

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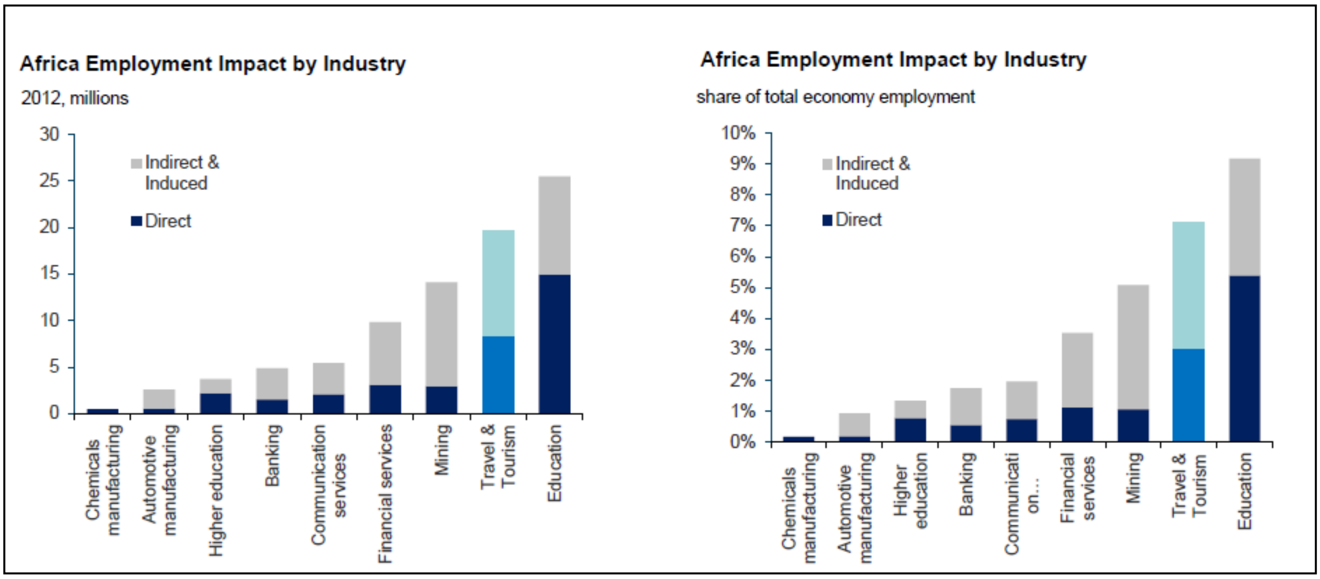
[52] World Travel and Tourism Council, *Travel and Tourism: Economic Impact 2014: Kenya* (WTTC, 2014), http://www.wttc.org/-/media/files/reports/economic%20impact%20research/country%20reports/kenya2014.pdf.

[53] World Travel and Tourism Council, *Travel and Tourism: Economic Impact 2014: Zimbabwe* (WTTC, 2014), http://www.wttc.org/-/media/files/reports/economic%20impact%20research/country%20reports/zimbabwe2014.pdf.

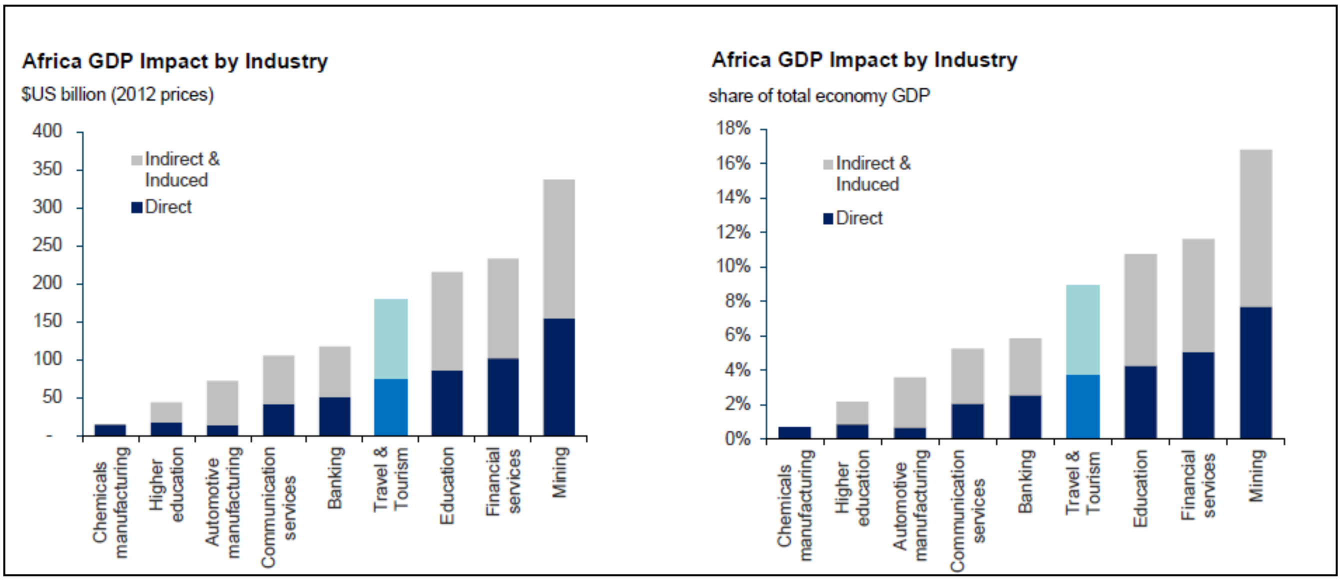
[54] World Travel and Tourism Council, *Travel and Tourism: Economic Impact 2014: Namibia* (WTTC, 2014), http://www.wttc.org/-/media/files/reports/economic%20impact%20research/country%20reports/namibia2014.pdf.

Figure 8 Value of Travel and Tourism to African Economies: Sector Comparison



Source: http://www.wttc.org/-/media/files/reports/benchmark%20reports/regional%20results/2013%20africa%20summary.pdf.

Figure 9 Value of Travel and Tourism to the Generation of Employment Compared to other Industries



Source: http://www.wttc.org/-/media/files/reports/economic%20impact%20research/regional%20reports/africa2014.pdf

The total value of the wildlife trade

The following table summarises the economic losses caused by poaching above the level of replacement. The following simplifying assumptions for the estimation of the total value are taken:

- Wildlife watching represents 80% of the total annual sales of trips to Africa and sales are increasing.[55] The number represents the tour operators participating in the UNWTO study and it probably produces an overestimate for countries like South Africa with other tourist attractions and an underestimate for other countries.
- The overall value of a capital asset like the wildlife of Africa is 20-30 times the annual income stream it provides. This estimate is in line with the usual assumptions on other long term capital assets.
- An extinction of one of the "big five" animal species would reduce the wildlife tourism by 20%.

Table 2 Valuation Estimates of Extinction of Rhinos in Four Countries

Rhino	South Africa	Namibia	Kenya	Zimbabwe
Annual (direct and indirect) tourism income	€ 24,7 billion	€ 1,2 billion	€4.8 billion	€1,4 billion
Annual wildlife tourism income[56]	€19.7 billion	€ 0,9 billion	€3,8 billion	€1.1 billion
Total capital value of wildlife tourism income (20-30 times annual income)	€395-592 billion	€18.7- 28 billion	€76.8-115.2 billion	€22.66-33.8 billion
Loss of extinction of all rhinos (20% of wildlife tourism income)	€79- 118 billion	€3.7-5.6 billion	€15.4-23 billion	€4.5-6.8 billion
Value of 1 % reduction in rhino population	€0.79-1.18 billion	€0,037-0,056 billion	€0.15-0.23 billion	€0.045-0.068 billion

*National currency figures converted into € using with exchange rates of the 2nd March 2015.

Table 3 Valuation Estimates of Extinction of Elephants

Elephants	All African range states
Annual (direct and indirect) tourism income	€74.1 billion
Annual wildlife tourism income[57]	€59.3 billion
Total capital value of wildlife tourism income (20-30 times annual income)	€1,185-1,778 billion
Loss of extinction of all elephants (20% of wildlife tourism income) annual	€237-356 billion
Value of 1% reduction in elephant population	€2.4 - 3.6 billion

The figure above is overall conservative as it estimates the total capital value for tourism of each elephant to €500,000 to €800,000. A recent study estimated that the value of a live elephant for tourism purposes is USD 1,607,624 throughout the course of its life.[58]

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[55] UNWTO, *Towards Measuring the Economic Value of Wildlife Watching Tourism in Africa.*

[56] Wildlife watching represents 80% of total annual sales of trips to Africa (gathered from 148 tour operators)

[57] Wildlife watching represents 80% of total annual sales of trips to Africa (gathered from 148 tour operators)

[58] Brandford, *Dead or Alive? Valuing an Elephant.*

4.1.2 Trophy Hunting/Ranching:

Another branch of wildlife tourism is the private ranching and trophy hunting business. Milliken and Shaw (2013) identify ranching as being a central component for the recovery and growth of rhino populations. They also explain that ranching provides an incentive for land owners and communities to keep and protect wildlife stating that what was once a mainly public industry has become a booming private one where wildlife is harvested, hunted and traded. The partial privatisation of the wildlife industry has also helped to professionalise the industry in aspects such as translocation, capture, immobilization and veterinary care which has improved standards for the entire industry (public and private).

Poaching is affecting this industry in two ways. Firstly, the cost of measures to keep animals protected and safe from poachers makes keeping rhino and elephant uneconomic. Secondly, ranchers that sell licenses to tourists to hunt big game lose the revenue they would make from the sale of the hunt of the animal. In 2014, the amount an individual was willing to pay to hunt a rhino was USD 350,000 (€ 312,640).[59]

Looking on websites offering trophy hunting services for elephant such as AfricaSkyHunting in South Africa, the potential revenue earned is between 25,000 and 35,000 USD (€ 22,330 to €31,264).[60] The poaching crisis is a threat to this industry as it is ever more expensive for public and private owners alike to protect their animals from poaching.[61]

4.1.3 Example for the value of a Rhino: Namibia and Community Led Conservation

In Namibia a community managed approach to wildlife conservation has led to great success in nationwide conservation of land and the rehabilitation of several once endangered species including the critically endangered black rhino. It started when an NGO called Integrated Rural Development and Nature Conservation (IRDNC) paid local communities to protect wildlife. The initial success of this approach led the newly independent government of Namibia in the 1990s to turn over ownership of wildlife to communal conservancies. The main idea was to create a situation that motivated villagers to manage and profit from the conservation of land and wildlife that existed already existed in the country.

Trophy hunting has become an important approach of communal conservancies because of its low level of initial investment especially compared to setting up wildlife lodges and touristic reserves.[62] Trophy hunting has been an easy and lucrative incentive for communities to maintain and benefit from wildlife. Each year the Namibian Ministry of Environment and Tourism sets limits to hunting endangered species and for example has an annual quota whereby five black rhinos can be sold for trophy hunting. The price for hunting a black rhino in 2014 was USD 350,000, of which all the money goes into a trust for rhino protection: paying for game rangers and anti-poaching protection measures such as tagging, implantation of chips, and capture/relocation.[63] In 1997 the Namibian government established the Game Products Trust

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59 Richard Conniff, "A Trophy Hunt That's Good for Rhinos," *The New York Times*, January 20, 2014, http://www.nytimes.com/2014/01/21/opinion/a-trophy-hunt-thats-good-for-rhinos.html.

60 "African Sky Hunting," Company Website, (2015), http://www.africanskyhunting.co.za/trophies/elephant-hunting.html.

61 Expenses include aspects like anti-poaching and monitoring patrols, ranger salaries, technology such as micro chips, drones and transport such as helicopters and vehicles.

62 N. Leader-Williams et al., "Trophy Hunting of Black Rhino Diceros Bicornis: Proposals to Ensure Its Future Sustainability," *Journal of International Wildlife Law & Policy* 8, no. 1 (January 1, 2005): 1–11, doi:10.1080/13880290590913705.

63 Conniff, "A Trophy Hunt That's Good for Rhinos."

Fund (GPTF) which transfers money from wildlife trophy hunting industry to conservation. Tourism and rhino hunting concessions provides 70-80% of the Fund's revenue.[64]

Currently some 20% of Namibia is community owned and managed and 44% of the country's land surface is under some kind of environmental protection.[65] While poaching of rhino and elephant have soared in neighbouring South Africa, Namibia has maintained very low levels of poaching. Only two black rhinos were poached in 2013 and the population has enjoyed stable growth from 1,435 individuals in 2007 to 1,750 in 2012. Other species once threatened have also made impressive come backs in Namibia using the hunting and community management approach. For example, the number of elephants has increased from 15,000 in 1995 to 20,000 in 2013. The number of zebra has increased from 1,000 in 1982 to 27,000 in 2013.[66]

Since community managed conservancies provide local individuals with jobs as rangers and guards as well as business opportunities for other tourist activities (e.g. handicrafts, indigenous artisanal, bed and breakfasts, restaurants, transportation services etc), there is a strong build up of social pressure and stigma against illegal poaching due to the very fact that the livelihoods and income of the communities are intrinsically tied to the protection and calculated harvest of wildlife for which the proceeds are shared collectively. The GPTF has also created other mechanisms that ensure that communities opt to protect wildlife rather than exploit or eliminate it. For example, for communities that experience economic losses to wildlife, for instance, the killing of cattle or infrastructure, the GPTF will provide remuneration. Community based resource management is an approach that puts wildlife conservation in the hands of local communities.

4.1.4 Qualitative Impacts of Poaching committed by Organised Criminal Groups

Rhino and elephant poaching is often associated with the immediate environmental impacts on biodiversity and conservation of an endangered species, but it also has important security and socio-economic impacts. The financial value of illicit trade has made wildlife trafficking the fifth most lucrative type of illegal trade in the world and is comparable to other types of illegal trade such as drug-smuggling, weapons, counterfeit goods and human trafficking.[67] In a literature review of wildlife crime and security implications, Chatham House cited numerous studies that detailed the involvement of organized criminal groups and armed non-state actors in wildlife trade of ivory and rhino as well as other goods such as tiger. In this overview of existing studies, wildlife trafficking was found to have strong stunting impacts on source countries by undermining institutions and the rule of law, creating political and economic instability and perpetuating conflict and violence (e.g. civil conflict, intrastate conflict and transnational organized crime syndicates).[68] Quantifying the cost (financially, socially, politically) of prolonged conflict in a country is difficult but undeniably relevant in many cases where wildlife trafficking has played a role,

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[64] "Can Rhinos Profit from Trophy Hunting?," *Project: African Rhino*, accessed March 9, 2015, http://africanrhino.org/2013/11/11/can-rhinos-profit-from-trophy-hunting/.

[65] WWF, *Communal Conservancies. Namibia's Gift to the Earth.*, 2011, http://www.namibiatourism.com.na/uploads/file_uploads/Namibian_Conservation_Fact_Sheet_copy.pdf.

[66] WWF, *Communal Conservancies. Namibia's Gift to the Earth.*

[67] Nigel South and Tanya Wyatt, *Comparing Illicit Trades in Wildlife and Drugs: An Exploratory Study*, Deviant Behavior (London, 2011), http://dx.doi.org/10.1080/01639625.2010.483162.

[68] Jeremy Haken, *Transnational Crime in the Developing World*, Global Financial Integrity, 2011, http://www.gfintegrity.org/storage/gfip/documents/reports/transcrime/gfi_transnational_crime_web.pdf.

such as the Democratic Republic of Congo (DRC),[69] the Central African Republic (CAR), Sudan and some other countries. A 2014 United Nations Environmental Program report cites that ivory has been an important source of income to militia groups in the DRC and CAR, and is one of the main sources of income to the Lord's Resistance Army (LRA) currently warring in the border region of South Sudan, CAR and DRC and decimating elephant populations in those countries.[70] Criminal networks, whether they are state militias, rebels, terrorist organizations or organized criminal syndicates, benefit from weak institutions and lawless environment and have an inherent interest to maintain such a state for their illegal operation. As Jeremy Haken points out, illegal wildlife trade does "immeasurable structural damage to developing economies by empowering forces which erode the capacity of the state."[71] Moreover, the long-term looting of resources and perpetuation of violence and crime have repercussions that extend from economic concerns to social ones that deeply effect the development trajectory of source countries.

Wildlife poaching and trafficking not only perpetuates state instability and institutional weakness in source countries, it also exports security threats globally. The associated costs for the international community are difficult to quantify in absolute terms but are clearly connected when looking at case by case examples. According to Interpol and the U.S. State Department, several Islamic extremist groups have used wildlife trafficking to fund their activities.[72]

5 Results and Conclusions

Overall the poaching of rhinos and elephants causes significant damage to African economies both by taking away current and legal income opportunities for African economies and also by reducing the natural capital on which all future income opportunities are based. Using the numbers developed in the paper, the overall impact on those economies can be estimated in the following ways:

Table 4 Economic Value Lost Due to Elephant Poaching

	Africa
Total population of Elephants in Africa 2010	500,000
Number of elephants poached 2010-2012	100,000
Lost potential legal income per Elephant	€22,331 - €31,264
Total loss of potential legal income 2010-2012	**€ 2.23 billion – € 3.12 billion**
Total loss of population 2010-2012	25.000 (5% of population)
Value of 1% population loss	€ 2,4 billion to € 3,6 billion

[69] Civil conflict in the DRC has also led to mass poaching by militia and rebel groups contributing to atrocious and unabated violence since 1996 and elephant population decline by 50% between 1995-2006[69]. Elephant poaching in the DRC is conducted by the Congolese army as well as armed non-state actor including Mai Mai rebel groups and the FDLR.

[70] Agger, Kasper and Jonathan Hutson, 'Kony's Ivory: How Elephant Poaching in Congo Helps Support the Lord's Resistance Army', Enough Project (June, 2013)

[71] Haken, *Transnational Crime in the Developing World*.

[72] Lawson and Vines, *Global Impacts of the Illegal Wildlife Trade: The Costs of Crime, Insecurity and Institutional Erosion.*

Total loss of natural capital 2010-2012	**€ 12 billion to € 18 billion**
Total economic loss per year	**€ 4 billion to € 6 billion**

The loss of economic value caused by illegal poaching is significant, as is made evident in the table above when analysing the data of elephant populations during the most extensive years of poaching for ivory 2010-2012. Following our valuation approach the biggest damage is caused by the loss of population which, has already incurred significant financial losses and if sustained, endangers an important natural capital asset and an important economic sector for African economies.

Table 5 Economic Value Lost Due to Rhino Poaching

	South Africa	Namibia	Kenya	Zimbabwe
Total population of rhinos 2012	20.954	2214 (2010)	914	792
Number of rhinos poached 2006-2014	3.827	5 (2006-2011)	101 (2006-2012)	378 (2006-2012)
Lost potential legal income per rhinos	€ 312.640	€ 312.640	€ 312.640	€ 312.640
Total loss of potential legal income per year	**€133 million**	**€0.26 million**	**€4.5 million**	**€16.9 million**
Total loss of population 2010-2012	0	0	0	67 (8%)
Value of 1% population loss	€790-1,180 million	€37- 56 million	€150 - 230 million	€45- 68 million
Total loss of natural capital 2006-2012	0	0	0	€360-544 million
Total loss of natural capital per year	**0**	**0**	**0**	**€51-76 million**
Total economic loss per year	**€133 million**	**€0.26 million**	**€4.5 million**	**€68 - 93 million**

The economic losses caused by rhino poaching are less than the losses caused by elephants mainly due to the much higher occurrence of elephant poaching due in part to the fact that there are more individuals . Except for Zimbabwe, rhino poaching does not yet exceed the natural growth of populations in the three other range states, meaning that only in Zimbabwe has a loss of natural capital occurred. However, current trends in South Africa and other countries are en route to having a situation where poaching outpaces birth rates. It is worth noting though that these estimates do only cover a small part of the overall societal costs of rhino and elephant poaching. The illegal trade does cause other costs which cannot be valued just yet. The significant annual expenditure for the safeguarding of the hunted animals and the impact of the trade on the governance of the source countries, increasing corruption and organised crime, are a serious impediment for economic development.

From a purely macro-economic viewpoint the direct and indirect contribution of wildlife tourism to economic growth and employment indicates the importance of this sector and also its potential to develop. One of the main objectives of this report, however, was to go beyond citing the importance of the wildlife watching sector and attempt to measure its economic value. So far there is very little available data for such an analysis and this report relied primarily on the initial findings from the 2014 UNWTO study. Economically elephants and rhino in a conserved and managed state are worth more than the immediate price of their tusks and horns. In other words, their value over time, whether it be for wildlife watching purposes or for carefully managed private ranching and hunting, has a more substantial value.

Moreover, the financial benefits tend to be more equitably shared in legal markets maintaining or contributing to alternative sources of income and employment opportunities for communities or as the case of Namibia illustrated, ranching and hunting when authorized through legal structures can be redistributed to maintain those resources and communities.

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Annex A

We also used the Environmental Valuation Reference Inventory (EVRI) a searchable database of empirical studies on the economic value of environmental benefits and human health effects to select several potentially relevant studies that are listed in Annex 1. From the EVRI database, one study supported the idea that the value of an elephant alive for nature watching purposes is significantly higher than its value accrued illegally through the sale of its ivory. The 2008 report by Scholes and Mennell found that estimates for the total value of African elephants per annum to South Africa ranged from US$4.1-77.2 million from the consumptive value of ivory to US$2.2 billion from non-use values. A 2005 study by Lindsey et al. on endangered wild dogs found that the revenue brought in through ecotourism by visitors specially interested in seeing this species far offset the costs of conservation plans to maintain them. Finally a study on the economic touristic value of gorilla for Uganda found that the economic value of gorilla tourism was estimated at $7.8-$34.3 million depending on the discount rate used. Our results are therefore broadly in line with similar endeavours.

Literature Review by EVRI

Title: "A Recreational Demand Model of Wildlife-Viewing Visits to the Game Reserves of the KwaZulu-Natal Province of South Africa" (2000)

Key: Value of wildlife viewing/reserves, South Africa

Estimated (Service Flow) Values:	The nested multinomial logit model provided a better fit to the data, so it is used for the welfare estimations. Table 1 reports the per trip welfare values for the loss of the game reserves of KwaZulu-Natal. The average household would suffer welfare losses of $49.71 per trip from the removal of Hluhluwe, while the loss from Mkuzi would be only about $18.67 (1995 U.S. Dollars). Table 1 also presents average per trip values for the loss of access to both Hluhluwe and Umfolozi. The value for the loss of both reserves simultaneously ($105.55) is considerably higher than the sum of the loss of both reserves individually ($80.18). This result illustrates the importance of substitution and the benefits of using a model that allows for such effects. If both are removed, the substitution possibilities are considerably diminished.

Title: "The Economic Returns to Wildlife Management in Southern Africa, in the Valuing the Environment in Developing Countries: Case Studies, edited by David Pearce, Corin Pearce and Charles Palmer" (2002)

Key: Botswana, Land and Animals

Estimated (Service Flow) Values:	Table 1 shows the financial and economic worth of eight typical wildlife/rangeland use enterprises in Botswana. Of the eight, crocodile farming generated the largest net present value (NPV) of 2.56 million pula (1 Botswana pula = 0.47 USD or 1.34 South African rand) per square km. of land (at 6% over ten years, 1991 prices) followed by ostrich farming and wildlife viewing. Table 2 shows the total economic value and contribution of each of the different wildlife use and livestock production to the national income per annum. The non-consumptive wildlife uses in particular Wildlife viewing generated a net value added of 92.52 million pula, about 78% of the total GVA from wildlife use and livestock production activities.

Title: "The Potential Contribution of Ecotourism to Africa Wild Dog Lycaon pictus Conservation in South Africa" (2005)

Key: Economic value of endangered species (African wild dog), South Africa

Estimated (Service Flow) Values:	Table 1 gives the estimated mean willingness to pay values of tourists for the opportunity to improve their chances of viewing the endangered African wild dog. Table 2 and 3 lists the net present values of conservation associated with the Kruger National Park, nature reserves and ranchland populations. The results for both nature reserves and ranchland are given under various prey compensation schemes where different costs arise when all prey is killed, half the prey is killed and no prey is killed. As well, costs for nature reserves were estimated based on whether adequate wild dog fencing was in place. All values are in 2002 US $ (ZAR in parentheses). 2002 US$/ZAR exchange rate is 1$=R10.99.

"The Existence Value of Biodiversity in South Africa: How Interest, Experience, Knowledge, Income and Perceived Level of Threat Influence Local Willingness to Pay" (2003)

Key: Willingness of communities to pay for conservation

Estimated (Service Flow) Values:	Overall, a mean willingness to pay of US $10.4 ($0.6 - $195.2) was found for nature conservation in South Africa. Extrapolation to 76% (% WTP contribution towards biodiversity conservation) of Western Cape households (1.06 million) provides an estimate of $ 8 378 240 per year for nature conservation. Based on data obtained through the dichotomous choice questions, overall estimated average willingness to pay for reduced climate change impacts is $36.7. The mean WTP based on the climate change open-ended question was $27. The extrapolated estimate 1.06 million Western Cape households are between $21 and $39 million. Table 1 gives the estimated Western Cape population WTP values for biodiversity in the 7 biomes.

Title: "Biodiversity and Nature-Based Tourism at Forest Reserves in Uganda" (2001)

Key: Value of biodiversity, forests, birds, Uganda

Estimated (Service Flow) Values:	Based on the calculated mean optimal entrance fee of $47 (standard deviation = 2.28, range = 37.5 - 62), the annual average revenue to the Mariba Forest Reserve was estimated to be $29,919 (standard deviation = 3,386, range = 12,198 - 25,196). The number of bird species was a strong determinant of revenue. For 20 species of birds seen, the revenue was $18,032 (standard deviation = 2,346, range = 12,198 - 25,196). For 80 species of birds seen the revenue was $40,423 (standard deviation = 4,287, range = 27,662 - 52,158). These values correspond to $0.60 to $1.35 per hectare. All values in 2001 $US.

Title: "Can local communities in Zimbabwe be trusted with wildlife management?: Evidence from contingent valuation of elephants" (2009)

Key: endangered species, elephants, Zimbabwe

Estimated (Service Flow) Values:	The median WTP for the preservation of 200 elephants is ZW$260 (US$4.73) for respondents who consider elephants a public good while the same figure is ZW$137 (US$2.49) for those who consider elephants a public bad and prefer translocation. The preservation of 200 elephants yields an annual net worth of ZW$10,828 (US$196) to CAMPFIRE households. The majority of households (62%) do not support elephant preservation

Title: The Economic Value of Congo Basin Protected Areas Goods and Services (2011)

Key: Cameroon, Central African Republic, Democratic Republic of Congo, Equatorial Guinea, Gabon, Republic of Congo

Estimated (Service Flow) Values:	Total economic value was estimated to be $603,468,014,907. Direct use values were estimated to be $13,884,954; indirect use values total $589,532,157,606; and option, existence, and bequest values are estimated to be $50,903,301. (Values are in U.S. dollars).

Title: "The Economic Value of Elephants, in the Assessment of South African Elephant Management: A Scientific Assessment of South Africa, edited by RJ Scholes and KG Mennell" (2008)

Key: Elephants, South Africa

Estimated (Service Flow) Values:	Table 1 provides a summary on valuation studies on African elephants (excluding studies from southern Africa) used in the study. The summary includes what has been valued, the valuation technique, the source, the values, and some remarks. Table 2 provides a summary of the main economic values of African elephants. Based on the economic values of elephants shown in Table 1, the value of an African elephant (in 2006/07 US$) was estimated to range from US$175 in terms of compensation costs to surrounding land owners to US$4,420 in terms of non-use values (mainly existence, bequest and experience value from European and American households). The total value of African elephants per annum was estimated to range from US$4.1-77.2 million from the consumptive value of ivory to US$2.2 billion from non-use values. The low value estimate of the direct consumptive use relative to its non-use and non-consumptive use values was expected since CITES included the African elephant in the list of endangered species in 1989. The non-use values were almost 30 times higher than the high-end estimation of the consumptive value of ivory.

Title: "Analysis of the Economic Significance of Gorilla Tourism in Uganda"Key: endangered species," (2000)

Key: Gorilla, Tourism

Estimated (Service Flow) Values:	Assuming an average consumer surplus of $196/tourist, the annual net economic benefit of gorilla tourism in the two parks at full capacity (8,760 tourists/year) is $1,716,960 (in US Dollars). Depending on the discount rates used, that annual net economic benefit value varies. At a 5% discount rate (conservation focused rate), the benefits account for $34,339,000 while at a 12% discount rate (Government of Uganda's official social discount rate) those benefits amount to $14,308,000. At a 22% discount rate (private time preference rate), the benefits are of $7,804,000. Because tourists are not expected to go see individual gorillas, the value has been accounted for viewing groups of mountain gorillas. Thus, at a 0% discount rate (ultra-conservationists or preservationists), the value of one of three groups available for viewing in two parks is infinity. At discounts of rate of 5%, 12% and 22%, the values are respectively of $11.4 million per group, $4.8 million/group and $2.6 million/group. The total sale impact for the whole country was estimated at $8.8 million/year, net foreign exchange earning of about $4.4 million, employment income of about $3.9 million, government revenue through various taxes at $2.7 million and an employment opportunity of 946 person years all from direct tourist expenditures of $7.6 million (table 1). In addition an annual benefit flowing to the Uganda Wildlife Authority of $2.1 million and to local communities ($678,000) is estimated (table 2). The economic value of gorilla tourism was estimated at $7.8-$34.3 million depending on the discount rate used.

European Union Action to Fight Environmental Crime

Marine Pollution

WP3 Quantitative Analysis

Deliverable No. 3.2d

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List of Abbreviations

BSAP	Baltic Sea Action Plan
BSIMAP	Black Sea Integrated Monitoring and Assessment Programme
EMSA	European Maritime Safety Agency
GDP	Gross Domestic Product
HELCOM	Baltic Marine Environment Protection Commission – Helsinki Commission
MARPOL	International Convention for the Prevention of Pollution from Ships
OGP	International Association of Oil and Gas Producers
OSPAR	Convention for the Protection of the Marine Environment of the North-East Atlantic
POP	Persistent Organic Pollutants
UNEP	United Nations Environment Programme

1 Introduction

1.1 Marine pollution

Marine pollution is a broad category, consisting of oil pollution (including accidents with offshore oil and gas installations) and all other marine pollution as defined e.g. in MARPOL and the London Convention.

MARPOL, the International Convention for the Prevention of Pollution from Ships, is the main international convention covering prevention of *pollution of the marine environment by ships* from operational or accidental causes. Its annexes list various forms of marine pollution, caused by oil, noxious liquid substances, harmful substances in packaged form, sewage and garbage from ships, etc.

The London Convention (Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter of 1972), which entered into force in 1975, aims to control pollution of the sea from *dumping*. It covers the deliberate disposal at sea of wastes or other matter from vessels, aircraft, and platforms.

In addition to the obvious (but difficult to quantify) *environmental* damage caused by marine pollution, there may be *health* damage as well as *social* and *financial* damage. The last two categories are taken together in this chapter and include damage to the operators of installations, lost profits for the tourism sector and fishing industry, etc. The term *economic* damage, like in other chapters of this report, is reserved for the total of all financial and monetised impacts, but only to the extent to which such information is available in existing studies or can be calculated.

It should be noted that marine pollution is not necessarily related to crime. However, the available data generally do not distinguish between intent, (gross) negligence, and other causes of marine pollution. In this chapter we will focus especially on dumping, leaving out accidents with offshore oil and gas installations.[1] Illegal waste shipment is analysed elsewhere in this report and will therefore not be discussed here either.

1.2 Availability of data

The data that are available in existing reports on marine pollution[2] always refer to a specific geographical area or to a specific sea. With respect to marine pollution caused by the discharge of particular substances (e.g. chemicals or oil), there are a number of sources containing either concrete data relating to a specific year, or more general information on trends (e.g. increase or decrease of pollution, increase or decrease in incidents). This chapter is constructed taking into account the information from both these kinds of sources.

The majority of sources concern *environmental* impacts. Some of the data contained in these reports is rather technical, as they refer to pollution levels caused by particular substances. We provide these data as presented in the original reports. Only some sources provide information on health or social impacts, and even fewer make quantitative data available.

[1] For a literature overview on these types of accidents, see EFFACE Deliverable 3.1.

[2] See Deliverable 3.1 and the additional sources contained in the list of references. The reports mentioned in Deliverable 3.1 covered marine pollution caused by accidents (more specifically, those involving offshore oil and gas installations); here we also present some additional sources on marine pollution caused by dumping.

1.3 Structure of this chapter

The remaining sections of this chapter discuss, respectively, methodology (section 2), quantitative impacts in the various seas (sections 3-7), and conclusions (section 8). This chapter is set up somewhat differently from other chapters in this report, because a wide variety of data relating to different seas was collected, and no economic analysis of these data could be conducted. This will be further explained in section 2.

The sections 3-7 on quantitative impacts provide a summary of data found in existing reports on marine pollution (dumping, littering, spills), by presenting information on environmental, health and social impacts, as well as data on financial damage, where available. The following waters are covered: Baltic Sea (3), Black Sea (4), Mediterranean (5), Arctic Sea (6), North Sea and English Channel (7).

The list of reports quoted is provided in a separate bibliography, included at the end of this chapter.

2 Methodology

This chapter includes a quantitative analysis of data, which we collected by means of an extensive literature survey. Considering the nature of these data, conducting a monetary analysis turned out to be impossible, as that would require monetising the environmental impacts of widely differing substances such as (but not limited to) heavy metals, persistent organic pollutants, and oil. Moreover, environmental pollution does not only have an impact on water quality, but also on coastal areas and animal welfare.

It is even more difficult to monetise any social impact of marine pollution, e.g. on fisheries and tourism, more particularly because it is difficult to obtain reliable quantitative impact data upon which such monetisation can be done.[3] Only some rough estimates exist in that respect. Moreover these data cannot be extrapolated, due to the fact that the sectors concerned differ enormously in importance (e.g. in terms of percentage of GDP) for each country.

With regard to accidents, more information is available, for example on types of accidents and fatalities, primarily through the European Maritime Safety Agency (EMSA) reports. However, these reports do not contain information on economic impacts (defined in the narrow sense of financial and monetised impacts).

The added value of this chapter lies in the collection and presentation of data on impacts and accidents; and in the classification of this information into the different heading for environmental, health, social and financial, and economic impacts.

It is important to note that the impacts of incidents that contribute to chronic problems in the marine environment are difficult to distinguish from long-term degradation. For example, for degradation of the marine environment by pollution, it can be hard (other than for major events) to determine the relative importance of 'legal' pollution (from permitted activities), accidents and illegal incidents in their cumulative effects. As a result, this chapter examines some of the impacts as a whole.

The data were collected mainly through web research and scanning of journal databases. Efforts were made to include as much publicly available sources as possible. The following sources proved to be especially useful:

- Baltic Sea: HELCOM, EMSA, articles and reports
- Black Sea: BSC, EMSA, articles and reports (mainly from Bulgaria)
- Mediterranean: articles and reports (mainly from Greece), EMSA
- Arctic: articles and reports only
- North Sea and English Channel: DG Environment, EMSA, UNEP, articles and reports.

[3] Newman et al (2015) conclude that in industries such as fisheries and tourism, the costs of marine litter are beginning to be quantified and are considerable. In other areas such as impacts on human health, or more intangible costs related to reduced ecosystem services, more research is needed.

3 Quantitative Impacts: Baltic Sea

As explained by the European Maritime Safety Agency (EMSA), the EU coast of the Baltic Sea and its approaches include the coastlines of Sweden, eastern Denmark, north-eastern Germany, Poland, Finland, Estonia, Latvia and Lithuania. The Baltic coast also includes two regions of the Russian Federation: the eastern end of the Gulf of Finland and Kaliningrad. Any data taken from the EMSA reports do not include the latter two regions.[4]

The data available in relation to marine pollution in the Baltic Sea is relatively detailed when compared to the other seas discussed in this report. This applies especially in relation to environmental impacts, as will be indicated in the following subsections.

3.1 Environmental Impacts

Presence of Hazardous Substances

HELCOM, the Baltic Marine Environment Protection Commission of the Helsinki Commission, defines hazardous substances as substances (including synthetic or natural compounds) that "cause adverse effects on the ecosystem and human health by being toxic, persistent and bioaccumulating." Heavy metals such as mercury, cadmium, and lead are toxic to organisms at high concentrations, whereas persistent organic pollutants (POPs), such as PCBs and organotin compounds, may be toxic even at low concentrations.[5]

In the period 1998-2007, all open sea areas of the Baltic Sea, except for the Kattegat, were classified as being 'disturbed by hazardous substances'. 98 of the 104 coastal assessment units were classified as being 'disturbed by hazardous substances', while only 7 out of the 144 assessment units were considered to be 'under-disturbed by hazardous substances'. The main basin of the Baltic Sea, together with certain parts of the Kiel and Mecklenburg Bights, were the areas most disturbed by hazardous substances.[6].

Organic Pollutants

In 2010, HELCOM identified a decreasing trend in the level of persistent organic pollutants in the Baltic Sea, and argued that such decrease may be due to bans or restrictions on the production or use of these substances.[7] Nevertheless, In the Baltic Sea, substances such as PCBs, lead, mercury and several others, still appear as contaminants with the highest concentrations in relation to the threshold levels set by HELCOM.

Marine Litter

HELCOM (2006) identifies that the amount of macroscopic marine litter amounts to less than 20 particles per 100 meters of coastal strip. However, this amount sometimes goes up to 700-1200.[8]

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[4] EMSA 2011, p. 27.

[5] HELCOM 2010(a), p. 18.

[6] HELCOM 2010(a), p. 18.

[7] HELCOM 2010(a), p. 20.

[8] HELCOM 2006, p. 4.

Oil spills

At least until 2008, no significant illegal and accidental hydraulic oil spill from ships has occurred since the 'Fu Shan Hai' incident in 2003. That particular incident resulted in the release of 318 tonnes of fuel oil after 616 tonnes had been recovered from the Baltic Sea.[9]

HELCOM identifies a decreasing number of deliberate illegal oil discharges, from 763 spills in 1989, to 210 spills in 2008.[10]

Incidents

The EMSA reported that 75 ships were involved in incidents at the Baltic Sea in 2009. This number includes sinkings, groundings, collisions, fires/explosions and other types of accidents.[11] 89 vessels were involved in incidents in 2010, which is an increase of 19% compared to 2009, but significantly lower than the 120 incidents that were reported in 2008.[12] The table below (adapted from EMSA 2011) provides an overview of all accidents, which seems to indicate that there is no particular trend that can be discovered in these numbers.

Types of accident	**2007**	**2008**	**2009**	**2010**
Sinkings	3	5	3	2
Groundings	49	52	33	32
Collisions/Contacts	23	35	24	28
Fires/Explosions	16	17	10	13
Other types	15	11	5	14
Total	106	120	75	89

According to Hänninen and Rytkönen (2006), around 80% of all accidents that took place in the Baltic Sea are due to human factors, such as improper handling of the cargo, inadequate supervision and navigational errors, and machine breakdowns and other technical problems.[13]

General information on environmental impacts

HELCOM identifies eutrophication and overfishing as the two main causes of ecosystem destruction in the Baltic Sea.[14]

Between 2004 and 2006, the Swedish Coast Guard detected on average 308 spills per year.[15]
EMSA points out that the relatively low level of accidents in 2009/2010, compared to the previous years, coincided with the economic crisis.[16]

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[9] HELCOM 2010(a), p. 31.

[10] HELCOM 2008(b), p. 7.

[11] EMSA 2010, pp. 27-28.

[12] EMSA 2011, p. 27.

[13] Hänninen and Rytkönen 2006, p. 6. The authors, studying the transportation by tankers of liquid bulk chemicals, identify the main accidents and divide them in three main categories: (i) accidents caused by *improper handling of the cargo*; (ii) accidents caused by *inadequate supervision and navigational errors* ; and (iii) accidents caused by *machine breakdowns and other technical problems*. Examples of each type of accident are provided in their report.

[14] HELCOM 2010(a), p. 5.

[15] Mullai et al. 2009, p. 323.

[16] EMSA 2011, p. 26.

3.2 Health impacts

No data could be found on health impacts in relation to pollution of the Baltic Sea.

3.3 Social and financial impacts

HELCOM points out that overfishing in the Baltic Sea does not only represent a cost for the environment, but also for the fishing industry itself. Some general data are provided regarding the economic value of the fishing and tourism industries.[17]

- Over 50.000 people are employed in the fishing sector of the Baltic Sea;
- The annual turnover of this industry has been estimated at €4.5 billion;
- Sport fishery has a yearly expenditure in Sweden of €265 million per year. In Finland, Denmark and Sweden together – the expenditure reaches €700 million per year;
- Fishing also has a cultural value, which has been calculated to be €200 million per year in Sweden.
- Tourism in the Baltic Sea is estimated at €90 billion per year.
    - Cruise tourism gives an annual turnover of €433 million per year;
    - The leisure boat industry in Sweden has an annual turnover of €265 per year.

3.4 Economic impacts

There is no information on economic (monetized) impacts caused by pollution of the Baltic Sea. However, there is information on the financial benefits estimated by HELCOM in relation to some pollution reduction targets. HELCOM developed a Baltic Sea Action Plan (BSAP) in order to combat the continuing deterioration of the marine environment resulting from human activities, and ultimately to improve the environmental conditions.[18] The benefits of achieving the BSAP target regarding eutrophication (caused by nutrient pollution) in the entire Baltic Sea region were estimated to be

- €4 830 million per year: benefits for avoiding effect of eutrophication estimated on the basis of the willingness of people to pay;
- €2 564 million per year: total benefits of improved water-quality based on meta-analysis.

[17] HELCOM 2010(a), p. 52.

[18] HELCOM 2010(a), p. 52.

These figures are derived in relation to the targets of the BSAP, which are as follows: [19]

Eutrophication	Hazardous Substances
• Concentrations of nutrients close to natural levels • Clear water • Natural level of algal blooms • Natural distribution and occurrence of plants and animals • Natural oxygen levels	• Concentrations of hazardous substances close to natural levels • All fish are safe to eat • Healthy wildlife • Radioactivity at the pre-Chernobyl level
Biodiversity	**Maritime Activities**
• Natural marine and coastal landscapes • Thriving and balanced communities of plants and animals Viable populations of species	• Enforcement of international regulations – no illegal discharges • Safe maritime traffic without accidental pollution • Efficient emergency and response capabilities • Minimum sewage pollution from ships • No introductions of alien species from ships • Minimum air pollution from ships • Zero discharges from offshore platforms Minimum threats from offshore installations

[19] For more information on the specific targets to be achieved, see HELCOM 2007.

4 Quantitative impacts: Black Sea

The Black Sea is one of the main inland sea areas around the EU. The EU parts of the Black Sea include the coastlines of Bulgaria and Romania. Other areas (Turkey, Georgia, Russia and Ukraine) may not always be included in the data below, notably the data on accidents provided by EMSA.[20] Furthermore, some data below only refer to the Bulgarian Black Sea.

4.1 Environmental impacts

Organic pollution discharge

According to Dineva, between 1998 and 2005, organic pollution discharge in the Bulgarian Black Sea varied between 3 tyr^{-1} (the Dyavolska River) and 1040 tyr^{-1} (the Veleka River).[21]

Eutrophication

Between 1998 and 2005, the total orthophosphate phosphorus discharge into the Bulgarian Black Sea by rivers ranges from 65 t P yr^{-1} to 1141 t P yr^{-1}, with the Kamchia River's rate between 36 and 222 t P yr^{-1}.[22]

Nitrogen and phosphorus emissions have been reducing in the last years. However the 2000-2005 values are still 1.5 points higher than their pristine levels between 1955-1965, as indicated by the Commission on the Protection of the Black Sea Against Pollution (BSC).[23]

Heavy metals discharge

Between 2003 and 2005, heavy metal discharge into the Bulgarian Black Sea by rivers is mainly formed by the Kamchia River:[24]

- Total *cadmium* discharge - up to 10 t yr^{-1},
- Total *zinc* discharge - up to 125 t yr^{-1},
- Total *lead* discharge - up to 118 t yr^{-1},
- Total *copper* discharge – up to 44 t yr^{-1}

Petroleum Hydrocarbons Discharge

The total petroleum hydrocarbons discharge into the Bulgarian Black Sea by rivers between 2004 and 2005 is up to 458 t yr^{-1}, with the Veleka River's discharge - up to 116 t yr^{-1}, and the Rezovska River's discharge - up to 50 t yr^{-1}.[25]

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[20] Notably the data on EMSA 2011, p. 29.

[21] Dineva 2011. The document does not contain page numbers, so a precise reference cannot be provided.

[22] Dineva 2011.

[23] BSC 2008, para. 2.6.

[24] Dineva 2011.

[25] Dineva 2011.

Between 1995 and 2005 the mean concentration of petroleum hydrocarbons in the bottom sediments of coastal areas of the Black Sea varied from very low levels to up to 0.8 mg/g.[26]

The most polluted coastal areas, exceeding the average concentration of 13-16 times, are located in Romanian, Turkish and Russian waters. These values are normally registered near large ports, refineries, or oil terminals for transportation.[27]

The maximum values were registered at Romanian and Turkish coasts at very shallow depths, amounting to c. 12 mg/g. These values are most likely due to fresh oil spills in 2005.[28]

Chlorinated pesticides

Most measurements by the Black Sea Integrated Monitoring and Assessment Programme (BSIMAP)[29] were below the detection limit (0.05 ng/l). However, some very condensed patches were detected, in particular near Romanian coastal waters near the town of Mangalia in April 2005.[30]

HCHs and DDTs are the most common pollutants in bottom sediments in the Black Sea. The EHP levels are considered to be 0.25 ng/g for γ-HCH and 12.5 ng/g for DDTs total.[31]

- HCH Pollution: the highest levels of HCH pollution were registered in Ukraine in 1992 (4.5 ng/gin) and in Romania in 1993 (29.0 ng/gin).
- DDT Pollution: the highest levels of DDT pollution were registered in the Odessa area in 2003 (63950 ng/g). According to BSC 2008, those levels can only be explained as an accidental event.

Waste water treatment

Dineva identifies municipal waste water treatment plants as one of the causes of pollution of the Bulgarian Black Sea. The plants discharging above 5000 $m^3\ d^{-1}$ are those of Varna, Dobrich, Devnya, Golden Sands, and Albena. The main ones discharging below 5000 $m^3\ d^{-1}$ are those of General Toshevo, Kavarna, Dolni Chiflik, Beloslav, and Provadia.[32]

Dineva (2007) considers that large amount of eutrophication matter comes into the Bulgarian Black Sea due to lack of biological treatment in the waste water treatment plants.

Incidents

The EMSA reported that 18 ships were involved in incidents at the Black Sea in 2010. This number includes sinkings, groundings, collisions, fires/explosions and other types of accidents, with an increase of 150% compared to 2009, and of 64% when compared to 2008.[33] With a percentage of 45%, collisions were found to be the predominant type of accident.[34] Mainly as a result of the Karim I sinking, 7 people were reported to have lost their lives in accidents in 2010.[35]

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[26] BSC 2008, para. 3.12.

[27] BSC 2008, para. 3.12.

[28] BSC 2008, para. 3.12.

[29] For more information, see http://www.blacksea-commission.org/main.asp.

[30] BSC 2008, para. 3.2.1.

[31] BSC 2008, para. 2.2.2.

[32] Dineva 2011.

[33] EMSA 2010, p. 30.

[34] EMSA 2010, p. 30.

[35] EMSA 2010, p. 30.

The following table includes data collected by EMSA on accidents relating both to the Black Sea and the Mediterranean Sea.[36]

Types of accident	**2007**	**2008**	**2009**	**2010**
Sinkings	11	9	3	9
Groundings	20	37	20	23
Collisions/Contacts	63	76	71	70
Fires/Explosions	20	13	11	16
Other types	14	14	9	26
Total (Black Sea & Mediterranean)	128	149	114	144

4.2 Health impacts

According to Rudneva, the most important health-related manifestation of marine degradation is the presence of microorganisms from infected sea water and subsequent consumption of contaminated seafood.[37]

4.3 Social and financial impacts

Fisheries

Up to 150,000 people were estimated to be economically dependent on Black Sea fisheries. Wages loss in processing plants is estimated at $10 mln annually.[38]
Due to pollution and introduction of alien species, only 5 of the 26 commercial fish species abundant in the 1970s in the Black Sea were still commercially viable in the 1990s. Black Sea fisheries, which supported about 2 mln fishers and dependents, suffered almost total collapse. Catch values from the mid 1980s to the early 1990s declined by about US$240 million.[39]

Tourism

According to Rudneva (2003), in the 1980s over 4 mln people visited the Black Sea coastline each summer. This number however declined in the 1990s. Rudneva estimates that this is most likely due to the deterioration of amenity values caused by pollution and eutrophication.

4.4 Economic impacts

There is no information on other economic impacts caused by pollution of the Baltic Sea, except from the information on financial losses for the fishing industry provided in the previous section.

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[36] EMSA 2010, p. 29.

[37] Rudneva 2003.

[38] BSC 2008, para. 11.4.1.

[39] BSC 2008, para. 11.4.1.

5 Quantitative impacts: Mediterranean Sea

EMSA (2011) reports that the EU parts of the Mediterranean Sea comprise the coasts of eastern Spain, southern France, Italy, Malta, Slovenia, Greece and Cyprus. The North African, eastern Adriatic, and eastern Mediterranean countries are not included in the EMSA data. Furthermore, some sources are focused only on the Ionian Sea and/or Aegean Sea.

5.1 Environmental impacts

Oil concentration

Ventikos and Psaraftis calculate that the oil concentration in the Mediterranean Sea has increased from 2 to 5 μg/l up to 100m of depth from 1981 to 1993.[40]

Incidents

The Hellenic Coast Guard has analysed the substances spilled into the Greek Sea between 1995 and 1996. Most incidents (more than 400) resulted to involve oil and petrol products. Moreover, over 200 incidents involved industrial run off.[41]

Ventikos and Psaraftis analysed the types of accident between 1978 and 1995 leading to oil pollution.[42]

- Sinking: 0%
- Grounding/Stranding: 1.800%
- Fire/Explosion: 1.542%
- Collision/Ramming: 11.020%
- Engine Trouble: 0%
- Rest: 9.258%

EMSA data concerning incidents at the Mediterranean Sea (combined with data on the Black Sea) was reported in the previous section.

5.2 Health impacts

The only information available in relation to health impacts are data on fatalities (see below) and animal health (presented in section 5.3)

[40] Ventikos and Psaraftis 1998, para. 1.

[41] Hellenic Coast Guard 1996.

[42] Ventikos and Psaraftis 1998, para 1.

Fatalities resulting from ship casualties

OGP identified the number of fatalities arising from ship casualties between 1996 to 2005 for some countries on the Mediterranean Sea.[43]

- Egypt: 4 fatalities out of 2 shipping casualties;
- Greece: 24 fatalities out of 11 shipping casualties;
- Italy: 11 fatalities out of 5 shipping casualties

5.3 Social and financial impacts

Fisheries

Storelli et al. measured the presence of mercury in several species of fish in June-August 1999 from the Ionian Sea.[44]

- Albacore (*thunnus alalunga*): between 0.84 to 1.45 mg kg^{-1} w.w.
- Bluefin tuna (*thunnus thynnus*): between 0.16 and 59 mg kg^{-1}

Storelli et al. also found trace metals in various tissues and organs of loggerhead turtles. Hepatic tissue (Hg: 0.43 µg g^{-1} wet weight; Cd: 3.36 µg g^{-1} wet weight) and kidney (Hg: 0.16 µg g^{-1} wet weight; Cd: 8.35 µg g^{-1} wet weight) exhibited the highest levels of mercury and cadmium.[45]

These data, which strictly speaking refer to environmental impacts, are nevertheless presented here as social impacts due to the importance of the fishing industry in the Mediterranean Sea. The 2014 Annual Economic Reports on the EU Fishing Fleet provides interesting data in that respect.[46] The main fleets to be considered in terms of value of landings are the Italian (€ 925 million) and the Spanish (€ 267 million euro) fleets, which on their own account for around 91% of the value of landings of the Mediterranean and Black Sea fleet.[47] The 2012 revenue generated by the whole Mediterranean and Black Sea fleet (excluding Spain), was estimated to be € 1,045 million. Italy is accountable for 87% of this amount (€ 910 million).[48]

5.4 Economic impacts

No data could be found on economic impacts in relation to pollution of the Mediterranean Sea.

[43] OGP 2010, p. 14.

[44] Storelli et al. 2002.

[45] Storelli et al. 2005, p. 164.

[46] See European Commission 2014. The authors of this report point out that these statistics are incomplete, because several Member States failed to provide data.

[47] European Commission 2014, p. 115-116.

[48] European Commission 2014, p. 116.

6 Quantitative impacts: Arctic Sea

The Arctic is surrounded by the United States (Alaska), Canada, Greenland, Russia and Norway. Data on impacts of environmental pollution are more difficult to find.

6.1 Environmental impacts

Pollution sources in the Artic Sea

The Arctic does not have significant pollution sources of its own; however it is a recipient of chemical contaminants released elsewhere in the world.[49] Poland et al (2003) note that the Arctic has very seriously polluted sites that are as bad as sites anywhere else in the world.[50]

Levy, in a 1986 study on the Canadian Arctic marine environment, found that baseline data on hydrocarbons in the eastern Arctic show that Arctic marine waters are clean in comparison with marine waters in the mid-latitudes.[51]

- Hydrocarbons in the water column
    - Davis strait: 0.53^2 µg/l^{-1}
    - West Lancaster Sound: 0.40 µg/l^{-1}
    - N. Baffin Bay: 0.52 µg/l^{-1}
    - Hudson Strait entrance: 0.35 µg/l^{-1}
    - Hudson Bay/Foxe Basin: 0.49 µg/l^{-1}
- Hydrocarbons in the surface microlayer
    - N. Baffin Bay: 8.0 µg/l^{-1}
    - N. E. Baffin Is. Shelf: 12.3 µg/l^{-1}
    - Hudson Strait entrance: 4.1 µg/l^{-1}
    - Hudson Strait/Hudson Bay/Foxe Basin: 12.4 µg/l^{-1}

Long range airborne contamination

According to Poland et al., major atmospheric pathways converging on the Arctic transport organic and metal pollutants, acidifying compounds and radioactive contaminants.[52]

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[49] Barrie et al. 1992.

[50] Poland et al. 2003, p. 377.

[51] Levy 1986. Note that these data are almost 30 years old.

[52] Poland et al. 2003, p. 372.

Animal welfare

Moore identified changes in habits of marine mammals as a result of water pollution in the Arctic sea.[53] Some of the affected behaviours are:

- One week-delay in southbound migration;
- Increase in calf production coincident with ice-free Chirikov basin in early spring;
- Reduction in calf numbers and changes in timing of occupation of breeding lagoons by gray whales;
- Lack of gray whales feeding during July in the Chirikov Basin;
- Gray whales feeding year-round offshore Kodiak Island, Alaska;
- Gray whale calls detected in the western Beaufort Sea over the winter of 2003/2004;

Poland et al. also estimate that after the Exxon Valdez oil spill more than 35,000 bird carcasses and 1000 sea-otter carcasses were retrieved in the Arctic and Antarctic [54]

6.2 Health impacts

Barrie et al. find that the Arctic ecosystem is particularly sensitive to contaminants because the highly lipophilic and persistent nature of contaminants causes them to accumulate in the lipid-rich tissues of animals at the top of the food chain (polar bears, whales and seals), which represent the basis of the diet of the inhabitants of the Arctic regions.[55]

6.3 Social and financial impacts

No data could be found on social and/or financial impacts in relation to pollution of the Arctic Sea.

6.4 Economic impacts

No data could be found on economic impacts in relation to pollution of the Artic Sea.

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[53] Moore 2008.

[54] Poland et al. 2003, p. 377.

[55] Barrie et al. 1992. See in that respect also Poland et al. 2003, p. 377, who note that levels of contamination in the Arctic are such that some indigenous peoples who rely on a traditional diet are among the most exposed in the world to certain contaminants.

7 Quantitative impacts: North Sea and English Channel

This region includes the coastlines of north-western France, the UK, Ireland, Belgium, the Netherlands, north-western Germany, western Denmark, Norway and Iceland. According to EMSA (2011), the northern part of the coastline of this region is particularly prone to accidents, due to weather effects and the density of shipping operating in that area.[56]

7.1 Environmental impacts

Plastic waste

A report by DG Environment of the European Commission states that the presence of plastic waste is monitored by the OSPAR Pilot Project on Monitoring Marine Beach Litter in the North Sea.[57] This project found the following:

- Greater North Sea Coast: 80% of beach litter; 900 items of litter per 100m of beach
- Southern North Sea Cast: 75% of beach litter; 400 items of litter per 100m of beach
- Celtic Sea Coast: 70% of beach litter; 650 items of litter per 100m of beach
- Iberian Coast and Bay of Biscay: 62% of beach litter; 200 items of litter per 100m of beach.

According to the European Commission's report, there is little information on the amounts, rates, fate or impacts of plastic waste on land, but there has been a major effort done by Barnet et al. (2009) to quantify impacts on shorelines and sea.

Oil concentration

Ventikos and Psaraftis found that oil concentration in the North Sea increased from 0.2 to 2.5 μg/l up to 100m of depth in the period from 1981 to 1993. They attributed this sharp increase to the number of platform activities in the area.[58]

Incidents: data

The EMSA reported that 437 ships were involved in incidents in 2009 in the North Sea and English Channel. [59] In 2010, 411 vessels were involved in incidents. There has been a decreasing trend (at least) since 2007, when the total number of incidents was 528. This is also depicted in the table below, adapted from EMSA (2011).[60]

[56] Adapted from EMSA 2011, p. 26.

[57] European Commission 2011, p. 8.

[58] Ventikos and Psaraftis 1998, para. 1. For more info on offshore installations, see EFFACE Deliverable 3.1.

[59] EMSA 2010, p. 32.Included in these data is the Atlantic Coast (Portugal, Spain, south-western France).

[60] EMSA 2011, p. 26. Included in these data is the Atlantic Coast (Portugal, Spain, south-western France).

Types of accident	2007	2008	2009	2010
Sinkings	41	47	22	21
Groundings	128	128	124	88
Collisions/Contacts	218	197	197	190
Fires/Explosions	55	59	46	54
Other types	86	54	48	58
Total	528	485	437	411

Tricolor incident

Schallier, Resby and Merlin analyse the Tricolor Incident. A collision took place on 14 December 2002 between the car carrier 'Tricolor' and the containership 'Kariba' in the French EEZ. The Tricolor sank on the spot.[61]

- The vessel carried 1988 tons of at least four different heavy fuel oils (HFO), 167 tons of marine diesel oil (MDO), some 50 tons of lubricating oil, and several tons of gas oil and gasoline;

Despite precautionary measures, other collisions resulting in pollution took place, stemming from the original incident:

- 16 December 2002: the small vessel 'Nicola' collided with the Tricolor wreck;
- 1 January 2003: the tanker 'Vicky', carrying 66.000m^3 of diesel, collided with the wreck. An amount of HFO escaped from the Tricolor, and the Vicky lost 200m^3 of oil;

On 22 January 2003, the hull of the wreck got damaged during the salvage operations and resulted in a major spilling accident. At least 200m3 of HFO were released.

Animal welfare

The DG Environment report underlines that once an animal is entangled in plastic waste, it can drown, incur wounds, or be less able to catch food. Ingestion of plastic waste and of micro plastics can also lead to the death of the animal.[62]

UNEP measured the level of plastic in the stomachs of stranded seabirds in the North Sea. The highest levels have been found in the 1990s, and nowadays levels are similar in quantity to those measured in the 1980s. A change in source has been found: nowadays' plastic mostly is of industrial nature, while in the 1980s consumer and industrial plastic were roughly at the same level.[63]

7.2 Health impacts

The only information available in relation to health impacts concerns some basic data on fatalities (presented here) and animal health (presented in the previous subsection).

[61] Schallier, Resby and Merlin 2004, para. 1.2.

[62] European Commission 2011, p. 16.

[63] UNEP 2011, p. 24.

Fatalities

The EMSA[64] found that 48 lives were reported lost in the area in 2010, higher than in 2009 when 34 lives were lost.[65]

The great majority of accidents in the region in 2010 took place in the waters around Germany, the Netherlands, Norway and the UK.[66] The highest number of accidents is reported to have taken place in the region's main bottlenecks.

The International Association of Oil and Gas Producers (OGP) identifies the number of fatalities arising from ship casualties between 1996 to 2005 for some countries on the North Sea.[67]

- Denmark: 31 fatalities out of 7 shipping casualties;
- Germany: 3 fatalities out of 3 shipping casualties;
- Netherlands: 3 fatalities out of 1 shipping casualty;
- Norway: 10 fatalities out of 4 shipping casualties;
- UK: 0 fatalities out of 0 shipping casualties.

7.3 Social and financial impacts

Fisheries

The UNEP estimates that marine litter costs to the Scottish fishing industry between $15 and $17 million per year, equating to about 5% of the total revenue of affected fisheries.[68]

Navigational Hazard

The UNEP reports that there have been 286 rescues of vessels with fouled propellers in the UK and in 2008, costing up to $2.8 million.[69]

7.4 Economic impacts

In addition to the financial impacts presented in section 7.3, here we present estimates of cleanup costs.

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[64] EMSA 2011, p. 27.

[65] EMSA 2010, p. 32.

[66] EMSA 2011, p. 27.

[67] OGP 2010, p. 14.

[68] UNEP 2011, p. 28.

[69] UNEP 2011, p. 28.

Cleanup of beaches and waterways

Mouat et al. estimate that cleanup of beaches and waterways in Belgium and the Netherlands amounts to about $13.65 million per year, while cleanup of beaches and waterways for municipalities in the UK costs about $23.62 million per year.[70]

[70] Mouat et al. 2010. The average cost per municipality is higher in the Netherlands and Belgium than in the UK, as these countries are more densely populated.

8 Conclusions

Different from the other chapters contained in this WP3 report, the scope of this chapter on marine pollution has been open-ended, focusing on various types of pollution (ranging from e.g. waste discharges and marine litter to oil and shipping incidents) and on five different seas. This chapter did not include a monetary analysis, but instead presented and categorized various types of data on environmental, health, social, financial and economic impacts in relation to the five seas covered here: Baltic Sea, Black Sea, Mediterranean, Arctic, and North Sea.

It is striking that the availability of data differs for each sea, depending on whether or not there are specific agencies involved. HELCOM in particular provides relatively detailed data on environmental impacts concerning the Baltic Sea. There are also some detailed reports on the North Sea, inter alia published by the European Commission. Some of the data presented throughout this report follow from academic or individual research, e.g on pollution of the Bulgarian Black Sea.

This chapter presented mostly data on environmental impacts. Sometimes the data source focused only on one type of environmental pollution, e.g. caused by plastics or littering. Obviously, these data sources are difficult to add together, even for one Sea. Extrapolation of data is not an (easy) option either. Social and financial impacts, in particular, are likely to differ greatly between jurisdictions. For these reasons (and many others), presenting an estimate of the overall impact of marine pollution, even for one sea, seems impossible. However, as indicated in the overview chapter (D3.2a), this has never been the aim of any of the chapters included in this EFFACE report.

We were able to find rather detailed data on the number of accidents and the number of fatalities in relation to marine pollution, especially those data collected by ESMA. These data can in principle be combined for the various seas, e.g. if one would like to know the number of accidents and fatalities across all European seas. However, no trends over time can be detected, except for (for some seas) a small decreasing trend, explained by EMSA by the economic climate.

Of course, accidents do not necessarily relate to illegal activities; the data do not indicate whether or not these accidents result from a violation of particular legal rules, such as safety requirements, or from grossly negligent behaviour. The same argument applies to e.g. oil pollution incidents. Separating legal from illegal activities in relation to marine pollution, as for many other areas covered in this report, is not possible based on the data we were able to find and present here.

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European Union Action to Fight Environmental Crime

Illegal e-waste shipments from the EU to China

Quantitative and monetary analysis of illegal shipments and its environmental, social and economic impacts

Deliverable 3.2e

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DISCLAIMER

ABSTRACT

This report provides a quantitative and monetary analysis of the environmental, human health and economic impacts of illegal e-waste shipments from the EU to China. The European Union's Waste Shipment Regulation bans the export of hazardous waste, including waste electrical and electronic equipment (WEEE), from the EU to non-OECD countries in the developing world. Since 2000, an import ban of e-waste has been officially implemented in China. Nonetheless, vast amounts of e-wastes from the EU are illegally entering China. This case study presents some of the key estimates of the scale of this e-waste trade and calculates the total volumes of e-waste that have been imported in China from the EU in 2005 and 2012. These illegal EU e-waste shipments into China affect the environment, human health and economies at multiple levels and carry significant risks on the ground. This report provides an overview on the quantitative environmental impacts of informal e-waste recycling in China, including impacts on air, dust, soil, sediments, and plants and presents the quantitative impacts of elevated lead levels in human body and IQ score of children in China. Furthermore, the negative impacts on water supply are due to contamination from e-waste recycling are monetised. Finally, the report provides an estimate on the EU e-waste recycling industry's economic loss and the job losses in the EU e-waste recycling industry as a result of these illegal e-waste shipments.

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LIST OF ABBREVIATIONS

BMD	benchmark dose
BMDL	benchmark dose level
bw	body weight
CAER	Coalition for American Electronics Recycling
Cd	Cadmium
EEE	electrical and electronic equipment
EFSA	European Food Safety Authority
EPA	Environmental Protection Agency
FAO	Food and Agriculture Organisation
FEVC	forced expiratory vital capacity
FoE	Friends of the Earth
FTE	full-time equivalent
FVC	forced vital capacity
kt	kilo tonnes = 1000 tonnes
µg	microgram=10^{-6} grammes
ng	nanogram=10^{-9} grammes
NGO	non-governmental organisation
OECD	Organisation for Economic Co-operation and Development
PAHs	polycyclic aromatic hydrocarbons
Pb	lead
PBDEs	polybrominated diphenyl ethers
PCBs	polychlorinated biphenyls
PCDD/Fs	polychlorinated dibenzo dioxins/furans
pg	picogram= 10^{-12} grammes
PM	particulate matter
POPs	persistent organic pollutants
PPP	purchase power parity
SD	standard deviation
StEP	Solving the E-Waste Problem

TDI	total daily dietary intake
TEQ	toxic equivalent
TSP	total suspended particles
UEEE	used electrical and electronic equipment
UNODC	United Nations Office on Drugs and Crime
WEEE	waste electrical and electronic equipment
WHO	World Health Organisation
WP	work package
WSR	Waste Shipment Regulation
Zn	zinc

Executive summary

In recent years, the cross-border transport of waste electrical and electronic equipment (WEEE) has significantly increased, which also led to the rise of illegal shipments of e-waste from developed countries to the developing world. At the international level, it is the Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and their Disposal which seeks to provide enhanced control over the transboundary movements of hazardous wastes, including e-waste. The EU, together with its Member States, is a party to the Basel Convention since 1994 and the convention is translated into EU law via the EU Waste Shipment Regulation, which bans the export of hazardous waste, including e-waste, from the EU to non-OECD countries in the developing world. Despite China's official e-waste import ban, which was introduced in 2000, China represents the largest downstream destination for e-waste exported from Europe and North America. According the United Nations Office on Drugs and Crime, approximately 80 per cent of the total global amount of e-waste goes to Asia, out of which 90 per cent ends up in China (UNODC, 2013). Most of the illegally imported e-waste enters China's informal refurbishment and recycling sector, which lacks environmental, health and safety standards.

The illegal EU e-waste shipments into China affect the environment, human health and economies at multiple levels and carry significant risks on the ground. This study aims to provide an insight into the negative economic impacts in the EU and the negative environmental and human health impacts in China and aims to quantify and monetise these impacts. Despite the growing body of literature available on illegal e-waste shipments empirical data suffers from high uncertainties and data for instance on the volumes of illegally exported e-waste from the EU to China cannot be considered fully reliable. Furthermore, calculations of monetary figures on the environmental, health and economic impacts of these e-waste trades have to often use assumptions and extrapolations. The e-waste volume figures and monetary impacts presented in this case study therefore should be seen as rough estimates and should be treated with caution.

Volumes of e-waste shipped from the EU to China

Quantifying the illegal export of e-waste from the EU (to China) is especially challenging as there is very little clear information upon which estimates can be based. For this exercise, estimates from 2005 and 2012 have been used on the following aspects:

- The amount of e-waste generated in the EU.
- The illegal share of the exported amount of e-waste from the EU.
- The EU share of illegally imported e-waste into China.

Using the above information it was estimated that **for 2005 and 2012 respectively around 0.74 and 1.16 million tonnes of e-waste have been imported in China from the EU**. In order to incorporate the underlying uncertainties related to these estimates in this case study we also propose a 'minimum China import scenario' and a 'maximum China import scenario'.

Environmental impacts

The illegal export of e-waste from the EU to China has resulted in the release of large amounts of contaminants in the local environment. The potential annual emissions of some environmental contaminants associated with e-waste imported illegally in China from the EU were estimated. It was for instance estimated that respectively **10 and 16 tonnes of PCBs from EU e-waste were potentially released in the Chinese environment in 2005 and 2012**.

The environmental impacts of the activities in the informal refurbishment and recycling sector lead to economic losses and additional costs. For example, due to local contamination of soils and water resources, drinking water needs to be brought in from other regions. It is estimated that in Guiyu, with an population of about 150,000 in the year 2013, establishing a piped water supply resulted in annual additional cost of around €1.6 million.

Human health impacts

As to health impacts, this study demonstrates that illegal exports from the EU are significantly increasing incidences of chronic disease in China, threatening not just workers but also current residents of e-waste recycling areas and adjacent regions and future generations. In order to demonstrate the health impacts, figures on the routes of exposure, on the human body burden and on scientifically established associations and/or causal links between e-waste exposure and several kinds of diseases and other health impacts are presented. Furthermore, a more detailed quantification was carried out for the impacts of lead poisoning resulting from e-waste exposure on children's neurological development, expressed in terms of children's IQ scores. For China as a whole it is conservatively estimated that around **81,300 children (58,000-93,000) born in the period 1995-2013 have been affected in their neurological development** as a result of e-waste exposure. It was subsequently estimated that these children in China **lost about 97,560 IQ points (69,600-111,600)** as a result of informal e-waste recycling and dumping activities. This amounts to **an average reduction of intelligence of 1.2 points per child**.

Economic impacts

As to the economic impacts of illegal exports of e-waste, it is estimated that the 2.98 million tonnes of illegally exported e-waste from the EU in 2012 correspond roughly with **€31.2 million to €37.5 million loss in profits to the EU e-waste recycling industry**. If one looks at the e-waste **exports to China only (1.16 million tonnes in 2012), the EU recycling industry is estimated to have lost €12.2 million to €14.6 million in profits in 2012**. Assuming that the average intrinsic value of WEEE is about €300 per tonne, **the economic value lost to the EU as a result of illegal exports to China is roughly estimated at €348 million for 2012 only**. The **economic value lost to the EU** as a result of all illegal exports out of the EU is **estimated at €892 million for 2012**.

As to the impact on jobs, the illegal export of e-waste from the EU in 2012 is estimated to represent a **potential loss** of about **38,000 FTE recycling jobs in the EU**. Assuming a typical multiplier of 2, these direct recycling jobs would result in another **38,000 indirect and induced jobs**, for a total of **76,000 jobs**. The **illegal export to China** in particular is estimated to represent a potential loss of circa **14,900 FTE jobs** and another **14,900 indirect and induced jobs**, for a total of **29,800 jobs.** A loss of 14,900 FTE jobs goes along with an estimated loss of economic value added of around €780 million. Though this figure needs to be treated with caution due to data availability and quality issues, it is indicative of the significance of losses in economic terms.

Major gaps and research needs

The quantitative and economic assessments in this report rely on a number of assumptions, while facing considerable data availability and quality issues. The analysis combines different sources, which themselves carry uncertainties. Consequently, a number of analyses that would be useful could not be conducted, such as an estimate of the economic impacts on human health and the health sector in regions with considerable activities in the informal e-waste sector. Likewise, the economic implications of contaminated water and soils could not be depicted entirely. For example, it would be desirable to assess the local and regional clean-up costs with regard to soil.

1 Introduction

1.1 Key aspects of illegal e-waste shipment from the EU to China

In recent years the cross-border transport of waste electrical and electronic equipment (WEEE) has significantly increased which also led to the rise of illegal shipments of e-waste from developed countries to the developing world. At the international level it is the Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and their Disposal which seeks to provide enhanced control over the transboundary movements of hazardous wastes, including e-waste. The EU, together with its Member States, is a party to the Basel Convention since 1994 and the convention is translated into EU law via the EU Waste Shipment Regulation (WSR)[1]. Compared to the Basel Convention the EU WSR poses stricter control on the export of e-waste as it bans the export of hazardous waste from the EU to non-OECD countries in the developing world intended not only for disposal but also recycling/recovery.[2] Despite these stringent regulations vast amount of e-waste still leave the EU illegally.

Box 1: Definition of waste electrical and electronic equipment in EU legislation

According to the EU Waste Electrical and Electronic (WEEE) Directive e-waste "*means electrical or electronic equipment which is waste within the meaning of Article 3(1) of Directive 2008/91/EC, including all components, sub-assemblies and consumables which are part of the product at the time of discarding*".

The Waste Framework Directive (2008/91/EC) defines waste as "*any substance or object which the holder discards or intends or is required to discard".*

Source: Directive 2008/91/EC and Directive 2012/19/EU

Although the import of WEEE into China has been officially banned since 2000[3] China represents the largest downstream destination of EU e-waste export and according to the United Nations Office on Drugs and Crime approximately 80 per cent of the total global amount of e-waste goes to Asia, out of which 90 per cent ends up in China (UNODC, 2013). Most of the illegally imported e-waste enters China's informal refurbishment and recycling sector, which lacks environmental, health and safety standards. The main informal refurbishing and recycling sites are located around key waterways and ports of entry and include towns like Guiyu, Taizhou and Longtang (see Figure 1).

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[1] Regulation (EC) No 1013/2006 of the European Parliament and of the Council of 14 June 2006 on shipments of waste. OJ L 190, 12.7.2006

[2] The Basel Convention only forbids the export of WEEE for disposal from OECD countries to non-OECD countries.

[3] The ban was introduced by the Notification of the Import of the Seventh Category of Solid Waste No. 19/2000.

Figure 1: Major informal e-waste recycling and dismantling sites in China[4]

Source: Ni and Zeng, 2009, p. 3992

The illegal EU e-waste shipments into China affect the environment, human health and economies at multiple levels and carry significant risks on the ground (see Figure 2). This study aims to provide an insight into the negative economic impacts in the EU and the negative environmental and human health impacts in China and a quantitative and monetary analysis of these impacts.[5] It is to be said that while there is a growing body of literature available on illegal e-waste shipments, empirical data suffers from high uncertainties. There are significant data reliability problems with the estimates on the e-waste volumes exported from the EU into China; the chapter in this study assessing the extent of the illegal activities therefore should be treated with caution. Furthermore, the quantification and monetisation of environmental, human health and economic impacts presented in this study should be seen as rough estimates which often use assumptions and extrapolations.

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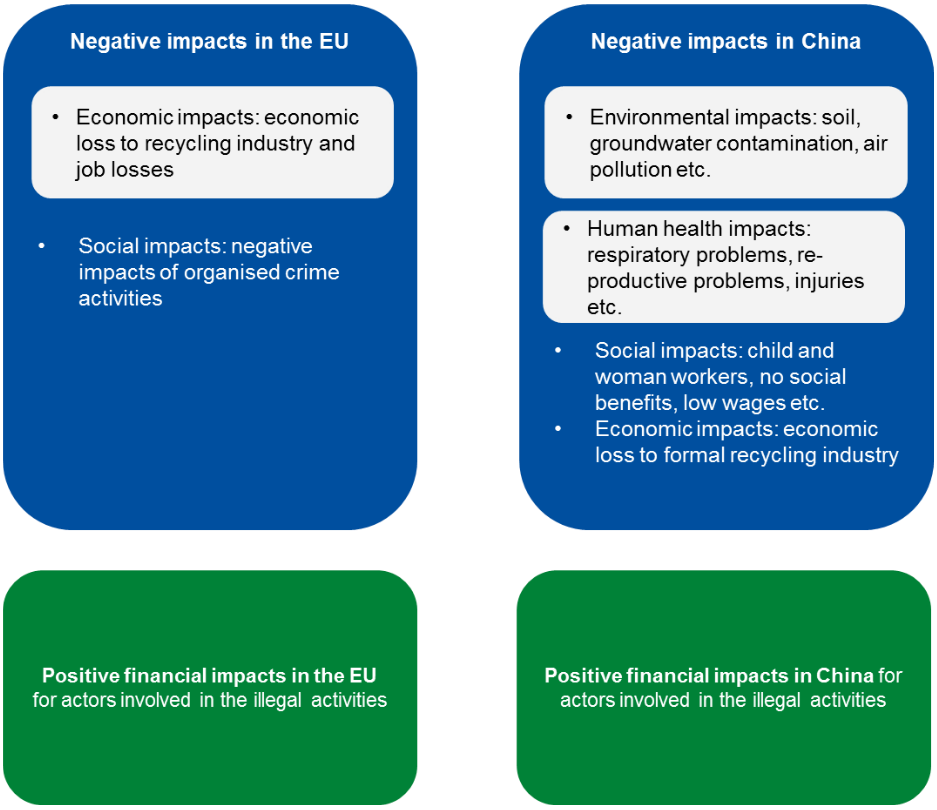
[4] Beijing is only indicated for the reader's geographic reference.

[5] As part of the EFFACE project other key aspects of illegal e-waste export from the EU to China, including the effectiveness of legislation governing the illegal activities, key stakeholders involved in the exports and the links to organised crime, were analysed in another case study. For more information see: Geeraerts et al. 2015.

Figure 2: Overview of key impacts of illegal e-waste shipment from the EU to China



Source: own representation. Note: key impacts analysed in this study are highlighted in bold.

1.2 Structure of the report

The remainder of the report is structured as follows:

- Chapter 2 provides an overview of the literature available on the volume of e-waste illegally exported from the EU to China and the environmental, health and economic impacts of these illegal shipments.
- Chapter 3 describes the methodological approach that has been applied in this study.
- Chapter 4 presents the quantitative analysis of the volumes of e-waste illegally shipped from the EU to China and the environmental, health and economic impacts of such illegal activities. This chapter also presents a brief case study on the impacts of elevated lead levels in children on their IQ scores.
- Chapter 5 presents the monetary analysis of environmental, health and economic impacts of illegal e-waste export from the EU to China.
- Chapter 6 outlines the main conclusions of the study.
- Annex 1 provides details of the Chinese soil quality standards.
- Annex 2 provides population statistics of Guiyu.

2 Background: summary of available literature

2.1 Illegal exports from the EU and imports in China

A number of publications and websites were found to present estimates of the amount of waste electrical and electronic equipment (WEEE) generated globally (e.g. Breivik et al. 2014 and StEP 2015) and generated in and exported from the EU (e.g. StEP 2015, Zoeteman et al. 2009 and Huisman et al. 2008) and from specific European countries to developing countries (StEP 2015). Literature is available on major sourcing countries of WEEE, such as the Netherlands (e.g. Huisman et al. 2012), Belgium (e.g. Seum and Hermann 2010), Germany (e.g. Sander and Schilling 2010) and the UK (e.g. Interpol 2009). Several publications were found to present rough estimates of the amount of WEEE imported illegally into China (e.g. UNODC 2013; Wang et al. 2013; Early 2013). The volume figures identified in relevant literature are seen as rough estimates which often use assumptions and extrapolations. An obvious reason for inaccuracy lies in the nature of any types of illegal activity but for instance the lack of differentiation between new and used EEE exported from the EU in statistical databases also makes data inaccurate.

2.2 Environmental and health impacts

Quite a number of studies have been published on the environmental and health impacts of e-waste recycling in China. Many studies investigate environmental and/or health impacts in Guiyu (e.g. Labunska et al. 2013; Leung et al. 2006; Leung et al. 2008; Bi et al. 2010; Deng et al. 2006; Zhu et al. 2012; Chan et al. 2013) and Taizhou (e.g. Han et al. 2010; Bai et al. 2011; Tang et al. 2010a, 2010b; Fu et al. 2008; Leun et al. 2010). Other places in China where recycling activities take place have been studied as well, such as Jiangsu (e.g. Xue et al. 2012), Shanghai (e.g. Fang et al. 2013), Longtang (e.g. Luo et al. 2011), Shijiao (Luo et al. 2008), Fengjiang (e.g. Wen et al. 2011) and Qingyuan (e.g. Zheng et al. 2012; Luo et al. 2009).

Many studies focus on **environmental** effects arising from one particular type of pollutant. Heavy metals have received a lot of attention from researchers, to give one example (Wu et al; 2014; Song & Li 2014a; Bi et al. 2010; Deng et al. 2006). Song & Li (2014a) for instance have reviewed studies in relation to environmental effects of heavy metals from e-waste recycling activities in China. Other studies focusing on metals have been carried out by among others Alabi et al. (2012), Zhao et al. (2010, 2011), Zhang and Hang (2009), Fu et al. (2008) and Bai et al. (2011). Several environmental and health studies on PBDEs have been found (e.g. Labunska et al. 2013). Studies focusing on polychlorinated biphenyls (PCBs) only are also quite common (e.g. Wang et al. 2011; Han et al. 2010). Some studies focus on the broader group of persistent organic pollutants (POPs), which include along others polybrominated diphenyl ethers (PBDEs), PCBs and polychlorinated dibenzo dioxins/furans (PCDD/Fs) (Bao et al. 2012).

Some studies on environmental impacts focus on concentrations of pollutants in several environmental media simultaneously (Labunska et al. 2013; Song & Li 2014a). Most studies however focus on one or two media only such as water and/or sediment (Bao et al. 2012; Leung et al. 2006), soil and/or vegetation (Alabi et al. 2012; Wang et al. 2011; Wu et al. 2014; Zhao et al. 2010, 2011; Fu et al. 2008) or air (Bi et al. 2010; Deng et al. 2006; Han et al. 2010; Fang et al. 2013) thereby usually studying several types of pollutants. Quite a number of studies in China deal with heavy metal contamination of (food) plants, mostly focusing on rice (Zhao et al. 2010-2011; Fu et al. 2008), though other plants such as vegetables and wild plants have also been studied (Luo et al. 2011).

As to **health** impacts, studies can be divided into four categories: studies focusing on certain routes of exposure (e.g. dietary intake), studies focusing on the human body burden (e.g. blood lead levels), studies focusing on the health effects (e.g. increased incidence of respiratory diseases); and studies bringing together these different health aspects of (informal) e-waste recycling and dumping (Song & Li 2014b).

As to the routes of exposure, most studies focus on dietary intake (e.g. Chan et al. 2013; Qin et al. 2011; Labunska et al. 2013; Leun et al. 2010) while some studies focus on inhalation from the air (e.g. Xing et al.

2009; Wen et al. 2011) and soil/dust ingestion and dermal exposure (Ma et al. 2008). The most commonly studied pollutants here are PBDEs. Other pollutants included in the studies are PBBs (e.g. Zheng et al. 2012), PCBs (e.g. Xing et al. 2009), PCDD/Fs (e.g. Song et al. 2011), PAHs (e.g. Wang et al. 2012b), heavy metals (Fu et al. 2008) and dioxin-like chemicals (Frazzoli et al. 2010).

As to studies focusing on body burdens of e-waste exposure, many have clearly indicated a causal relationship between emissions from informal e-waste recycling and dumping activities and the level of pollutants in human bodies (Song & Li 2014b). These studies commonly include the following types of human tissues: placenta (e.g. Leung et al. 2010; Zhang et al. 2011; Guo et al. 2010), umbilical cord blood (e.g. Wu et al. 2011), breast milk (e.g. Leung et al. 2010), blood and serum (e.g. Yang et al. 2013; Zhang et al. 2012; Wang et al. 2011; Huo et al. 2007), urine (e.g. Wang et al. 2011b), hair (e.g. Wang et al. 2009c) and other tissues (Zhao et al. 2009).

Many studies have been conducted on the health consequences of exposure to e-waste in China. The majority of select studies reviewed by Grant et al. (2013) showed associations between exposure to e-waste and physical health outcomes, including lung function (Zheng et al. 2013), physical growth (Zheng et al. 2013; Huo et al. 2007), reproductive health (Guo et al. 2010, 2012; Wu et al. 2011, 2012; Xu et al. 2012), thyroid function (e.g. Zhang et al. 2010) and changes in cellular expression and function (Yuan et al. 2008; Liu et al. 2009; Chen et al. 2010; Wang et al. 2011a). Negative associations were also shown for blood lead levels on the one hand and mental health outcomes (Liu et al. 2011; Xu et al. 2012; Li et al. 2008a) and IQ scores in children on the other hand (Wang et al. 2012a).

In addition to articles reporting results from one study, review articles have been identified: on environmental effects of heavy metals from e-waste recycling (Song & Li 2014a), on the human body exposure of e-waste (Song & Li 2014b) and on the health consequences of e-waste exposure (Grant et al. 2013).

2.3 Economic impacts

Only a limited number of reports exist in which economic impacts have been assessed for the waste management/recycling sector in source countries. An Amec study from 2012 for instance has estimated the overall economic impact of illegal waste exports from the UK, including the loss in profits to WEEE treatment facilities in the UK (Amec 2012).

Economic/financial reports on status of the EU or American (electronic) waste markets are mostly very pricey and thus not readily available for the research team. A 2013 study (or at least a power point presentation) by Sullivan & Frost on the EU e-waste recycling market was freely available though (Sullivan & Frost 2013). A document from the WEEE Forum also includes several economic figures on the e-waste recycling sector, including the average intrinsic value of e-waste (WEEE Forum 2013). A UNODC study from 2013 also provides figures on the economic value of e-waste. The website of the European Recycling Platform provides figures on the value of certain precious metals contained in waste and used electronic equipment (ERC 2014).

As to jobs, a 2013 study for the Coalition for American Electronics Recycling (CAER) has calculated the number of jobs that could be created by diverting e-waste from landfill or illegal export (DSM 2013). A 2004 study by the London South Bank University (LEPU 2004) did the same for different types of waste within the context of London. A Friends of the Earth Report assessed the potential for job creation through higher rates of recycling in the UK and EU (FoE 2010). Several other reports assess the (actual or potential) number of jobs dependent on environment and resource efficiency improvements in general (e.g. Rademaekers et al. 2012).

3 Research Methodology

3.1 Environmental impacts

The environmental impacts have among others been **quantified** through calculating the potential amounts of pollutants released in the Chinese environment originating from the illegal imports from the EU (building further on the quantification of the amounts of e-waste being shipped from the EU as a whole and from the EU to China in particular). The annual emission estimates of some contaminants are based on estimated concentrations in components of Swiss e-waste (figures from Morf et al. 2007) and on estimates of the 2005 and 2012 illegal imports in China from the EU. These concentrations are multiplied with the estimated amounts of e-waste that are being shipped illegally from the EU to China.

In addition, concentrations of pollutants (mostly heavy metals) in different environmental media (air, dust, sediment, soil, plants) in traditional e-waste recycling towns such as Guiyu and Taizhou have been presented to sketch a quantitative picture of the pollution resulting from the informal recycling of e-waste. It was however not possible to determine the exact share of pollution stemming from European e-waste at particular sites, although figures are presented which might give a rough indication of the share of EU-borne e-waste processed in China. In this respect it should be noted that these Chinese recycling towns also recycle or have recycled huge amounts of e-waste imported from other developed countries (US in particular) and are probably recycling more and more e-waste generated domestically as living standards and domestic consumption in China have been rising substantially in the last 10 years.

The environmental impacts have been **monetised** by estimating the (actual) direct costs for constructing water supply infrastructure (drinking water pipelines) in Guiyu.

3.2 Health impacts

As to the **quantification** of health impacts on e-waste workers and inhabitants of e-waste recycling towns (and inhabitants of neighbouring areas) figures on the routes of exposure, on the human body burden and on scientifically established associations and/or causal links between e-waste exposure and several kinds of diseases and other health impacts are presented. For purposes of illustration and to keep track it was decided to present select figures, mostly in relation to heavy metal pollution in two traditional informal e-waste recycling towns, i.e. Guiyu and Taizhou. A more detailed quantification has been carried out for the impacts of lead poisoning on children's neurological development. Similar quantifications can be done for other pollutants, for other health impacts and for other groups of people (children vs. adults, e-waste workers vs. inhabitants, inhabitants of e-waste recycling towns vs. inhabitants of adjacent regions) but these are not presented in this publication. Quantification efforts in this respect might bump into several obstacles such as a lack of data and/or limited access to data, especially data on the number of people effectively affected in China by e-waste exposure and the precise contribution of EU-borne e-waste to the overall e-waste exposure and to the overall exposure to pollutants.

As to the impacts of lead poisoning on children's neurological development, the number of children exposed to critical blood lead exposure in Guiyu have been estimated and subsequently extrapolated to China as a whole (i.e. to the number of children living in e-waste recycling areas). For the extrapolation estimates from Breivik et al. (2014) on the number of e-waste recycling workers in Guiyu vs. China as a whole and on the tonnes of e-waste being informally processed in Guiyi vs. China have been used.

For determining the number of children affected by IQ losses the 95th percentile lower confidence limit of the benchmark dose (BMD) of 1 per cent extra risk ($BMDL_{01}$) determined by the European Food Safety Authority (EFSA) Panel on Contaminants in the Food Chain at 12 μg/L (B-Pb) has been used (see Box 2). This level has been set by the EFSA Panel as a as a reference point for the risk characterisation of lead when assessing the risk of intellectual deficits in children measured by the Full Scale IQ score.

Subsequently the total amount of IQ loss has been quantified based on the assumption that for each 10 μg/dl of blood lead, IQ is reduced by 2 points.

As to the **monetisation** of the health impacts, the following could be assessed: opportunity costs in terms of *lost productivity* (i.e. decreased current value of expected lifetime revenues); *direct resource educational costs* related with compensatory education; opportunity costs of *lost income during remedial compensatory education*; *medical treatment costs*; and, *disutility* resulting from human development disabilities. In order to estimate the medical treatment costs in China for instance, one could start working from a range of values that correspond to the lowest and highest values given in existing studies – we have identified 2 studies from the US. These figures need to be adapted to the situation in China if one wants to use these for estimating the medical treatment costs. This also requires an analysis into the medical treatment of certain diseases in China and/or in Chinese e-waste recycling areas in particular: to what extent get lead-poisoned children a medical treatment and what type of treatment do they get.

Box 2: The benchmark dose approach

In general, a dose-response relationship describes the relationship of the likelihood and severity of adverse health effects (the responses) and the amount and condition of exposure to an agent (the dose provided). The benchmark dose (BDM) approach is applicable to all toxicological effects. The BDM by definition is a dose level which is associated with a specified change in response, the Benchmark Response (BMR). The BMDL is the BMD's lower confidence bound and it is usually used as a reference point. For instance, for a BMR of 5 per cent the BMDL, the benchmark dose lower confidence limit, can be interpreted as $BMDL_{05}$ and it means a dose where the response is likely to be smaller than 5 per cent. The term 'likely' is defined by the statistical confidence level, usually 95 per cent.

Source: EFSA 2009, p. 7-12

3.3 Financial/economic impacts

The economic impacts for Europe resulting from the illegal export of e-waste are twofold: the economic loss to the EU e-waste recycling sector (loss in profits and loss of economic value) and the loss of jobs in this sector. Our methodology for **estimating potential loss of jobs** in the EU e-waste recycling sector is based on best available published data for the EU27, the UK and the US and applying lessons learnt from more detailed studies on jobs per 1,000 tonnes of material recycled, and multipliers for indirect and induced jobs (DSM 2013; LEPU 2004; CASCADIA 2009). We have made some simplifying assumptions in arriving at conservative estimates of potential job loss in the recycling sector.

Employment loss from illegal e-waste exports outside of the EU may be calculated by following an approach whereby the lost total turnover for the EU e-waste recycling sector is divided by the sector's labour intensity (i.e. the level of turnover per full-time equivalent (FTE) job). As no data were retrieved on the sector's labour intensity, another approach has been followed, i.e. an approach using the lowest estimates for coefficients for FTE jobs for diverting 1,000 tonnes of e-waste from landfill, incineration or illegal export. It should be noted that some of these jobs could be overseas if recycling rates are achieved by exporting recycled material. Subsequently, indirect and induced (loss of) employment has been estimated by applying a typical multiplier of 2 taken by 2013 study commissioned by the Coalition for American Electronics Recycling. Alternatively, multipliers could be applied from a 2010 study by Friends of the Earth UK (FoE 2010). In this study indirect and induced (loss of) employment has been estimated by applying separate multipliers for indirect and induced jobs: a multiplier of 1.50 to calculate the indirect jobs resulting from direct employment in the recycling sector and a multiplier of 1.75 to calculate the induced jobs. The authors the Friends of the Earth (FoE) study consider these multipliers as conservative, as these are lower than those generally applied in other economic sectors (FoE 2010).

The direct job loss for the EU e-waste recycling industry may be **monetised** by multiplying the number of jobs lost with the average economic value added per employee in the sector.

The **loss in profits** for the EU e-waste recycling sector arising from illegal e-waste exports may be monetised by extrapolating an estimate of the loss in profits to the UK industry (taken from an existing study) to the EU as a whole. The **economic value lost** to the EU may be monetised by building further on an estimate of the average intrinsic value of WEEE per tonne (and multiplying this average value with the estimated volumes of WEEE exported out of the EU). In addition or alternatively, the economic value lost may be monetised (fully or partially) using a bottom-up approach: the monetary value of precious metals within WEEE exported from the EU may be estimated by building further on existing estimates of the value of precious metals contained in one kilogram of WEEE.

4 Quantitative analysis of impacts

4.1 Volumes of e-waste shipped from the EU to China

Quantifying the illegal export of e-waste from the EU (to China) is especially challenging as there is very little clear information upon which estimates can be based. Nevertheless an attempt has been made to come up with estimates for 2005 and 2012.

As a first step the estimate by Zoeteman et al. (2009) of the amounts of e-waste imported from the EU to China in 2005 was taken. As a second step the e-waste imported from the EU to China in 2012 was assessed thereby applying Zoeteman's methodology to 2012 figures.

Generation of WEEE in the EU

According to Zoeteman et al. (2009) 0.62 million tonnes of e-waste were imported from the EU to China in 2005. This estimate was calculated as follows. Firstly, the generation of WEEE in the EU in 2005 was calculated by multiplying the number of EU inhabitants by the average WEEE generated:

*457,000,000 inhabitants x 0.015 tonnes (15 kg/inhabitant/year) = **7,005,000 tonnes in 2005***

Zoeteman et al. (2009) estimated that:

- 50 per cent of this volume of e-waste was large household appliances (category 1 - fridges and washing machines): 7.5 kg/inhabitant/year;
- 10 per cent small household appliances (category 2 - vacuum cleaners, toasters) : 1.5 kg/inhabitant/year;
- 20 per cent office and communication waste (category 3 - computers, cell phones): 3.0 kg/inhabitant/year; and
- 20 per cent entertainment electronics (category 4 - categoryradios, TVs, stereos): 3.0 kg/inhabitant/year.

According to the StEP project **9,918,000 tonnes** of e-waste have been generated in the EU **in 2012** (StEP 2015c).

Export from the EU

Zoeteman et al. (2009) assumed that **10-20 per cent** of the total amount of **WEEE** generated in the EU is *illegally* exported (between 700,500 and 1,401,000 tonnes); furthermore **30 per cent** of **used computers, TVs and mobiles** are *legally* exported to developing countries for reuse (approximately 840,600 tonnes). It is important to note that part of this UEEE either becomes WEEE during the transport (e.g. if there is no appropriate protection of the product during the transport) or a short period of time after arriving in the destination country (BIO IS 2013) and therefore are treated as illegally exported WEEE in the below calculation. In this context it is also interesting to note the typical lifespan of different EEE items (see Table 1).

Table 1: Typical lifespan of various EEE items

Type of EEE	Typical lifespan (years)
Computer	3
Mobile telephone	2

Radio	10
Television	5
Video recorder and DVD player	5
Dish washer	10
Electric cooker	10
Freezer	1
Kettle	3
Microwave	7
Toaster	5
Washing machine	8

Source: Robinson 2009

By adding up these figures Zoeteman et al. (2009) concluded that **1.9 million tonnes** of WEEE and used EEE left the EU in **2005** (see also Table 2 and Table 3).

0.15 x 7,005,000 (free riders part) + 0.3 x (0.2 + 0.2) x 7,005,000 (tonnes categories 3 + 4 WEEE) = ***1.9 million tonnes in 2005***

According to a study for the European Commission (BIO IS 2013) around **15 per cent of used electrical and electronic equipment (UEEE)** is exported, mainly for re-use. As no detailed data are available for 2012 on the amounts within the four WEEE categories applied by Zoeteman et al. (2009), we apply this percentage of 15 to the UEEE figure as a whole instead of applying the 30 per cent to only two out of the four categories.

0.15 x 9,918,000 (free riders part) + 0.15 x 9,918,000 (used EEE legally exported for reuse) = ***2,975,400 tonnes*** *or ca.* ***2.98 million tonnes in 2012***

Illegal imports of WEEE in China

Zoeteman et al. (2009) assumed that 20 per cent of EU export ends up in West-Africa and 20 per cent in Eastern Europe and North Africa. The remainder 60 per cent was assumed to go to Asia of which 65 per cent was assumed to go to China and 35 per cent to India, thereby neglecting export to smaller other Asian Countries. The import in China of e-waste from the EU was calculated as follows:

0.6 x 0.65 x 1,900,000 tonnes = 741,000 tonnes ≈ ***0.74 million tonnes (in 2005)***

A similar calculation was done for 2012 thereby assuming that only the total amount of e-waste exported from the EU has changed and that the relative distribution of these exports over the various destination countries has not changed: in other words it was assumed that in 2012 ca. 40 per cent of EU exports still end up in China.

0.6 x 0.65 x 2,975,400 tonnes = 1,160,406 tonnes ≈ ***1.16 million tonnes (in 2012)***

Table 2: Total export of e-waste from the EU and import to China from EU, estimations for 2005 for overall WEEE and per WEEE category

WEEE category	Total export from EU in 2005 (in million tonnes)	China import from EU in 2005 (in million tonnes)
Total	**1.9**	**0.74**
Category 1: large household appliances	0.97	0.39
Category 2: small household appliances	0.19	0.07
Category 3: office and communication equipment	0.37	0.14
Category 4: entertainment electronics	0.37	0.14

Source: adapted from Zoeteman et al. (2009). Note: large household appliances include fridges and washing machines; small household appliances include vacuum cleaners and toasters; office and communication equipment include computers and cell phones; entertainment electronics include radios, TVs and stereos. As indicated above Zoteman et al. (2009) estimated that of the total amount of generated WEEE in the EU in 2005 50 per cent are large household appliances, 10 per cent are small household appliances, 20 per cent are office and communication equipment and 20 per cent were entertainment electronics. For the export and import figures presented in this table it was assumed that the share of the categories is the same as for the generated WEEE.

Table 3: Estimates of e-waste generated in the EU, e-waste exported from the EU and e-waste imported in China in 2005 and 2012 (in million tonnes)

Year	E-waste generated in the EU	E-waste exported from the EU	E-waste imported in China	E-waste imported in China from EU
2005	7,005,000	1,900,000	2,030,000[a]	**741,000**
2012	9,918,000	2,975,400	8,000,000[b]	**1,160,406**

Source: own representation. Note: [a] estimate found in Wang et al. (2013); [b] estimate from UNODC (2013)

In order to incorporate the uncertainty about this percentage of EU exports ending up in China[6], we might want to work with a 'minimum China import scenario' and a 'maximum China import scenario', alongside this 'default scenario'. This approach might at the same time incorporate the uncertainty surrounding the percentage of EU generated e-waste that is 'guesstimated' to be illegally exported outside of the EU, by working with a 'minimum EU export scenario' and a 'maximum EU export scenario'. If we combine these two sets of minimum and maximum scenarios, we might come up with a 'minimum export/import scenario' and a 'maximum export/import scenario'.

For the below presented minimum and maximum scenarios the following estimates were used:

- According to Zoetman et al. (2009) 10-20 per cent of the total amount of WEEE generated in the EU is illegally exported.
- According to a study for the European Commission (BIO IS 2013) around 15 per cent of used electrical and electronic equipment (UEEE) is exported, mainly for re-use.

[6] Given control measures being implemented in China, future flows of e-waste management may be diverted to less affluent countries or jurisdictions where costs related to environmental regulation are minimised (Lepawsky & Billah 2011; Lepawsky & McNabb 2010). This is however extremely difficult to monitor and quantify.

- Finally, Zoetman et al. (2009) estimated that 60 per cent of EU export WEEE ends up in Asia of which 65 per cent was assumed to go to China, i.e. approximately 40 per cent of EU WEEE export is shipped to China.

Based on the above estimates, it is suggested to use following percentages for the '**minimum export/import scenario**':

- 10 per cent of EU WEEE is exported illegally out of the EU,
- 10 per cent of the EU WEEE is exported legally as UEEE, and
- 35 per cent of these EU exports are imported in China.

Table 4: The minimum export/import scenario in 2005 and 2012 (in tonnes)

	2005	2012
E-waste illegally exported from the EU	7,005,000 x 0.1 = 700,500	9,918,000 x 0.1 = 991,800
E-waste legally exported from the EU as UEEE	7,005,000 x 0.1 = 700,500	9,918,000 x 0.1 = 991,800
Total amount of e-waste exported from the EU	700,500 + 700,500 = **1,401,000**	991,800 + 991,800 = **1,983,600**
E-waste exported from the EU imported into China	1,401,000 x 0.35 = **490,350**	1,983,600 x 0.35 = **694,260**

Source: own calculation. Note: According to Zoetman et al. (2009) 7,005,000 tonnes of WEEE were generated in the EU in 2005, while StEP project estimated that 9,918,000 tonnes of e-waste have been generated in the EU in 2012 (StEP 2015c).

The '**maximum export/import scenario**' would be as follows:

- 20 per cent of EU WEEE is exported illegally out of the EU,
- 20 per cent of the EU WEEE is exported legally as UEEE, and
- 45 per cent of these EU exports are imported in China.

Table 5: The maximum export/import scenario in 2005 and 2012 (in tonnes)

	2005	2012
E-waste illegally exported from the EU	7,005,000 x 0.2 = 1,401,000	9,918,000 x 0.2 = 1,983,600
E-waste legally exported from the EU as UEEE	7,005,000 x 0.2 = 1,401,000	9,918,000 x 0.2 = 1,983,600
Total amount of e-waste exported from the EU	1,401,000 + 1,401,000 = **2,802,000**	1,983,600 + 1,983,600 = **3,967,200**
E-waste exported from the EU imported into China	2,802,000 x 0.45 = **1,260,900**	3,967,200 x 0.45 = **1,785,240**

Source: own calculation. Note: According to Zoetman et al. (2009) 7,005,000 tonnes of WEEE were generated in the EU in 2005, while StEP project estimated that 9,918,000 tonnes of e-waste have been generated in the EU in 2012 (StEP 2015c).

4.2 Environmental impacts

As illegally imported e-waste by its nature is mostly processed in the informal Chinese recycling sector, the environmental impacts of e-waste shipments from the EU mostly relate to informal recycling activities in China (as opposed to formal recycling activities).

The intensive informal and thus uncontrolled recycling of e-waste in China has resulted in the release of large amounts of contaminants in the local environment, such as **heavy metals**, polybrominated diphenil ethers (**PBDEs**), polychlorinated dibenzo dioxins/furans (**PCDD/Fs**), polychlorinated biphenyls (**PBCs**), chlorofluorocarbons (**CFCs**) and polycyclic aromatic hydrocarbons (**PAHs**).
Table 6 lists the potential environmental contaminants associated with e-waste imported illegally in China from the EU. The annual emission estimates of some contaminants are based on concentrations in components of Swiss e-waste (Morf et al. 2007) and on the estimates of the 2005 and 2012 illegal imports in China from the EU (see previous chapter). Although recycling may remove some contaminants, large amounts still end up concentrated in e-waste recycling and dumping sites. To give one example: according to our rough estimates/calculations respectively **10 and 16 tonnes of PCBs** were (potentially) **released in the Chinese environment** in **2005 and 2012** as a result of the informal recycling and dumping of e-waste imported from the EU (see
Table 6). Other figures from the table are for instance the potential release of **1,258 and 1,972 tonnes of antimony** and of ca. **30,000 and 35,000 tonnes of copper** in 2005 and 2012 respectively.

In reality the amounts of contaminants released into the environment in China as a result of informal (and formal) e-waste recycling and dumping is multiple times higher as the figures above only relate to the imports from the EU. Whereas we estimated that in 2012 1.16 million were imported illegally from the EU experts estimate that in total 8 million tonnes of e-waste are imported in China each year (UNODC 2013). E-waste-related environmental problems however also result from China's rising domestic generation of e-waste. For 2012 it was estimated that 7,253,010 tonnes of e-waste were generated in China (StEP 2015b; see also Figure 3 in paragraph 4.4.2).

Field and laboratory research in China has shown that e-waste is significantly **degrading air, soil, and water quality**, as well as a **range of biota** (Walters & Santillo 2008). In Guiyu and other informal e-waste recycling centres concentrations of multiple pollutants including **heavy metals, PAHs, PBDEs, PCDD/Fs and PCBs** have been found to exceed various international standards and local norms, in some cases by several scale factors (Deng et al. 2007; Deng et al. 2006; Luo et al. 2007; Li et al. 2007; Leung et al. 2006; Wong et al. 2007). There also exists a more nuanced environmental impact from WEEE trade relating to global concerns about resource scarcity and sustainability. Whilst WEEE trade aims at recycling the valuable materials, it is an evidently **wasteful** and **energy intensive industry** and as such operates counter to global and EU ambitions for resource efficiency. There is also an **indirect impact on climate change** given the carbon insensitivity of the primary production of the substances of EEE. Illegal e-waste shipments also have adverse consequences for the ozone layer as obsolete refrigerators, freezers and air conditioning units contain ozone depleting substances such as CFCs which may escape from inadequately recycled or dumped items (Robinson 2009).

Informal recycling activities have caused in particular high concentrations of heavy metals (such as lead, cadmium, mercury, copper, zinc, etc.) in the surrounding air, dust, soils, sediments and plants. In the next paragraphs we focus - for the sake of maintaining the overview – on the environmental impacts related to heavy metals and lead and zinc in particular. We hereby refer mostly to studies investigating adverse environmental impacts in Guiyi and Taizhou, as the most representative locations for informal e-waste recycling. Studies on negative environmental impacts of e-waste recycling in other places in China (such as Jiangsu, Shanghai, Longtang, Shijiao and Fengjiang) have been identified but are not discussed within the framework of this study. It should be noted that Taizhou has gradually phased out the informal recycling of e-waste as local manufacturing has shifted away from the production of electronics in recent years. As a result of this shift in local manufacturing and stricter regulation of polluting activities related to e-waste recycling (e.g. leaching of circuit boards), very little informal e-waste recycling has been identified in Taizhou in recent years (Wang et al. 2013). However, if informal recycling activities decrease or disappear in one place, these informal activities are usually shifted to other places (in or outside China). But also in Guiyu the public authorities are increasingly trying to regulate the informal recycling sector and to manage the environmental impacts from e-waste recycling and dumping, among others by remediating contaminated soil.

Table 6 provides a non-exhaustive list of potential environmental pollutants in China from informal e-waste recycling or dumping with the contaminants' typical e-waste concentration and emissions from e-waste imported from the EU in 2005 and 2012. Other important contaminants and their relationship with e-waste are presented in Table 7.

Table 6: Potential environmental pollutants in China from informal e-waste recycling or dumping

Contaminant	Relationship with e-waste	Typical e-waste concentration (mg/kg)	Emission in China from e-waste imported from EU in 2005 and 2012 (tonnes)[a]
Polychlorinated biphenyls (PCB)	Condensers, transformers	14	10.36 / 16.24
Antimony	Flame retardants, plastics	1,700	1258 / 1972
Cadmium (Cd)	Batteries, toners, plastics	180	133 / 209
Copper (Cu)	Wiring	41,000	30,340 /35,194
Lead (Pb)	Solder, CRTs, batteries	2,900	2,146 / 2,489
Mercury (Hg)	Fluorescent lamps, batteries, switches	0.68	0.50 / 0.79
Nickel (Ni)	Batteries	10,300	7,622 /11,948
Tin (Sn)	Solder, LCD screens	2,400	1,776 / 2,748
Zinc (Zn)		5,100	3,774 / 5,916

Source: adapted from Robinson 2009 and Morf et al. 2007. Note: [a] Assuming 0.74 million tonnes of illegal imports in China from the EU in 2005 and 1.16 million tonnes in 2012 (see 4.1).

Table 7: Important contaminants and their link to e-waste

Contaminant	Relationship with e-waste
Polybrominated diphenyl ethers (PBDEs) polybrominated biphenyls (PBBs) tetrabromobisphenol-A (TBBPA)	Flame retardants
Chlorofluorocarbon (CFC)	Cooling units, insulation foam
Polycyclic aromatic hydrocarbons (PAHs)	Product of combustion

Polyhalogenated aromatic hydrocarbons (PHAHs)	Product of low-temperature combustion
Polychlronated dibenzo-p-dioxins (PCDDs), polychlorinated dibenzofurans (PCDFs)	Product of low-temperature combustion of PVCs and other plastics
Americium (Am)	Smoke detectors
Beryllium (Be) Silicon-	Silicon-controlled rectifiers
Arsenic (As)	Doping material for Si
Barium (Ba)	Getters in cathode ray tubes (CRTs)
Chromium (Cr)	Data tapes and floppy disks
Gallium (Ga)	Semiconductors
Indium (In)	LCD displays
Lithium (Li)	Batteries
Selenium (Se)	Rectifiers
Silver (Ag)	Wiring, switches
Rare earth elements	CRT screens

Source: adapted from Robinson 2009 and Morf et al. 2007.

4.2.1 Air

Several studies have indicated high levels of heavy metals in the indoor and outdoor atmosphere of Guiyu (Bi et al. 2010; Deng et al. 2006). For instance the level of lead in total suspended particles (TSP)[7] from a typical workshop for recycling of printed circuit boards in Guiyu (4.42 µg/m^3) exceeds 28.5 times the upper limit of average air lead levels for non-urban European sites (these levels are usually below 0.15 µg/m^3). It also exceeds several times urban air lead levels in most European cities which are typically between 0.15 and 0.5 µg/m^3 (Theakston 2001) (see

Table 8). This – together with other results from the study – demonstrates that the recycling of printed circuit boards is an important contributor to heavy metal contamination of the local environment (Bi et al; 2010).

Deng et al. (2006) also demonstrated high levels of lead outdoors (both in TSP and $PM_{2.5}$), though obviously lower than inside e-waste recycling workshops (0.444 µg/m^3). However, in these air samples chromium (1.161 µg/m^3) and zinc (1.038 µg/m^3) were the metals the most present in TSP, followed by copper and lead (see also

Table 8).

These high levels of heavy metals in the air generate serious environmental and biological problems (Eckelman & Graedel 2007).

[7] Total suspended particles refer to the mass concentration of particulate matter in community air.

Table 8: Environmental pollution of heavy metals in air

Pollutants	Sampling time	Sampling site	Location	Sample size	Concentrations in µg/m³	Reference values	References
Lead and zinc in particles	09/2007	Circuit boards recycling workshop	Guiyu	4 TSP	Pb: 4.42 Zn: 3.32	Urban air lead levels in EU cities: 0.15 – 0.5 µg/m³	Bi et al. 2010
Lead and zinc in TSP and $PM_{2.5}$	08-09/2004.	Roof of a 3-story building, approximately 10 m above ground level	Guiyu	29 TSP, 30 $PM_{2.5}$	TSP: Pb: 0.444; Zn: 1.038 PM2.5: Pb: 0.392; Zn: 0.924		Deng et al. 2006

Source: adapted from Song & Li 2014a

4.2.2 Dust

The composition of settled dust can be an indicator as well of air pollution such as the heavy metal contamination in the atmosphere. This is because the composition of settled dust is similar to suspended particles in the atmosphere (Song & Li 2014a).

In 2004 Leung et al. (2008) found high average concentrations of heavy metals in e-waste workshop dust (e.g. Pb 110,000 mg/kg and Zn 4,420 mg/kg) and in dust on adjacent roads (e.g. Pb 22,600 and Zn 2,370 mg/kg) (see also Table 9). The lead in road dust was 330 and 106 times higher than for non-e-waste sites located 30 km and 8 km away.

More recently Zhu et al. (2012) investigated heavy metals in dust from family-run workshops in Guiyu. For lead and zinc high levels in workshop dust were found: 892 and 1120 mg/kg respectively.

From the above, it can be concluded that the average lead concentrations in workshop dust were much higher in the past. This might be explained by the fact that more primitive recycling techniques were used in the past in Guiyu, compared with (more formal) e-waste recycling in recent years (Song & Li 2014a).

Table 9: Environmental pollution of heavy metals in dust

Pollutants	Sampling time	Sampling site	Location	Sample size	Concentrations in mg/kg	Reference values	References
Pb and Zn in dust	12/2004	Workshop dust; adjacent roads	Guiyu	-	Workshop dust: Pb: 110,000; Zn: 4,420 Adjacent roads: Pb: 22,600; Zn: 2,370		Leung et al. 2008
Sb and As in dust	2010	Family-run workshops from 13 e-waste recycling villages	Guiyu	34 indoor dust	Sb: 6.1–232 As: 5.4–17.7		Bi et al. 2011
Pb and Zn in dust	2010	Family-run workshops in e-waste recycling impacted area	Guiyu	29 indoor dust	Pb: 892; Zn: 1120		Zhu et al. 2012

Source: adapted from Song & Li 2014a

4.2.3 Soil

Studies on heavy metal contamination of soil in China focus mainly on three typical informal e-waste recycling sites: Guiyu, Taizhou and Longtang (Song & Li 2014a). We have selected the studies on Guiyu and

Taizhou (see Table 10). Within these studies three types of sampling sites can be identified: (1) e-waste recycling and disposal sites (workshops, dump sites, open burning sites); (2) the roadside; and (3) agricultural soil (paddy soil, sewage irrigation areas).

Table 10: Environmental pollution of heavy metals in soil

Pollutants	Sampling time	Sampling site	Location	Sample size	Concentrations in mg/kg	Reference values	References
Pb and Zn in soil	08/2003	Burnt plastic dump site, printer roller dump site, reservoir (control area)	Guiyu	-	Pb: 104; 190; 55 Zn: 258; - ; 78	35; 250 100; 200	Leung et al. 2006
Pb in soil	08/2009 and 10/2010.	E-waste dumpsite soil, Guiyu roadside soil	Guiyu	20-30	Pb: 1431; 540.9	35; 250	Alabi et al. 2012
Pb in soil	11/2005	Paddy soil	Taizhou	6	Pb: 55.81	35; 250	Fu et al. 2008
Pb and Zn in soil	12/2007	Paddy soil	Luqiao in Taizhou	32	Pb: 46.84 Zn: 209.85	35; 250 100; 200	Zhang and Hang 2009
Pb and Zn in soil	03/2008	Sewage irrigation area	Luqiao in Taizhou	54	Pb: 152.96 Zn: 289.99	35; 250 100; 200	Bai et al. 2011
Pb and Zn in soil	08-09/2008	Agricultural soils	Luqiao in Taizhou	20	Pb: 115.1 Zn: 163.4	35; 250 100; 200	Tang et al. 2010a
Pb and Zn in soil	10/2006	Paddy soil	Wenling in Taizhou	96	Pb: 48.3 Zn: 137.0	35; 250 100; 200	Zhao et al. 2010, 2011
Pb, Zn and Hg in soil	08/2008	Large-scale e-waste recycling plants; large-scale gold recovering plants; household e-waste recycling workshops	Wenling in Taizhou	39	Pb: 163.9; 143.6; 956.5 Zn: 300.3; 203.4; 392.4 Hg: 0.4; 1.8; 221.7	35; 250 100; 200 0.15; 0.3	Tang et al. 2010b

Source: adapted from Song & Li 2014a and Wang & Shan 2013; Note: the reference values are the 1995 Chinese Class-1 environmental quality standards for soils (soil background levels) and the Class-2 quality standards for soils with a pH level below 6.5. The Chinese soil quality standards are referred to as the GB15618-1995 standards. See Annex A for full overview of the Chinese soil quality standards. Furthermore, the different concentrations refer to the specific sampling sites listed in the table.

Three studies have examined the heavy metal effects of e-waste recycling processes on soil in Guiyu (two of which are referred to in Table 10). The most seriously polluted soils were found in e-waste dumpsites in Guiyu. The levels of lead and zinc in soil samples at these sites were at least two times higher than those in samples from the control sites (Leung et al. 2006). A more recent study revealed even higher levels of heavy metal contamination, including in roadside soil (Alabi et al. 2012), especially by lead and copper (Pb: 540.9 mg/kg; Cu: 683.8 mg/kg). It can be concluded that, overall, soil contamination with heavy metals decreases as the distance from the e-waste recycling sites increases (Song & Li 2014a).

Taizhou (located in Zhejiang province) has been the target of most of the studies on heavy metals in the agricultural soil. The most serious soil pollution was found in Luqiao, Taizhou (Bai et al. 2011; Tang et al.

2010a), often exceeding the maximum allowable concentrations as determined by the Chinese national environmental quality standard for soils (the GB 15618–1995 standards, see Annex A: Chinese soil quality standards for more information). Equally, the average concentrations of lead (115.1 mg/kg), zinc (163.4 mg/kg), cadmium and copper in agricultural soils in Luqiao were higher than the Class 1 values (Tang et al. 2010b).

As Taizhou is located in Zhejiang province, the most important producer of rice in China, it was also subject to studies on heavy metal levels in paddy soil. Zhang and Hang (2009) found that paddy soils from Luqiao were heavily contaminated with cadmium (6.4 mg/kg) and weakly contaminated with copper (256.4 mg/kg) and zinc (209.9 mg/kg).

In addition, a study by Fu et al. (2008) indicated that the paddy soil in one village in Taizhou was primarily contaminated by cadmium (1.19 mg/kg), followed by copper (98.8 mg/kg). Equally, the paddy soils in Wenling (Taizhou) were also contaminated with heavy metals including lead and zinc and for some areas potential high-risk levels of cadmium, copper, nickel and zinc were indicated (Zhao et al. 2010, 2011). A study by Tang et al. (2010b) found extremely high levels of contamination by mercury, but also by lead and zinc. The soil samples of the family e-waste recycling workshops in particular showed contamination levels significantly higher than the respective Chinese soil quality standards (see Table 10).

Box 3: The 1995 Chinese environmental quality standards for soil

According to Wang & Shan (2013) these standards urgently need to be revised. There is lack of consideration of human exposure risk. There is a need for covering more contaminants and various land uses. The standards currently focus on farm land, vegetable and tea producing fields, orchards, soil, pastures and natural reserved areas.

More detailed information on the GB15618-1995 soil quality standards are presented in Annex A.

Source: Wang & Shan 2013

4.2.4 Sediments

Sediments can absorb and accumulate pollutants and act as a contamination source even long after the pollution has occurred or abated. Polluted sediments impose negative impacts on aquatic organisms and human beings through the food web (Song & Li 2014a).

Table 11: Environmental pollution of heavy metals in sediment

Pollutants	Sampling time	Sampling site	Location	Sample size	Concentrations in mg/kg	Reference values	References
Pb and Zn	08/2003	Duck pond A, duck pond B, Lianjiang River 1, River 2, River 3, reservoir	Guiyu	-	Pb: 57.7; 53.1; 316; 94.3; 118; 39.4 Zn: 79.6; 84.5; 86.8; 249; 175; 45.2	-	Leung et al. 2006
Pb, Zn and Cu	04/2006	Lianjiang River and Nanyang River	Guiyu	-	Pb: 54.97; 394.5 Zn: 133.72; 482.75 Cu: 66.7; 2153.88	-	Wang et al. 2009a-b
Pb and Cu	06/2007	Nanguan River	Taizhou	-	Pb: 377.33 Cu: 4787.5	-	Chen et al. 2010

Source: adapted from Song & Li 2014a

Table 11 shows the heavy metal sediment pollution in rivers and ponds nearby Guiyu and Taizhou. The first study referred to in the table indicates that the levels of heavy metals at sampling site "Lianjiang River 2" (which was closer to the e-waste recycling site than the other sampling sites) were highest of all sampling sites, except for the lead level. Compared to the control area (reservoir), the sediment pollution of heavy metals of the sampling sites in Guiyu was more serious (Leung et al. 2006).

Wang et al. (2009a) found significantly higher levels of heavy metal contamination in the Nanyang river compared with the Liangjiang River. This held in particular for copper (2153.88 mg/kg), zinc (482.75 mg/kg) and lead (394.5 mg/kg). Also in the Nanyang river in Taizhou very high levels of contamination were measured, among others for lead (377.33 mg/kg) and copper (4787.5 mg/kg) (Chen et al. 2010).

According to Song & Li (2014a) these studies indicate that the heavy metals in the river sediments were mostly from e-waste recycling activities.

4.2.5 Plants

Heavy metal contamination in plant samples is a reflection of the metals' presence in the soil. Heavy metals in food plants can accumulate in the human body through the food chain and as a result impose adverse impacts on human health. Most studies in China on heavy metal contamination of (food) plants have focused on the contamination of rice. Rice is the most important agricultural crop in China. Therefore maintaining its quality is critical to human health (Song & Li 2014a).

Table 12: Environmental pollution of heavy metals in plants

Pollutants	Sampling time	Sampling site	Location	Sample size	Concentrations in mg/kg	Reference values	References
Pb and Cd	08/2009and 10/2010	Sorghum bicolor, Moench and rice stalks at e-waste dump site and roadside	Guiyu	20-30	Pb: 11.41; 18.74 Cd: 2.92; 0.90	-	Alabi et al. 2012
Pb	11/2005	Rice and hull	Taizhou	13	Pb: 0.69	0.20	Fu et al. 2008
Zn	10/2006	Rice grain	Wenling (nearby Taizhou)	96	Zn: 20.69	-	Zhao et al. 2010-2011

Source: adapted from Song & Li 2014a

In Table 12 we have selected a few studies on plant contamination in Guiyu and Taizhou. In Taizhou the mean level of lead in polished rice (0.69 mg/kg) was more than three times higher than the maximum allowable concentration (MAC) (0.20 mg/kg) (NY5115-2002). The average level of cadmium was only slightly higher than the MAC, but the cadmium in 31 per cent of the rice samples exceeded the MAC (0.20 mg/kg). Furthermore, cadmium and lead levels in local rice samples were much higher than in commercial rice samples from the control areas. Heavy metal concentrations were generally higher in rice hulls than in polished rice (Fu et al. 2008).

Zhao et al. (2010-2011) concluded that the enrichment index[8] (from soil to rice) varied significantly with heavy metals in paddy fields. The study also revealed that high levels of soil organic matter and sand increased the accumulation and availability of heavy metals in rice.

[8] The enrichment index is defined as the metal concentration in rice divided by that in soil (Zhao et al. 2010).

Some studies also investigated the levels of heavy metal levels in other plants than rice collected from e-waste recycling sites. Alabi et al. (2012) for instance reported values of heavy metal concentration in plants, which were considerably higher than those in rice, suggesting that these plants had a higher enrichment index than rice. Other studies (not referred to in Table 12) revealed concentrations of cadmium and lead in most vegetables which exceeded the Chinese food safety limit several times (Song & Li 2014a).

4.3 Health impacts

Unsurprisingly the environmental impacts associated with WEEE have translated directly into a serious public health risks and impacts. Many of these risks are already apparent in medical diagnoses and statistical research. Some longer term risks may be yet to develop and will still need to be understood (Grant et al. 2013; ILO 2012). Evidence suggests that WEEE is significantly increasing incidences of chronic disease, threatening not just workers but also current residents and future generations. High prevalence of skin, gastric, respiratory, hematic, neurological, prenatal, natal and infant diseases related to WEEE are becoming increasingly well documented (Geeraerts et al. 2015; Grant et al. 2013; Song & Li 2014b). Health impacts and risks from WEEE can also be indirect, for instance if pollution enters the food and water systems. Contamination already outlined presents opportunities for harmful toxins to accumulate in agricultural crops, livestock and eventually humans (Song & Li 2014a; ILO 2012). In places like Guiyu and Taizhou, where rice is still cultivated, these risks are amplified and high concentrations of toxins found in agricultural soils and vegetation there suggest that this is already a reality (Sepúlveda et al. 2010; Song & Li 2014a).

4.3.1 Sources and routes of exposure

Sources of exposure of e-waste can be classified into three sectors: informal recycling, formal recycling and exposure to hazardous e-waste compounds remaining in the environment (i.e. environmental exposure) (Song & Li 2014b; Grant et al. 2013). Given that illegally imported e-waste is mostly processed in the informal sector, this study is primarily interested in the human health impacts from informal recycling and environmental exposure. In general, people are exposed to e-waste and its hazardous components through inhalation, dietary intake and soil/dust ingestion (Song & Li 2014b; Frazzoli et al. 2010). In the next paragraphs only a selection of figures on exposure is presented, thereby focusing primarily on PBDEs and lead and on the e-waste recycling areas of Guiyu and Taizhou for purposes of illustration.

Dietary intake

According to estimates more than 90 per cent of total human exposure to heavy metals and organic pollutants is from food. Table 13 presents the results from various studies on the dietary intake from PBDEs in Guiyu, Taizhou and Qingyuan. The food groups included in the studies are: freshwater fish, marine fish, shellfish, meat and meat products, viscera, eggs, cereals, vegetables and fruits.

One study showed that Guiyu inhabitants had a PBDEs dietary intake of 931 ng/kg bw[9]/day), thereby exceeding the US EPA's reference dose (100 ng/kg bw/day) multiple times (US EPA, 2008), while Taizhou (44.7 ng/day·kg bw) and Lin'an (1.94 ng/kg bw/day) inhabitants had lower intakes (Chan et al. 2013; see Table 13).[10] Another study of food consumption of local habitants in Taizhou found that the estimated daily dietary intake of PBDEs (3.84 ng/kg bw/day) was higher than in the control groups (1.73 ng/kg bw/day), but much lower than the value in Chan et al. (2013).

[9] bw = body weight

[10] The food consumption data were obtained via semi-quantitative food intake questionnaires.

Table 13: Daily dietary intake from e-waste pollution

Pollutants	Sampling time	Exposure pathways	Exposed group	Sampling site	Daily intake (ng/kg bw/day)[a]	Reference values	References
PBDEs	-	9 food groups	habitants	Guiyu, Taizhou; Lin'an	931; 44.7; 1.94		Chan et al. 2013
PBDEs	-	Chicken meat and eggs	habitants	Wenling	Chicken: 1.8 Eggs: 11.7		Qin et al. 2011
PBDEs	2005	Milk	Breastfed babies	Taizhou	572 ± 839		Leun et al. 2010
PBDEs	11/2011	Duck eggs	habitants	Taizhou; San Men County	3.18-102.48; 0.54		Labunska et al. 2013
PBDEs	-	Chicken and duck	habitants	Qingyuan	1.33		Luo et al. 2009
PBDEs (PBBs)	07/2010	Eggs	habitants	Qingyuan; control areas	200.14; 9.61		Zheng et al. 2012
PBDEs (PBBs, PCBs)	04/2007	7 food groups and water	habitants	Taizhou; Yandang	3.84; 1.73		Zhao et al. 2009
Pb (and Cd)	11/2005	Rice	habitants	Taizhou	Pb: 3.7 µg/kg bw/day	1 µg/kg bw/day [b, c]	Fu et al. 2008

Source: adapted from Song & Li 2014b Note: [a] 1 nanogram = 1 ng = 10^{-9} kg; [b] FAO/WHO's tolerable daily intake; [c] According to the EFSA Panel on Lead in Food, the $BMDL_{01}$ dietary intake value for developmental neurotoxicity in 6 year old children (i.e. blood lead level of 12 µg/L) corresponds to a dietary lead intake value of 0.50 µg/kg b.w. per day (EFSA 2010).

The highest dietary intake of PBDEs from eggs was found in Qingyuan (200.14 vs. 9.61 ng/kg bw/day), many times higher than in the control group (9.61 ng/kg bw/day) or in groups of other exposed sites (11.7 ng/kg bw/day in Wenling and 3.18-102.48 ng/kg bw/day) in Taizhou (Labunska et al. 2013; Qin et al. 2011; Zheng et al. 2012).

Leung et al. (2010) estimated the intake of PBDEs of 6-month-old breastfed babies living on the e-waste site in Taizhou at values between 572 and 839 ng/kg bw/day. These were at least 57 times higher than the intake of babies from the control site (10.1-4.60 ng/kg bw/day) (Leung et al. 2010). Furthermore, the maximum calculated value (2240 ng/kg bw/day) exceeded the US EPA's chronic oral reference dose for penta-BDE (2000 ng/kg/day) (Leung et al. 2010; Jones-Otazo et al. 2005).

Only one study included in the review by Song & Li (2014b) assessed the dietary intake of heavy metals (from rice) (Fu et al. 2008). As lead is concerned, the average daily intake of lead through rice consumption only by a group of habitants in Taizhou was 3.7 µg/kg bw/day, several times higher than FAO/WHO's tolerable daily intake (1 µg/kg bw/day) (Fu et al. 2008).

Inhalation

Dietary exposure is the most important source of human exposure to e-waste. Despite this, several studies have stated that inhalation of contaminated air can be a significant exposure route, in particular for e-waste workers. As no studies have been identified in relation to inhalation of PBDEs and metals from the air, two studies are discussed that looked into PCDD/Fs (see Table 14).

Wen et al. (2011) estimated the total average daily intake of PCDD/Fs in Fengjiang at 62.11 pg WHO-TEQ/kg bw/day for adults and 110.11 pgWHO-TEQ/kg bw/day for children, which substantially exceeds the WHO tolerable daily intake of 1-4 WHO-TEQ/kg·day-1 (WHO 2013). Li et al. (2007) estimated the PCDD/Fs intakes from inhalation based on particulate and gas samples in Guiyu. They concluded that inhabitants from Guiyu were subject to high exposure and high health risk from PCDD/Fs (daily inhalation of 4.5 and 2.54 pgWHO-TEQ/kg bw/day for children and adults respectively) compared with inhabitants from Guangzhou (0.090 and 0.069 pgWHO-TEQ/kg bw/day for children and adults) (Yu et al. 2006). The daily intake doses for children in both studies were almost twice those for adults. It is therefore not surprising that children are the most vulnerable for bad air quality and that for instance 80 per cent of children in Guiyu suffer from respiratory diseases (Li et al. 2007).

Box 4: The toxic equivalency factor

The toxic equivalency factor (TEF) expresses the toxicity of dioxin-like compounds, including PCBs. By using the toxic equivalency factor the toxicity of a mixture of dioxins and dioxin-like compounds can be expressed in a single number – the toxic equivalency (TEQ). The total TEQ is the sum of the concentrations of individual compounds multiplied by their relative toxicity (TEF) (Van den Berg et. Al 2006). The TEQ reports the toxicity-weighted masses of mixtures of dioxin-like compounds.

The WHO-TEQ refers to the universally accepted scheme presented by the World Health Organisation.

Table 14: Daily intake of e-waste pollution through inhalation and other routes

Pollutants	Sampling time	Exposure pathways	Exposed group	Sampling site	Daily intake (pg WHO-TEQ/kg bw/day)[a]	Reference values	References
Inhalation from the air							
PCDD/Fs	07/2006 & 01/2007	Outdoor air	Adults & children	Fengjiang area	Adults: 62.11 Children: 110.11	-	Wen et al. 2011
PCDD/Fs	09/2005	Outdoor air	Adults & children	Guiyu	Adults: 2.54 Children: 4.5	-	Li et al. 2007
Soil/dust ingestion and dermal exposure							
PCDD/Fs	-	Soil/dust	Adults & children	Taizhou; Guiyu	Children: 2.3; 1.982 Adults: 0.363; 0.499	Wenling: Children: 0.0013 Adults: 0.0003	Ma et al. 2008

Source: adapted from Song & Li 2014b. Note: [a] 1 picogram = 1 pg = 10^{-12} kg ; TEQ = toxic equivalents

Soil/dust ingestion and dermal exposure

Ma et al. (2008) assessed the daily intakes of TEQs of PCDD/Fs via soil/dust ingestion and dermal exposure, based on collected soil and dust samples from e-waste recycling facilities. Children and adults living in Taizhou were respectively exposed to 2.3 and 0.363 pg TEQ/kg bw/day (see Table 14). These doses were at least 1200 times higher than those in Wenling, a reference site. The estimated exposure values for children and adults in Wenling were only 0.0013 and 0.0003 pg TEQ/kg bw/day respectively. Ma et al. (2008) assessed as well the daily intakes via soil/dust ingestion and dermal exposure in Guiyu. These were assessed at 1.982 and 0.499 pg TEQ/kg bw/day for children and adults respectively (see Table 14).

4.3.2 Body burdens of e-waste exposure

The presence of pollutants in the environmental media does not necessarily imply significant human exposure to these pollutants. Assessing the level of pollutants in human bodies however can provide direct information about the level of human exposure and the potential public health risks. In recent years many studies in China have clearly indicated a causal relationship between emissions from informal e-waste recycling and dumping activities and the human body burden. These studies commonly include following types of human tissues: placenta, umbilical cord blood, breast milk, blood and serum, urine, hair and other tissues (Song & Li 2014b). For purposes of illustration figures on pollutant levels are only presented in relation to placenta, blood and serum and in relation to selected pollutants (primarily metals such as lead and cadmium). The other human tissues are only discussed briefly in a qualitative way.

Placenta

Table 15 presents several studies which have examined the human body exposure to e-waste in placentas collected after childbirth. These studies compared pollutant levels in placenta from mothers in e-waste recycling towns with those in control towns (Guiyu) with no exposure to e-waste pollution. Concerning the heavy metal levels, these studies concluded that most heavy metal levels in placentas from Guiyu were higher than those in placentas from the control towns, although there were some exceptions. In two studies (Li et al. 2011; Zhang et al. 2011), the cadmium levels in placentas from Guiyu were significantly higher than those from Chaonan. Guo et al. (2010) however did not find a significant correlation as to cadmium levels. As to lead, levels showed significant correlations in the two studies (Guo et al. 2010; Zhang et al. 2011). The total lead value in placentas in Guiyu was circa two times higher compared to the value of the control group in Chaonan (301.43 vs. 165.82 ng/g) (Guo et al. 2010). Zhang et al. (2011) also found lead values about two times higher in Guiyu compared to the control town (Shantou), though the correlation was less significant (521.01 vs. 273.24 ng/g, p = 0.299).

Leung et al. (2010) investigated the PBDE body burdens of women. The ones from the e-waste recycling site (Taizhou) were 19 times higher than those from the control site (19.5 ± 29.9 vs. 1.02 ± 0.36 ng/g fat) (see Table 15).

Table 15: Neonate body burdens (placenta) in Guiyu and Taizhou

Pollutants	Sampling time	Exposed groups	Sampling sites	Human body burden	Reference values	References
Pb, Cd (Cr, Ni)	10/2008-05/2009	220 mother-infant pairs (101:119)	Guiyu; Chaonan	Pb: 301.43; 165.82 ng/g (p = 0.010) Cd: 108.75; 104.15	-	Guo et al. 2010
Cd	2006	423 mother-infant pairs (101:119)	Guiyu; Chaonan	170 ± 480; 100 ± 110 ng/L (p〈0.01)	-	Li et al. 2011a
Cd, Pb	10/2008-06/2009	mother-infant pairs (101:119)	Guiyu; Shantou	Cd: 83.99; 51.59 ng/g (p〈 0.001) Pb: 521.01; 273.24 ng/g (p = 0.299)	-	Zhang et al. 2011
PBDE	2005	10 women	Taizhou; Lin'an	19.5 ± 29.9; 1.02 ± 0.36 ng/g fat	-	Leung et al. 2010

Source: adapted from Song & Li 2014b.

Blood and serum

Table 16 presents studies on the human body burden in blood and serum as a result of e-waste exposure which focus on heavy metals and lead in particular. It should be noted that several studies have also investigated levels of PBDEs, PCBs and dioxins in blood and/or serum (see e.g. Song & Li (2014b)). All studies presented in Table 16 indicated that the lead levels in the blood of children living in the e-waste recycling areas (Guiyu and Taizhou) were significantly higher than the blood lead levels in the control areas (all $p < 0.01$) and that most blood lead levels exceeded the control standards. According to the diagnostic blood lead level criteria for children as defined by the U.S. Centers for Disease Control (CDC 1991) children with a lead level in their blood equal to or higher than 100 µg/L (or 10 µg/dL) are considered to have an elevated blood lead level. These lead levels were correlated with location of residence, parents' involvement in e-waste recycling, the use of the home as a recycling workshop and the nibbling of toys by children (Song & Li 2014b). It should also be noted that the EFSA Panel for Lead in Food determined the 95th percentile lower confidence limit of the benchmark dose (BMD) of 1 per cent extra risk (BMDL01) of 12 µg/L blood lead as a reference point for the risk characterisation of lead when assessing the risk of intellectual deficits in children measured by the Full Scale IQ score (see 4.3.4 for more details)

Furthermore, Zheng et al. (2008) concluded that both the blood lead levels in Guiyu and the proportions of blood lead levels higher than 10.0 mg/dL increased with the children's age: older children were more likely to have higher blood lead levels than younger children. They also discovered that the blood lead levels of children aged 5, 6 and 7 years in Guiyu were significantly higher than those in Chendian (all $p < 0.01$).

Table 16: Human body burden in blood and serum

Pollutants	Sampling time	Exposed groups	Sampling sites	Average human body burden (γg/L) ± Standard deviation	US CDC ref. level ((γg/L)[a]	References
Pb (and Cd)	-	246 children aged 3-8	Guiyu	73.0 (43.3-154.3); no statistical analysis	100	Yang et al. 2013
Pb (Cu and Cd)	-	104 persons (48:56)	Southeast China	114.9 vs 91.04 (p<0.01)	100	Wang et al. 2011a
Pb (Cd, Cu, Cr, Hg and PCBs)	-	76 workers and habitants (40:36)	Taizhou vs control area	150.63 ± 184.41 vs 84.37 ± 33.81	100	Zhang et al. 2012
Pb	2004	226 children (165:61)	Guiyu vs Chendian	153.0 ± 57.9 vs 99.4 ± 40.5 (p<0.01)	100	Huo et al. 2007
Pb	2006	278 children aged 1-7 (154:124)	Guiyu vs Chendian	131.7 ± 59.8 vs 100.4 ± 48.5	100	Zheng et al. 2008
Pb	01-02/2008	303 children aged 3-7 (153:150)	Guiyu vs Chendian	144.3 ± 69.3 vs 87.2 ± 43.4 (p<0.01)	100	Liu et al. 2011
Pb	09-11/2004	226 children aged 1-6 (165:61)	Guiyu vs Chendian	153.0 ± 57.9 vs 99.4 ± 40.5 (p<0.01)	100	Xu et al. 2006
Pb	2006	136 children aged 3-6 (85:51)	Guiyu vs Chendian	117.8 vs 89.3 (p<0.01)	100	Han et al. 2007
Pb	-	226 children aged 1-6	Guiyu vs	153.0 vs 57. 9 (p<0.01)	100	Peng et al.

		(165:61)	Chendian			2005
Pb	06/2010	178 children aged 11-12 (108:70)	Luqiao vs Chun'an	6.97 vs 27.8 ($p<0.01$)	100	Wang et al. 2012a
Pb	-	138 persons (59:79)	South China (exposed vs control)	99.83 vs 92.25 ($p<0.01$)	100	Zhang et al. 2007

Source: adapted from Song & Li 2014b. Note: The EFSA Panel for Lead in Food determined the 95th percentile lower confidence limit of the benchmark dose (BMD) of 1 per cent extra risk ($BMDL_{01}$) of 12 µg/L (B-Pb) as a reference point for the risk characterisation of lead when assessing the risk of intellectual deficits in children measured by the Full Scale IQ score (see 4.3.4 for more details).

Other human tissues

Studies on pollutant levels in umbilical cord blood found that newborn babies from Guiyu are exposed to e-waste and have a serious human body burden of pollutants. Several studies also found significantly higher levels of pollutants in hair samples from inhabitants of e-waste recycling towns and e-waste workers. Some studies also found significantly higher levels of heavy metals in urine samples from e-waste workers compared to samples from control groups (Song & Li 2014b).

4.3.3 Health impacts

Many studies have been conducted on the health impacts of exposure to e-waste in China. The majority of select studies reviewed by Grant et al. (2013) showed associations between exposure to e-waste and physical health outcomes, including lung function, physical growth, reproductive health, thyroid function and changes in cellular expression and function. Negative associations were also shown for blood lead levels and IQ in children. Outcomes were mainly reported from Guiyu and Taizhou.

Lung function

Zheng et al. (2013) examined associations between exposure to heavy metals (amongst others chromium and nickel) and lung function in 144 schoolchildren (aged 8–13 years) from Guiyu and from Liangying, a control town with no evidence of e-waste recycling. Boys aged 8 and 9 years living in Guiyu had a lower forced vital capacity[11] than those living in Liangying (1859 mL vs 2121 mL, p=0.03). Significant negative correlations were found between blood chromium concentrations and forced vital capacity in children aged 11 and 13 years and serum nickel concentrations in children aged 10 years (Zheng et al. 2013; Grant et al. 2013).

Physical growth

Two studies showed that children living in the e-waste recycling town of Guiyu had significantly lower weight, height and body-mass index than children living in the control area Liangying (Zheng et al. 2013; Huo et al. 2007). One study also found negative correlations between height and weight on the one hand and concentrations of manganese in blood and nickel in serum on the other (Zheng et al. 2013). The other study however did not show adverse effects of lead on growth in young children (Huo et al. 2007).

[11] Forced vital capacity (FVC), also called "forced expiratory vital capacity" (FEVC), is the volume change of the lung between a full inspiration to total lung capacity and a maximal expiration to residual volume and whereby the exhalation is done forcefully. The preceding maximal inhalation does not need to be done forcefully. (Source: www.spirxpert.com/indices5.htm)

Reproductive health

In most studies reviewed by Grant el. 2013 consistent effects of e-waste exposure have been found with increases in spontaneous abortions, stillbirths, and premature births, and reduced birth weights and birth lengths (Guo et al. 2012; Wu et al. 2011, 2012; Xu et al. 2012). Adverse birth outcomes have been associated with increased exposures to PAHs and POPs, including PBDEs, PCBs and perfluoroalkyls (Guo et al. 2012; Wu et al. 2012). The principal exception is the lack of association between exposures to metals and adverse birth outcomes (Guo et al. 2010).

Mental health outcomes

In two studies lead was investigated as the main chemical agent associated with mental health outcomes resulting from e-waste exposure (Liu et al. 2011). In one study researchers assessed temperament in children (Liu et al. 2011), in the other researchers generated neonatal behavioural neurological assessment scores (Xu et al. 2012). It was found that children in Guiyu had higher concentrations of lead in their blood than children living in towns with no e-waste recycling (Liu et al. 2011; Xu et al. 2012).

Neonates (newborn babies) had increased concentrations of lead in cord blood and meconium. These were correlated with the mother's involvement in e-waste recycling, time spent living in Guiyu before and during pregnancy, time spent in e-waste recycling facilities while pregnant, and the father's involvement in e-waste recycling activities (Li et al. 2008a). Neonatal behavioural neurological assessment scores differed considerably among the babies in Guiyu and those in the control town (Li et al. 2008a). Increased concentrations of lead were associated with abnormalities in temperament scores (Liu et al. 2011) and low neonatal behavioural neurological assessment scores (Li et al. 2008a).

Thyroid function

The studies reviewed by Grant et al. 2013 showed that reported effects on thyroid-stimulating hormone (TSH) were not consistent.

DNA damage

Most studies identified Grant et al. (2013) as studies including health outcomes at cellular level found that inhabitants of e-waste recycling towns or e-waste recycling workers had greater DNA damage than those living in the control town (Yuan et al. 2008; Liu et al. 2009; Chen et al. 2010; Wang et al. 2011a). According to Grant et al. (2013) the studies however do not have the power to exclude other contributory factors next to e-waste exposure. A study by Li et al. (2008b) reported significant differences in lymphocytic DNA damage in newborn babies from Guiyu and newborn babies from the neighbouring fishing town Chaonan. Newborn babies from Guiyu had greater DNA damage than did neonates from Chaonan. Significant correlations between chromium levels in blood and DNA damage in neonates (Li et al. 2008b).

Intelligence quotient

One study investigated the educational outcomes of e-waste exposure whereby lead was analysed as the main chemical agent (Wang et al. 2012a). The effects of lead levels in blood on intelligence quotient (IQ) in children aged 11 and 12 years were assessed. The children were from the e-waste recycling town Luqiao in Thaizou, the tinfoil manufacturing area Lanxi and the control town Chun'an. As no significant differences in IQ were recorded among the three sites, the results from the three individual site samples were combined. 38.9 per cent of the children from Luqiao and 35.1 per cent of the children from Lanxi had blood lead levels above 10 μg/dL, whereas all examined children from Chun'an had levels below 10 μg/dL. Wang et al. (2012) associated health risks for intellectual function in children with blood lead levels below 10 μg/dL with a decrease in IQ levels. Every 1 μg/dL increase in blood lead level resulted in a 0.71 point decrease in IQ (Wang et al. 2012a).

Discussion

E-waste contains a unique combination of persistent hazardous substances. Other sources of exposure are however difficult to rule out, in particular in China. In studies that accounted for confounding variables, researchers still found important associations between e-waste exposure and adverse outcomes in exposed populations. Even when other variables such as age and smoking were accounted for, e-waste exposure came out as an independent risk factor (Grant et al. 2013).

4.3.4 Case: elevated lead levels in human body and IQ scores of children

All data presented in the previous sections suggest that the contaminant levels in different environmental media at and nearby e-waste recycling sites are significantly higher than those at reference sites. As a result e-waste recycling workers and people living at or nearby e-waste recycling sites are subject to increased health risk. Long-range transport of e-waste-derived pollutants, e.g. through the food chain, can also subject inhabitants of adjacent regions to unintended health risks (Ni & Zeng 2013). In the following paragraphs we undertake an effort to asses these health risks more in detail in relation to lead as a major pollutant arising from informal e-waste recycling and dumping in China and its impacts on the neurological development of children in terms of IQ scores.

Hazard identification and characterization

It is well known that heavy metals (such as lead) persist in the environment and lead to poisoning at low concentrations through bioaccumulation in plants and animals or bioconcentration in the food chain (Song & Li 2014a; see also previous sections). Oral ingestion of contaminated food has been proved to be an important pathway for the transfer of lead (and other metals) from the environment to the human body. Children in particular are amenable to heavy metal exposure due to high gastrointestinal uptake and the permeable blood-brain barrier (Ogunseitan 2013). The high concentrations of toxic metals in dust will pose a severe health threat to children, due to the involuntary or direct ingestion of contaminated dust particles via the ''hand to mouth'' pathway (Song & Li 2014a). In human bodies lead interferes with behaviour and learning abilities; copper results in liver damage; and chronic exposure to cadmium increases the risk of lung cancer and kidney damage (Grant et al. 2013; Song & Li 2014a).

In its scientific opinion on lead in food the EFSA Panel on Contaminants in the Food Chain identified the developmental neurotoxicity in young children, cardiovascular effects and nephrotoxicity in adults as the basis for its risk assessment of lead in food (EFSA 2013). As to developmental neurotoxicity in young children, the EFSA Panel determined the 95^{th} percentile lower confidence limit of the benchmark dose (BMD) of 1 per cent extra risk ($BMDL_{01}$) at 12 μg/L (B-Pb) as a reference point for the risk characterisation of lead when assessing the risk of intellectual deficits in children measured by the Full Scale IQ score. For this benchmark dose level (BMDL) of 12 μg/L (B-Pb), the EFSA Panel has calculated a corresponding dietary intake: the $BMDL_{01}$ dietary intake value for developmental neurotoxicity in 6 year old children corresponds to a dietary lead intake value of 0.50 μg/kg bw per day (EFSA 2013).

Risk characterisation

From the previous chapters we learned the following on lead: it was estimated that inhabitants of Taizhou had a dietary intake of 3.7 μg/kg bw per day through consumption of rice only. This exceeds the EFSA dietary lead intake value for developmental neurotoxicity in children several times and does not even take into account consumption of other food (e.g. which may contain lead (or other pollutants), other exposure routes such as inhalation of air and soil/dust ingestion, the latter being in particular of relevance for children. It is very likely that similarly high values (or even higher values) would have been measured in Guiyu (and other informal e-waste recycling towns), given the relatively high concentrations of lead found in several environmental media such as air, dust and soil and in rice and other plants (e.g. vegetables) in Guiyu and surroundings (see 4.2). Furthermore, very high average blood lead levels have been found in children in Guiyu (and Taizhou) (see Table 16), which exceeded the $BMDL_{01}$ of 12 µg/L multiple times, with most blood lead levels significantly higher than 100 µg/L (or 10 µg/dl). Based on this, we can conclude that almost all children living in informal e-waste recycling centers such as Guiyu (and Taizhou) are or have been subject to lead exposure which exceeds the $BMDL_{01}$ intake level of 0.50 µg/kg b.w. per day for neurodevelopmental effects (the benchmark dose approach is explained in Box 2 on page 18). This implies that potentially all children living in these informal e-waste recycling areas have been affected in their neurological development. Assuming that 70 per cent of the villages/neighborhoods of Guiyu are engaged in informal e-waste recycling and that this has remained stable over the years, it is estimated that 22,400 children (16,000-25,600) born in the period 1995-2013 have had blood lead levels exceeding the $BMDL_{01}$ and as a result have been affected in their neurological development. This implies that 22,400 children (16,000-25,600) have been subject to a drop in intelligence as a result from informal e-waste recycling and dumping (see Box 5 for more details on the calculation of the number of children affected).

Breivik et al. (2014) estimate the amounts of e-waste treated in the informal recycling sector at 1,350 kt/year (550-2,000) and 4,900 kt/year (3,600-7,200) in Guiyu and China respectively. Assuming that the ratio between the amounts treated in Guiyu and in China as a whole is the same as the ratio between the number of children affected in Guiyu and in China as a whole, it is roughly estimated that around 81,300 children (58,000-93,000) in China born in the period 1995-2013 have been affected in their neurological development as a result of the informal recycling of e-waste.

These are likely to be conservative estimates as long-range transport of e-waste-derived pollutants such as lead, e.g. through the food chain, can also subject inhabitants of adjacent regions to unintended health risks (Ni & Zeng 2013; Huo et al. 2007). In fact in many control areas blood lead levels of children were mostly also significantly higher than the $BMDL_{01}$ of 12 µg/L used by the EFSA Panel as a reference point for assessing the risk of intellectual deficits in children (see Table 16). Huo et al. (2007) for instance indicated that children's blood lead levels in rural control areas nearby Guiyu were also higher than blood lead levels in nearby urban areas (such as Zhongshan City and Shenzhen City). According to Huo et al. (2007) this was very likely the result of lead contamination having spread from Guiyu to the nearby rural control area by dust, river and air.

Box 5: Estimate of children affected by lead poisoning as a result of informal e-waste recycling

Based on the population numbers of Guiyu for 2003 and 2013 (Wang et al. 2013), the urban population growth rates in China, the overall Chinese population growth rates and overall birth rates for China, the number of children born in Guiyu in the period 1995-2013 have been estimated/calculated (see **Annex B: Guiyu population statistics** for details on these calculations**)**. The resulting number is about 32,000 children. Given that 20 of Guiyu's 28 villages are engaged in informal e-waste recycling (Wang et al. 2013), it is assumed that ca. 70 per cent of these children are affected, thereby assuming that those 20 villages represent 20/28 or ca. 70 per cent of Guiyu's overall population. Given uncertainties about the proportion of children living in these villages in combination with the likelihood that children from villages in Guiyu which are not engaged in e-waste recycling are also subject to blood lead levels higher than 12 µg/L, lower and upper bound assumptions are being applied: respectively 50 and 80 per cent of Guiyu's children are being affected. The default estimate would be 22,400 children, while the lower bound and upper bound estimates would be 16,000 and 25,600 respectively.

Epidemiological studies indeed show that exposure to lead during the early stages of children's development is linked to a drop in intelligence. Studies suggest that for each 10 µg/dl (microgram per decilitre) of blood lead, IQ is reduced by at least 1-3 points (Morgan 2013 in: University of Bristol 2013). This small effect on many individuals could be a significant burden to society, with reduced overall intellectual performance and resulting economic losses (University of Bristol 2013).

Assuming that all 81,300 children (58,000-93,000) exposed to lead as a result of living in e-waste recycling areas in China had average blood lead levels of 12 µg/dl (or 120 µg/l) of blood lead – conservative estimates based on the body burden figures from Table 16 – and assuming that for each 10 µg/dl of blood lead, IQ is reduced by 2 points (see above), it is roughly estimated that for the population of children living in informal e-waste recycling areas and born in the period 1995-2013 the intelligence has dropped with 195,120 IQ points (139,200-223,200). Based on data from the studies referred to in Table 16 it is roughly assumed that 50 per cent of the blood lead originates from informal e-waste recycling and dumping activities. This means that the children living in e-waste recycling areas in China **lost** about **97,560 IQ points (69,600-111,600)** as a result of these activities. This amounts to an **average reduction of intelligence of 1.2 points per child**. It should be noted that these figures provide an initial order of magnitude estimate.

4.4 Economic impacts

The focus of this section is on economic losses to the EU and to the EU e-waste recycling sector in particular. The economic impacts in China are only briefly touched upon. In section 4.1 of this report it was estimated that ca. 1.16 million tonnes of e-waste have been shipped illegally from the EU to China in 2012. This estimate built further on Zoeteman's estimate of 0.74 million tonnes of illegal imports of e-waste from the EU in China (Zoeteman et al. 2009). If one assumes a linear increase of annual imports in China from the EU between 2005 and 2012, **the average annual import in the period 2005-2012 amounts to 0.95 million tonnes**. These figures are used as a basis for estimating the economic impacts on the EU e-waste recycling sector and the Chinese informal e-waste collection and recycling sector. In order to give estimates for the economic impacts resulting from overall illegal exports from the EU, one might start working from following figures: in 2005 and 2012 respectively 1.9 and 2.98 million tonnes of e-waste have been exported from the EU (according to our estimates produced in section 4.1 of this report).

4.4.1 In Europe

Economic loss to the EU waste recycling sector

The illegal export of e-waste has also led to a point where the legitimate modern recycling facilities in the EU cannot obtain the expected amount of WEEE (Geeraerts et al. 2015). For instance, the UK recycling industry anticipated an annual volume of 1.5 million tonnes of WEEE to be processed once the WEEE Directive had entered into force in the UK. However, in reality the actual quantity was only a third of this amount two years after the Directive's entry into force (UNODC 2013).

In a 2012 report Amec estimated that the overall economic impact of illegal waste exports from the UK is £8.7 million (€10.7 million), including a £5.1 million (€6.2 million)[12] loss in profits to WEEE treatment facilities in the UK. As Amec recognised the inherent uncertainties in their estimates, it suggested that the true economic cost of illegal exports could be considerably higher (Amec 2012). These estimates of economic loss built further on Amec's estimates of total UK waste exports in 2012 (15 million tonnes), of total illegal waste (between 0.6 and 1.7m tonnes) and the share of WEEE illegal exports (0.5-0.6 million tonnes) (Amec 2012). Based on these 2012 figures and our estimate of overall illegal e-waste exports from the EU in 2012 (see Box 6), a rough and simple extrapolation is made for the EU as a whole. If 0.5-0.6 million tonnes of illegal e-waste exports from the UK in 2012 correspond with a £5.1 million loss in profits to the UK recycling industry, **2.98** million tonnes of illegally exported e-waste from the EU in 2012 might correspond roughly with **£25.3 million to £30.4 million loss in profits to the EU recycling industry**

[12] The Eurostat's annual average exchange rates for 2012 have been used to convert GBP to Euros (available at: http://appsso.eurostat.ec.europa.eu/nui/show.do?dataset=ert_bil_eur_a&lang=en).

(€31.2 million – €37.5 million)[13] (assuming that the ratio between tonnes of illegal exports and loss to the industry is the same for the UK and the EU). If one looks at the illegal e-waste **exports to China only** (**1.16** million tonnes in 2012), the **EU recycling industry** might have **lost £9.86 million to £11.83 million in profits in 2012 (€12.2 million - €14.6 million**). To put this into perspective, the EU e-waste recycling market earned revenues of US$1.3 billion in 2012 (or €1.012 billion) and is estimated to earn US$1.79 billion (or € 1.39 billion) in 2020 (Frost & Sullivan 2013).

Box 6: Estimates of e-waste export from EU and China import from EU in 2012

Section 4.1 presented the following estimates of e-waste volumes exported from the EU to China:

- According to the StEP project 9,918,000 tonnes of e-waste have been generated in the EU in 2012 (StEP 2015c).
- The amount of illegally exported e-waste from the EU in 2012 was estimated to be around **2.98 million tonnes**.
- According to UNODC (2013) in 2012 8 million tonnes of WEEE were imported into China.
- The volume of illegally exported e-waste from the EU into China amounted to **1.16 million tonnes** in 2012.

It should also be noted that many European based "recyclers" of e-waste act as brokers for China bound WEEE and thus profit from this illegal trade.

If larger volumes of e-waste would have been treated within the EU, it would have helped EU companies to become more competitive. The EU recycling sector currently suffers from a lack of available material to treat and recycle. The volumes that now are being shipped illegally to China (and other developing countries) could have helped EU companies to compete with e.g. Chinese companies which process significant amounts of e-waste (BIO IS 2010).

According to the WEEE Forum (2013) the average intrinsic value of WEEE was about € 300 per tonne.[14] Building further on this figure the **economic value lost to the EU** as a result of illegal **exports to China** (1.16 million tonnes in 2012) is roughly estimated at **€ 348 million for 2012** only. The **economic value lost** to the EU as a result of **all illegal exports out of the EU** (2.98 million tonnes in 2012) is estimated at **€ 892 million for 2012**.

A major part of the economic value lost to the EU arising from illegal e-waste exports relates to the **loss of precious metals in these exports.** Metals are one of the main elements that can be extracted from WEEE, recycled and treated for re-use in new devices. Gold is one example of such a precious metal. Each year millions of mobile phones are discarded and exported that contain significant amounts of gold (next to many other precious metals). According to the European Recycling Platform about two grams can be recovered from 10 kilograms of discarded mobile phones, which is the amount of gold that is required to produce a wedding ring. Furthermore, the conventional way to get these two grams is to refine 10 tonnes of gold ore (ERC 2014), of which the extraction and refining is significantly more harmful to the environment and the climate.

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[13] The Eurostat's annual average exchange rates for 2012 have been used to convert GBP and USD to Euros (available at: http://appsso.eurostat.ec.europa.eu/nui/show.do?dataset=ert_bil_eur_a&lang=en).

[14] According to INTERPOL one tonne of e-waste is valued at US$500 and if the differences in the numerous types of e-waste are taken into consideration an average figure of US$375 per tonne can be estimated (UNODC 2013).

Job loss in the EU waste recycling sector

The illegal exports are likely to present a potential loss of jobs in enterprises treating e-waste, through the decrease of the quantities of e-waste having to be treated in the EU. The loss of jobs resulting from illegal e-waste exports outside of the EU can be calculated using an approach whereby the lost total turnover for the EU e-waste recycling sector is divided by the sector's labour intensity (i.e. the level of turnover per full time equivalent (FTE) job).

However, the potential loss of jobs can also be calculated using coefficients for FTE jobs for diverting 1,000 tonnes of e-waste from landfill, incineration or illegal export. For WEEE, however, only two coefficients have been identified in the literature. The first one is from a 2004 study by the London South Bank University (LEPU 2004) which calculated that for every 1,000 tonnes of waste recycled per year in London, 6 jobs would be gained across the entire waste chain. The collection and sorting of WEEE, however, provided the biggest employment opportunities, with a total of 40 jobs being created per 1,000 tonnes of material processed (LEPU 2004). This implies that **1 job** would be created per **25 tonnes** of e-waste processed.

The second coefficient is from a 2013 study for the Coalition for American Electronics Recycling (CAER) (DSM 2013). According to this study, every extra 172,000 pounds (78,018 kg or roughly **78 tonnes**) of e-waste processed in the United States which is not exported outside of the US, would generate a **new job**. This estimate represents only those jobs directly involved in processing the added e-waste. Additional indirect and induced jobs generated by spending of employees associated with the (new) direct jobs in e-waste recycling would also be generated. Typically indirect and induced jobs increase twofold the number of direct jobs established. It should be noted that this estimate is based among others on the assumption that the smelting of shredded circuit boards and other materials containing precious metals would continue to take place primarily outside of the US, as would recycling of significant quantities of CRT glass (DSM 2013).

It was decided to apply the most conservative coefficient, which is the one from the American study: **1 job per 78 tonnes** (DSM 2013). Compared to the LEPU 2004 coefficient, it is a more recent figure and it has been determined in a study fully focused on the (US) e-waste recycling industry. Assuming that a similar ratio applies to the EU e-waste recycling industry, the illegal export of e-waste from the EU in 2012 might represent a **potential loss** of about **38,000 full time equivalent (FET) recycling jobs in the EU**. Assuming a typical multiplier of 2, these direct recycling jobs would result in another **38,000** indirect[15] and induced jobs[16], for a total of **76,000 jobs**. The **illegal export to China** in particular represents a potential loss of about **14,900 FTE jobs** and another **14,900 indirect jobs**, for a total of **29,800 jobs.**[17]

4.4.2 In China

The economic impact of the illegal exports of e-waste from the EU (and other developed countries) is the creation of an industry, made up mostly of informal enterprises in China which process e-waste, attempting to generate profit from the resale of copper, steel, aluminium, gold and computer chips. There,

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[15] Indirect jobs are created as the recycling industry supports other economic activity by purchasing goods and services from other types of business establishments (such as office supply companies, accounting and legal firms, etc.) (FoE 2010).

[16] Induced jobs are jobs 'induced' or generated as a result of the spending by both direct and indirect employees in the local economy.

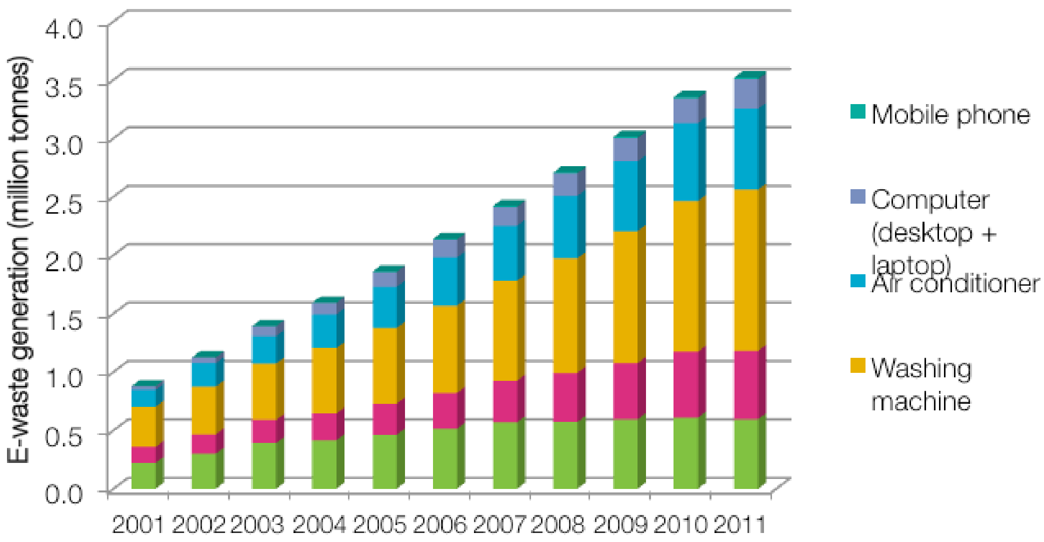
[17] Alternatively, multipliers could be applied from a 2010 study by Friends of the Earth UK (FoE 2010). In this study indirect and induced (loss of) employment has been estimated by applying separate multipliers for indirect and induced jobs: a multiplier of 1.50 to calculate the indirect jobs resulting from direct employment in the recycling sector and a multiplier of 1.75 to calculate the induced jobs. The authors of the FoE study consider these multipliers as conservative, as these are lower than those commonly used in other economic sectors (FoE 2010).

potential in resource reutilisation and for income generation, allow WEEE to be perceived as an economic opportunity. Whilst environmental and health impacts are generated mostly by recycling practices, there exist complex value chains, at various stages of which the economic value of waste is extracted.

This has seen Guiyu, for example, transform from a poor rice growing village into a highly dynamic and for some lucrative economic hub, processing up to 150 million tonnes of WEEE each year (Hicks et al. 2005). According to Wang et al. (2013) Guiyu is home to more than 300 companies and 3,000 individual workshops that are involved in the recycling of e-waste, with nearly 100,000 migrant labourers fuelling the business. Those engaged on dismantling and processing e-waste earn an average wage equivalent of US$1.50 per day (ILO 2012). About 150 million tonnes of electronic components are being recycled with an output value of nearly 1.56 billion yuan (approximately €0.19 billion)[18], accounting for more than 90 per cent of the town's industrial output value (ILO 2012). Wang et al. (2013) estimate that around 250,000 workers work in the informal e-waste recycling sector in China as a whole (see also Breivik et al. 2014).

It should be noted however that the situation in Guiyu (and other traditional recycling areas) seems to have changed in recent years as a result of governmental initiatives to regulate informal recycling activities and to clean up the local environment (Personal Communication 2015). In addition, domestically generated WEEE has become increasingly important for the informal e-waste recycling sector over the years (see Figure 3) and as a result the share of e-waste from the EU and other parts of the developed world seems to be decreasing. The picture is however not fully clear as most English publications on Guiyu are several years old. It is not fully clear either to what extent illegal imports of e-waste from the EU and elsewhere have been effectively reduced since the Chinese government started to control imports more strictly under the Green Fence program (see for more details: Geeraerts et al. 2015).

Figure 3: Generation of e-waste in China 2001-2011 (in million of tonnes)



Source: Balde et al. 2015

Given that the informal recycling of illegally imported e-waste has resulted in substantial environmental and health effects, China has suffered and will suffer economic losses as well. To give one example: given that around 81,300 children (58,000-93,000) in China born in the period 1995-2013 are very likely to have

[18] The Eurostat's annual average exchange rates for 2012 have been used to convert CNY to Euros (available at: http://appsso.eurostat.ec.europa.eu/nui/show.do?dataset=ert_bil_eur_a&lang=en).

been affected in their neurological development (as they all had blood lead levels of more than 100 γg/L (or 10 g/dL) (see section 4.3.4), given that studies suggest that for each 10 μg/dL of blood lead, IQ is reduced by at least 1-3 points (Grosse et al. 2002) and given that a decrease of 1 IQ point in children can be associated with a decrease of later productivity of about 2 per cent (EFSA 2010), the informal recycling of e-waste in China has and will lead to **significant economic losses** as a result of **reduced overall intellectual performance**.

4.5 Conclusions

Quantifying the illegal export of e-waste from the EU (to China) is especially challenging as there is very little clear information upon which estimates can be based (given the illegal nature of these e-waste shipments). Nevertheless an attempt has been made to come up with estimates for 2005 and 2012. For these years it was estimated that around 0.74 and 1.16 million tonnes of e-waste have been imported in China from the EU. These estimates are based on reliable estimates of the volumes of WEEE generated in the EU and assumptions about the percentage of WEEE that is being exported from the EU and the percentage of WEEE from the EU that is being imported in China. Given the uncertainty surrounding these 'guesstimates', it is suggested to work with minimum and maximum EU export scenarios and minimum and maximum import in China scenarios.

The illegal export of e-waste from the EU to China has resulted – through the intensive informal recycling of e-waste in China – in the release of large amounts of contaminants in the local environment such as heavy metals, PBDEs, PCDD/Fs, PBCs, CFCs and PAHs. It has caused in particular high concentrations of heavy metals (such as lead, cadmium, mercury, copper, zinc, etc.) in the surrounding air, dust, soils, sediments and plants. The exceedance of various national and international environmental quality standards imposes significant negative effects on the environment.

The potential annual emissions of some environmental contaminants associated with e-waste imported illegally in China from the EU were estimated. These estimates are based on concentrations in components of Swiss e-waste and on the estimates of the 2005 and 2012 illegal imports in China from the EU. To give one example: it was estimated that respectively 10 and 16 tonnes of PCBs from EU e-waste were potentially released in the Chinese environment in 2005 and 2012.

Given the scale and complexity of the e-waste problem, it was within the context of this project not possible to quantify the total environmental impacts from informal e-waste recycling in China or in particular areas of China, not to mention the impacts from the informal recycling in China of the millions of tonnes of e-waste imported annually from the EU (i.e. the share of EU-borne e-waste impacts within the overall impacts from informal recycling in China). This resulted among others from uncertainties about the sources of the contamination in recycling areas and control areas and in particular from the exact shares of these sources in the overall contamination. It should however be noted that many studies were able to indicate that certain pollution cases were mainly originating from e-waste recycling activities.

Given the above and in particular given the complexity of the environmental impacts from e-waste recycling and dumping, the report presents only select data. It focuses mostly on the environmental impacts (i.e. pollution levels) related to heavy metals and lead and zinc in particular. And mostly studies investigating or results regarding adverse environmental impacts in Guiyi and Taizhou, as the most representative locations for informal e-waste recycling, are being referred to.

As to health impacts, this study demonstrates that illegal exports from the EU are significantly increasing incidences of chronic disease in China, threatening not just workers but also current residents of e-waste recycling areas and adjacent regions and future generations. Illegal exports from the EU result (through the informal recycling and dumping) in high prevalence of skin, gastric, respiratory, hematic, neurological, prenatal, natal and infant diseases in China. As to health impacts, figures on the routes of exposure, on the human body burden and on scientifically established associations and/or causal links between e-waste exposure and several kinds of diseases and other health impacts are presented. As with environmental effects, only select figures were presented, primarily in relation to heavy metal contamination in Guiyu and

Taizhou. Select scientific studies (in China) show associations between exposure to e-waste and physical health outcomes such as:

- decreased lung function (i.e. lower forced vital capacity);
- decreased physical growth of children (i.e. lower weight, height and body-mass index);
- reduced reproductive health (i.e. increases in spontaneous abortions, stillbirths, and premature births, and reduced birth weights and birth lengths);
- changes in cellular expression and function (i.e. increased DNA damage).

Negative associations were also shown for blood lead levels and IQ in children.

A more detailed quantification has been carried out for the impacts of lead poisoning resulting from e-waste exposure on children's neurological development, expressed in terms of children's IQ scores. Based on data from various studies it was concluded that almost all children living in informal e-waste recycling centers such as Guiyu and Taizhou are or have been subject to lead exposure which exceeds the $BMDL_{01}$ intake level of 0.50 µg/kg b.w. per day as determined by the EFSA Panel on Contaminants in the Food Chain for neurodevelopmental effects. This implies that potentially all children living in these informal e-waste recycling areas have been affected in their neurological development. It is estimated that roughly 22,400 children (16,000-25,600) born in Guiyu in the period 1995-2013 have had blood lead levels exceeding the $BMDL_{01}$ and as a result have been subject to a drop in intelligence as a result from informal e-waste recycling and dumping. For China as a whole it is conservatively estimated that **around 81,300 children (58,000-93,000) born in the period 1995-2013** have been **affected in their neurological development** as a result of e-waste exposure. It was subsequently estimated that these **children in China lost about 97,560 IQ points** (69,600-111,600) as a result of informal e-waste recycling and dumping activities. This amounts to an **average reduction of intelligence of 1.2 points per child**. It should be noted that these figures provide an initial order of magnitude estimate. More accurate estimates might be generated on the basis of more detailed data on among others the number of children exposed to e-waste or on blood lead levels in children and the contribution of e-waste exposure to these blood lead levels, if those data could be retrieved.

Similar quantifications could be done for other pollutants, for other health impacts and for other groups of people (children vs. adults, e-waste workers vs. inhabitants, inhabitants of e-waste recycling towns vs. inhabitants of adjacent regions). Neither have health impacts in the longer term been quantified. As a matter of fact, some longer term risks may be yet to develop and will still need to be understood.

As to the economic impacts of illegal exports of e-waste, it is estimated that the 2.98 million tonnes of illegally exported e-waste from the EU in 2012 correspond roughly with **€ 31.2 million to € 37.5 million loss in profits to the EU e-waste recycling industry**. This figure relates to the overall EU exports. If one looks at the e-waste **exports to China only** (1.16 million tonnes in 2012), the **EU recycling industry** is estimated to have **lost € 12.2 million to € 14.6 million in profits in 2012**. These figures should be considered initial order of magnitude estimates, as they were generated on the basis of a simple and rough extrapolation of an estimate for the UK only to Europe as a whole. Thereby the assumption was made that the ratio between the volumes of illegal exports and the loss to the industry is the same for the UK and the EU as a whole.

Assuming that the average intrinsic value of WEEE is about € 300 per tonne, the **economic value lost to the EU** as a result of illegal **exports to China** is roughly estimated at **€ 348 million for 2012** only. The **economic value lost** to the EU as a result of **all illegal exports out of the EU** is estimated at **€ 892 million for 2012**. This estimate too is an initial order of magnitude estimate and may be further fine-tuned on the basis of a revised average intrinsic value of WEEE. The value of € 300 per tonne refers to the average value of WEEE, whereas more precise figures might be generated for those WEEE categories that are being shipped to and recycled in China.

As to the impact on jobs, the illegal export of e-waste from the EU in 2012 is estimated to represent a **potential loss** of about **38,000 FTE recycling jobs in the EU.** Assuming a typical multiplier of 2, these direct recycling jobs would result in another **38,000 indirect and induced jobs**, for a total of **76,000 jobs.** The **illegal export to China** in particular is estimated to represent a potential loss of circa **14,900 FTE jobs** and another **14,900 indirect and induced jobs**, for a total of **29,800 jobs.** Our findings provide an initial order of magnitude estimate of the potential loss of recycling jobs and indirect and induced jobs in the EU.

5 Monetary analysis

5.1 Environmental impacts

The illegal export of e-waste is indirectly (through the informal recycling and dumping) leading to severe impacts on the environment, compromising among others the river and ground water quality and as a result depriving inhabitants from clean access to drinking water. Different estimates have shown that lead levels in water have been 8 times higher than the local drinking water standard (0.05 mg/L) (Robinson 2009). The two main rivers flowing through Guiyu, the Liangjiang and the Nanyang, are or used to be directly exposed to different parts of the e-waste processing chain. For example, the Nanyang had a number of acid leaching sites on its banks. The water and sediments in the rivers have been found to contain high levels of heavy metals (Greenpeace 2008). In Guiyu, the presence of pollutants (such as heavy metals) in surface and groundwater has created many years ago a market for drinking water to be delivered from neighbouring towns (Hicks et al. 2005). This resulted in direct costs for Guiyi's inhabitants, as they were required to buy expensive drinking water instead of using water from the river or from ground water resources. In the meantime tap water pipelines have been constructed in Guiyu giving inhabitants access to clean drinking water. More recently, the local government in Guiyu has started with soil remediation, due to the presence of heavy metals and other chemicals in the soil originating from e-waste recycling (Personal Communication 2015).

The impacts of e-waste exports/recycling on the aquatic environment can thus be monetised by estimating/calculating the (actual) direct costs for buying drinking water, by calculating the (actual) costs for constructing water supply infrastructure (drinking water pipelines) and by calculating the (potential and/or actual) costs for remediating contaminated soil in Guiyu. The additional cost that need to be incurred to mitigate the negative impacts of contaminated soil and water resources are site specific. Data from the WHO (2004) on the cost of water supply improvement can help illustrate the magnitude of these costs. According to this estimate, a supply based on piped water approximately amounts to annual cost of US$9.95[19] (approximately €10.77)[20] per person supplied (WHO 2004). This is a broad estimate for the Asian region and includes investment costs and the cost of operation and maintenance over the economic life of piped water infrastructure of 40 years. Accordingly, **the construction of a piped water supply for Guiyu from 2005 onwards would have led to additional annual cost of US$1,333,300 ($9,95 x 134,000) (approximately €1,443,590)**[21] i.e. the estimated population in 2005. However, if water resources planners accounted for the expected population growth in Guiyu, the supply systems would have to be dimensioned accordingly. For example, **supplying water to the estimated population in 2013 would lead to higher expected additional annual cost of US$1,492,500 (US$9.95 x 150,000) (approximately €1,615,959**)[22].

[19] This refers to USD year 2000.

[20] The Eurostat's annual average exchange rates for 2000 have been used to convert USD to Euros (available at: http://appsso.eurostat.ec.europa.eu/nui/show.do?dataset=ert_bil_eur_a&lang=en).

[21] Given that the original estimate refers to USD in year 2000 the Eurostat's annual average exchange rates for 2000 have been used to convert USD to Euros (available at: http://appsso.eurostat.ec.europa.eu/nui/show.do?dataset=ert_bil_eur_a&lang=en).

[22] Given that the original estimate refers to USD in year 2000 the Eurostat's annual average exchange rates for 2000 have been used to convert USD to Euros (available at: http://appsso.eurostat.ec.europa.eu/nui/show.do?dataset=ert_bil_eur_a&lang=en).

5.2 Health impacts

Some of the health impacts in China arising from illegal e-waste shipments (and informal recycling and dumping in particular) have direct economic costs and others can usefully be represented by economic or monetary values to help communicate the importance of preventive and remedial action. In section 4.3 the impacts of lead exposure on children's neurological development in terms of IQ change have been quantified. It was estimated that around 81,300 children (58,000-93,000), born in 1995-2013 in the Chinese informal e-waste recycling areas, have been affected in their neurological development as a result of the informal recycling of e-waste (see paragraph 4.3.4). It was subsequently estimated that the intelligence of these children had collectively dropped with **97,560 IQ points** (69,600-111,600), which amounts to an **average loss of intelligence of 1.2 points per child** due to lead exposure originating from informal e-waste recycling and dumping (see paragraph 4.3.4).

A monetary valuation of these impacts on children's IQ might include an assessment of: opportunity costs in terms of *lost productivity* (i.e. decreased current value of expected lifetime revenues); *direct resource educational costs* related with compensatory education; opportunity costs of *lost income during remedial compensatory education*; *medical treatment costs*; and, *disutility* resulting from human development disabilities (Hunt 2011).

Given time and resource constraints and limited availability of or access to Chinese data, it was not possible within the context of this project to estimate or calculate these direct costs and opportunity costs. Figures from other studies however have been identified which may give an indication of the scale of these costs. Schwartz (1994) for instance related a 1 point reduction in IQ to a 4.5 per cent increase in the risk of failure to graduate from high school. Grosse et al. (2002) studied economic benefits from projected improvements in worker productivity from the reduction in children's exposure to lead in the US and estimated that each IQ point raises worker productivity by 1.76 to 2.38 per cent using a causal model of cognitive ability and economic productivity and they estimated from there an economic benefit. Therefore, a **decrease of 1 IQ point** in children can be associated with a **decrease of later productivity** of about **2 per cent** (EFSA 2010). Applying this to our case, it can be estimated that **about 81,300 children** (58,000-93,000) will suffer on average a **decrease of later productivity of about 2.4 per cent** as a result of exposure to contamination from e-waste recycling.

Hunt (2011) identified two studies which estimated the **medical treatment costs** for children with elevated blood lead levels (the treatment was chelation therapy)[23] (see Table 17). In order to estimate the medical treatment costs in China, it might be sensible to start working from a range of values that correspond to the lowest and highest values given in the Mathtec Inc study: USD 428 and 4,400 per child respectively (2000 prices). These figures obviously need to be adapted to the situation in China if one wants to use these for estimating the medical treatment costs. It also needs to be figured out to what extent and which medical treatment lead-poisoned children get in China.

Table 17: Estimates of medical treatment costs incurred by lead-poisoned children

Study	Cost element	Impact valued	Cost per child (USD, 2000)
US EPA (1985)	Medical costs	Preventing blood levels rising to 25 µg/dl or above	1531
Mahtec Inc (1987)	Medical costs; screening/education programmes; opportunity costs of	Blood level > 40 µ/dl; EP level > 53 µg/dl	4398
		Blood level > 40 µ/dl; EP level 35-53 µg/dl	2196

[23] Chelation therapy is a medical procedure with the aim to remove heavy metals from the body.

parents time	Blood level 21-40 µ/dl; EP level 33-53 µg/dl	1016
	Blood level 21-40 µ/dl; EP level 0-32 µg/dl	428
	Blood level 0-20 µ/dl; EP level > 33 µg/dl	565
	Blood level 0-20 µ/dl; EP level 0-32 µg/dl	428

Source: adapted from Hunt 2011

Scasny et al. (2008) provide a review of the available evidence of the **opportunity costs** next to the **costs** accrued by **remedial education**. They started from the guidance provided by the US Environmental Protection Agency (US EPA 1997) and arrived at cost estimates as presented in Table 18. The US EPA study combines the value of lifetime revenue with the estimate of percent salary loss per IQ point and subtracts the direct education and opportunity costs to result in a total net impact of IQ loss on revenue of USD 2,505 or USD 3,410 per IQ point (depending on which estimates of percent salary loss per IQ point are being applied) (US EPA 1997).

Table 18: Loss in earnings and education costs from IQ loss

	USD
Loss in earnings (LE)	4,090
Costs of education (CE)	267
Opportunity costs while in school (OC)	531
Total (LE-CE-OC)	3,292

Source: adapted from Hunt 2011 and Scasny et al. 2008

Based on a literature review on IQ valuation Spadaro and Rabl (2004; 2008) arrived at a **unit value of USD 18,000 per IQ point** (including adjustment for purchase power parity).

5.3 Economic impacts

5.3.1 Economic loss to the EU e-waste recycling sector

In fact, the economic loss to the EU e-waste recycling sector had already been monetised in paragraph 4.4.1 by extrapolating a figure for the UK to the EU as a whole. As a result it was roughly estimated that **2.98** million tonnes of illegally exported e-waste from the EU in 2012 represent roughly **£25.3 million to £30.4 million loss in profits to the EU recycling industry (€ 31.2 million – € 37.5 million)**[24]. If one looks at the illegal e-waste **exports to China only** (**1.16** million tonnes in 2012), the **EU recycling industry** might have **lost £9.86 million to £11.83 million in profits in 2012 (€ 12.2 million - € 14.6 million**). To put this into perspective, the EU e-waste recycling market earned revenues of US$1.3 billion in 2012 (or € 1.012 billion) and is estimated to earn US$1.79 billion (or € 1.39 billion) in 2020 (Frost & Sullivan 2013) (see paragraph 4.4.1 for more details).

Building further on an estimate from the WEEE Forum (2013) of the average intrinsic value of one tonne WEEE (about € 300), the **economic value lost** to the EU as a result of illegal exports to **China** (1.16 million

[24] The average historical exchange rates for 2012 have been used to convert from USD and Pound to euro (http://www.oanda.com/currency/historical-rates).

tonnes in 2012) was estimated at **€ 348 million for 2012** (see paragraph 4.4.1). The economic value lost to the EU as a result of all illegal **exports out of the EU** (2.98 million tonnes in 2012) is estimated at **€ 892 million for 2012**.

The economic value lost to the EU arising from illegal e-waste exports might also be estimated through a bottom-up approach whereby the monetary values of the different components and/or materials (including the precious metals) contained within these exports is estimated and these values are aggregated. Or at least the value of some precious materials lost might be assessed (e.g. gold, silver and palladium).

The **high intrinsic value** of the **precious metals** contained in e-waste can be demonstrated by following figures: according to Johnson (2011) the average mobile phone and DVD player contain around £15 (or €17.19) and £28 (or €32.09) of precious metals respectively and overall WEEE contains precious metals worth around £1 (or €1.146) for every kilogram of discarded electronic equipment (if September 2011 values are being applied) (Johnson 2011). Using the September 2011 values for these precious metals, the **value of precious metals within WEEE exported from the EU** in 2011 is roughly estimated at about **€ 3.23 billion** for all exports and about **€ 1.26 billion for exports to China only** (assuming 2.82 million tonnes of e-waste were exported illegally from the EU in 2011 and 1.1 million tonnes to China only, see table in Annex B: Guiyu population statistics). It should be noted that the accuracy of these estimates is difficult to assess with any degree of certainty. These estimates are however reliable as 'order of magnitude' estimates.

5.3.2 Job loss in the EU e-waste recycling sector

In paragraph 4.4.1 it was assessed that the illegal export of e-waste from the EU in 2012 might represent a potential loss of about 38,000 full time equivalent (FET) direct recycling jobs in the EU and another 38,000 indirect and induced jobs, for a total of 76,000 jobs. The illegal export from the EU to China in particular was estimated to represent a potential loss of about 14,900 FTE jobs and another 14,900 indirect jobs, for a total of 29,800 jobs. In economic terms, the loss of jobs implies the loss of economic value that could be generated in the e-waste recycling sector in Europe. Data limitations, especially in terms of separate statistical data for e-waste recycling (as a subsector of within the waste collection, treatment and disposal sector (including materials recovery) only allow rough assessments of these economic losses. According to Eurostat figures, the average annual economic value added per employee amounted to € 52,344 for the EU-27's waste and materials recovery sector (NACE Division 38) (Eurostat 2015). Thus, **direct employment losses of 14,900 FTE in the e-waste recycling sector imply an annual loss of economic value added of around € 780,000,000** (€ 52,344 per FTE employee x 14,900 FTE = 779,930,952). However, these values need to be treated with caution, as the e-waste recycling sector will differ in its characteristics form the aggregate waste collection, treatment and disposal sector. Nonetheless, the figures are indicative of considerable economic impacts.

5.4 Conclusions

The **environmental impacts** in China arising from illegal e-waste imports from the EU have not been monetised in full. We have illustrated these impacts by focusing on the environmental impacts of informal e-waste recycling and dumping on the aquatic environment in Guiyi town. These impacts have been illustrated by estimating the (actual) direct costs for constructing water supply infrastructure (drinking water pipelines) in order to substitute locally contaminated water supply resources in Guiyi. **To supply the estimated current population in Guiyu, the additional annual cost amount to around**

US$1,500,000 (approximately €1,630,000)[25]. This is a conservative and lower-bound value, as the supply infrastructure would likely to take into account further population growth in the area.

As to the **health impacts**, we have undertaken an effort to monetise the loss of intelligence in the group of children that were born in the Chinese informal e-waste recycling areas in the period 1995-2013. A monetary valuation of these impacts on children's IQ might have included an assessment of: opportunity costs in terms of *lost productivity* (i.e. decreased current value of expected lifetime revenues); *direct resource educational costs* related with compensatory education; opportunity costs of *lost income during remedial compensatory education*; *medical treatment costs*; and, *disutility* resulting from human development disabilities. However, given time and resource constraints and limited availability of or access to Chinese data, it was not possible within this study to estimate or calculate these costs. In order to estimate the medical treatment costs in China for instance, it is suggested to start working from a range of values that correspond to the lowest and highest values given in a study done in the US: USD 428 and 4,400 per child respectively (2000 prices). These figures obviously need to be adapted to the situation in China if one wants to use these for estimating the medical treatment costs. It also needs to be figured out to what extent and which medical treatment lead-poisoned children get in China.

As to the **economic impacts**, the loss in profits for the EU e-waste recycling sector arising from illegal e-waste exports had already been monetised earlier in the report, by extrapolating a figure for the UK to the EU as a whole. The economic value lost to the EU had also been monetised earlier, thereby building further on an estimate of the average intrinsic value of one tonne WEEE. In addition, the **value of precious metals** within WEEE exported from the EU was estimated for 2011, building further on 2011 information indicating that every kilogramme of WEEE overall contains precious metals worth around € 1.146. As a result this value was estimated to be about **€ 3.23 billion** for **all exports** from the EU and about **€ 1.26 billion** for **exports to China** only. It should be noted however that the accuracy of these estimates is difficult to assess with any degree of certainty. These estimates are however reliable as 'order of magnitude' estimates.

The **direct job loss** for the EU e-waste recycling industry has been monetised by multiplying the number of jobs lost with average economic value added per FTE employee in the sector. **It amounts to an annual loss of around € 780 million of value added for the EU-27.**

[25] Given that the original estimate refers to USD in year 2000 the Eurostat's annual average exchange rates for 2000 have been used to convert USD to Euros (available at: http://appsso.eurostat.ec.europa.eu/nui/show.do?dataset=ert_bil_eur_a&lang=en).

6 Conclusions

6.1 Summary of the extent of impacts

6.1.1 Illegal exports of e-waste from the EU

It was estimated that for 2005 and 2012 respectively around 0.74 and 1.16 million tonnes of e-waste have been imported in China from the EU.

6.1.2 Environmental impacts

The illegal export of e-waste from the EU to China has resulted – through the intensive informal recycling of e-waste in China – in the release of large amounts of contaminants in the local environment such as heavy metals, PBDEs, PCDD/Fs, PBCs, CFCs and PAHs. It has caused among others high concentrations of heavy metals such as lead, cadmium, mercury, copper and zinc in the surrounding air, dust, soils, sediments and plants.

The potential annual emissions of some environmental contaminants associated with e-waste imported illegally in China from the EU were estimated. It was for instance estimated that respectively 10 and 16 tonnes of PCBs from EU e-waste were potentially released in the Chinese environment in 2005 and 2012.

Given the complexity of the e-waste problem, the report presents only select data as to the environmental impacts of the e-waste crime. The focus was mostly on the environmental impacts (i.e. pollution levels) related to heavy metals and lead and zinc in particular. And mostly studies investigating or results regarding adverse environmental impacts in Guiyi and Taizhou, as the most representative locations for informal e-waste recycling, are being referred to.

As to the monetisation of the **environmental impacts** in China, we have focused on the impacts of informal e-waste recycling and dumping on the aquatic environment in Guiyi town. These impacts have been monetised by estimating the (actual) direct costs for constructing water supply infrastructure (drinking water pipelines) in Guiyu. **To supply the estimated current population in Guiyu, the additional annual cost amount to around US$1,500,000 (approximately €1,630,000)**[26]. This is a conservative and lower-bound value, as the supply infrastructure would likely to take into account further population growth in the area.

6.1.3 Health impacts

As to health impacts, this study demonstrates that illegal exports from the EU are significantly increasing incidences of chronic disease in China, threatening not just workers but also current residents of e-waste recycling areas and adjacent regions and future generations. Illegal exports from the EU result (through the informal recycling and dumping) in high prevalence of skin, gastric, respiratory, hematic, neurological, prenatal, natal and infant diseases in China.

In order to demonstrate the health impacts, figures on the routes of exposure, on the human body burden and on scientifically established associations and/or causal links between e-waste exposure and several kinds of diseases and other health impacts are presented. As with environmental effects, only select figures were presented, primarily in relation to heavy metal contamination in Guiyu and Taizhou.

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[26] Given that the original estimate refers to USD in year 2000 the Eurostat's annual average exchange rates for 2000 have been used to convert USD to Euros (available at: http://appsso.eurostat.ec.europa.eu/nui/show.do?dataset=ert_bil_eur_a&lang=en).

Select scientific studies (in China) show associations between exposure to e-waste and physical health outcomes such as:

- decreased lung function (i.e. lower forced vital capacity);
- decreased physical growth of children (i.e. lower weight, height and body-mass index);
- reduced reproductive health (i.e. increases in spontaneous abortions, stillbirths, and premature births, and reduced birth weights and birth lengths);
- changes in cellular expression and function (i.e. increased DNA damage).

Negative associations were also shown for blood lead levels and IQ in children.

A more detailed quantification has been carried out for the impacts of lead poisoning resulting from e-waste exposure on children's neurological development, expressed in terms of children's IQ scores. Based on data from various studies it was concluded that almost all children living in informal e-waste recycling centers such as Guiyu and Taizhou are or have been subject to lead exposure which exceeds the $BMDL_{01}$ intake level of 0.50 μg/kg b.w. per day as determined by the EFSA Panel on Contaminants in the Food Chain for neurodevelopmental effects.

This implies that potentially all children living in these informal e-waste recycling areas have been affected in their neurological development. It is estimated that roughly **22,400 children** (16,000-25,600) born **in Guiyu** in the period **1995-2013** have had blood lead levels exceeding the $BMDL_{01}$ and as a result have been subject to a **drop in intelligence** as a result from informal e-waste recycling and dumping.

For **China** as a whole it is conservatively estimated that **around 81,300 children (58,000-93,000) born in the period 1995-2013** have been **affected in their neurological development** as a result of e-waste exposure.

It was subsequently estimated that these **children in China lost about 97,560 IQ points** (69,600-111,600) as a result of informal e-waste recycling and dumping activities. This amounts to an **average reduction of intelligence of 1.2 points per child**.

As to the **monetisation** of the health impacts, we have undertaken an effort to monetise the loss of intelligence in the group of children that were born in the Chinese informal e-waste recycling areas in the period 1995-2013. Given time and resource constraints and limited availability of or access to Chinese data, it was not possible within the boundaries of this study to monetise this. In order to estimate the medical treatment costs in China for instance, it is suggested to start working from a range of values that correspond to the lowest and highest values given in a study done in the US: USD 428 and 4,400 per child respectively (2000 prices). These figures obviously need to be adapted to the situation in China if one wants to use these for estimating the medical treatment costs. It also needs to be figured out to what extent and which medical treatment lead-poisoned children get in China.

6.1.4 Economic impacts

As to the economic impacts of illegal exports of e-waste, it is estimated that the 2.98 million tonnes of illegally exported e-waste from the EU in 2012 correspond roughly with **€ 31.2 million to € 37.5 million loss in profits to the EU e-waste recycling industry**. This figure relates to the overall EU exports.

If one looks at the e-waste **exports to China only** (1.16 million tonnes in 2012), the **EU recycling industry** is estimated to have **lost € 12.2 million to € 14.6 million in profits in 2012**.

Assuming that the average intrinsic value of WEEE is about € 300 per tonne, the **economic value lost to the EU** as a result of illegal exports to China is roughly estimated at **€ 348 million** for **2012** only.

The economic value lost to the EU as a result of all illegal exports out of the EU is estimated at **€ 892 million** for **2012**.

In addition, the **value of precious metals** within WEEE exported from the EU was estimated for 2011, building further on 2011 information indicating that every kilogramme of WEEE overall contains precious metals worth around € 1.146. As a result this value was estimated to be about **€ 3.23 billion** for **all exports** from the EU and about **€ 1.26 billion** for **exports to China** only.

As to the impact on jobs, the illegal export of e-waste from the EU in 2012 is estimated to represent a **potential loss** of about **38,000 FTE recycling jobs in the EU**. Assuming a typical multiplier of 2, these direct recycling jobs would result in another **38,000 indirect and induced jobs**, for a total of **76,000 jobs**. The **illegal export to China** in particular is estimated to represent a potential loss of circa **14,900 FTE jobs** and another **14,900 indirect and induced jobs**, for a total of **29,800 jobs.**

Table 19: Overview of estimated economic impacts in the EU for 2012

Loss in profits for the EU recycling industry	Arising from illegal EU exports to China		€ 12.2m - € 14.6m	
	Arising from total illegal EU exports		€ 31.2m - € 37.5m	
Lost economic value to the EU	Arising from illegal EU exports to China		€ 348m	
	Arising from total illegal EU exports		€ 892m	
Potential job loss in the EU (FTE)	Arising from illegal EU exports to China	*Direct jobs*	14,900	29,800
		Indirect and induced jobs	14,900	
	Arising from total illegal EU exports	*Direct jobs*	38,000	76,000
		Indirect and induced jobs	38,000	

The **direct job loss** for the EU e-waste recycling industry has been monetised by multiplying the number of jobs lost with the annual wage in the EU waste recycling (and reuse) industry. It amounts to annual losses of around € 780 million of value added in this sector.

Finally, it should be noted that most of our quantification and monetisation efforts in this report provide initial order of magnitude estimates.

6.2 Major gaps and research needs

6.2.1 Quantitative analysis

Quantifying the illegal export of e-waste from the EU (to China) is highly challenging as there is very little clear information upon which estimates can be based (given the illegal nature of these e-waste shipments). The estimates are based on reliable estimates of the volumes of e-waste generated in the EU and assumptions about the percentage of e-waste that is being exported from the EU and the percentage of e-waste from the EU that is being imported in China. Especially in relation to these percentages more in-depth research could be undertaken. More research could also be undertaken into the (sub)volumes of the different e-waste categories that are being shipped out of the EU and imported in China.

Given the uncertainty surrounding these 'guesstimates', it is suggested to work with minimum and maximum EU export scenarios and minimum and maximum import in China scenarios.

The potential annual emissions of some environmental contaminants associated with e-waste imported illegally in China from the EU have been estimated. These estimates are based on concentrations in components of Swiss e-waste and on the estimates of the 2005 and 2012 illegal imports in China from the EU. Further research could be undertaken to update the estimates of concentrations of certain environmental contaminants of e-waste and to extend these estimates to other contaminants – figures are only available for a select number of contaminants.

Given the scale and complexity of the e-waste problem, it was within the context of this project not possible to quantify the total environmental impacts from informal e-waste recycling in China or in particular areas of China, not to mention the impacts from the informal recycling in China of the millions of tonnes of e-waste imported annually from the EU (i.e. the share of EU-borne e-waste impacts within the overall impacts from informal recycling in China). This resulted among others from uncertainties about the sources of the contamination in recycling areas and control areas and in particular from the exact shares of these sources in the overall contamination. It should however be noted that many studies were able to indicate that certain pollution cases were mainly originating from e-waste recycling activities.

As a result, the report presents only select data as to the environmental impacts of the e-waste crime. The focus was mostly on the environmental impacts (i.e. pollution levels) related to heavy metals. And mostly studies investigating or results regarding adverse environmental impacts in Guiyi and Taizhou, as the most representative locations for informal e-waste recycling, are being referred to.

Also in relation to the health impacts select figures were presented, primarily in relation to heavy metal contamination in Guiyu and Taizhou. Figures on the routes of exposure, on the human body burden and on scientifically established associations and/or causal links between e-waste exposure and several kinds of diseases and other health impacts are presented.

As to the estimates of the number of children subject to a drop in intelligence and of the total loss of intelligence, it should be noted that these provide an initial order of magnitude estimate. More accurate estimates might be generated on the basis of more detailed data on among others the number of children exposed to e-waste or more detailed data on blood lead levels in children and the contribution of e-waste exposure to these blood lead levels, if those data could be retrieved.

Similar quantifications could be done for other pollutants, for other health impacts and for other groups of people (children vs. adults, e-waste workers vs. inhabitants, inhabitants of e-waste recycling towns vs. inhabitants of adjacent regions).

Neither have health impacts in the longer term been quantified. As a matter of fact, some longer term risks may be yet to develop and will still need to be understood.

The estimates of the loss in profits for the EU e-waste recycling industry should be considered initial order of magnitude estimates, as they were generated on the basis of a simple and rough extrapolation of an estimate for the UK only to Europe as a whole. Thereby the assumption was made that the ratio between the volumes of illegal exports and the loss to the industry is the same for the UK and the EU as a whole. These estimates of the loss in profits could be further fine-tuned by applying the methodology used to generate the estimate for the UK to the EU as a whole. This obviously requires more detailed research into the economic parameters of the EU e-waste recycling industry.

The estimate of the total economic value lost to the EU (based on the assumed average intrinsic value of WEEE per tonne) too is an initial order of magnitude estimate and may be further fine-tuned on the basis of a revised average intrinsic value of WEEE. The value of € 300 per tonne refers to the average value of WEEE, whereas more precise figures might be generated for those WEEE categories that are being shipped to and recycled in China.

6.2.2 Monetary analysis

In this study the **environmental impacts** in China arising from illegal e-waste imports from the EU have not been monetised in full. We have focused on the impacts of informal e-waste recycling and dumping on the aquatic environment in Guiyi town, by estimating the costs for constructing water supply infrastructure (drinking water pipelines) in Guiyu. In future research, attempts could be made to monetise these impacts on the aquatic environment for China as a whole.

As to the monetisation of the **health impacts**, we have undertaken an effort to monetise the loss of intelligence as a result of lead poisoning in the group of children that were born in the Chinese informal e-waste recycling areas in the period 1995-2013. An attempt was undertaken to estimate the medical treatment costs in China. However, only a range of values was identified from studies in the US. Future research might be carried out to investigate the actual costs of medical treatment in China and to investigate to what extent lead-poisoned children get medical treatment in China and if so what type of treatment. Future research might also gather data to monetise other costs than medical treatment costs, such as opportunity costs in terms of *lost productivity* (i.e. decreased current value of expected lifetime revenues); *direct resource educational costs* related with compensatory education; opportunity costs of *lost income during remedial compensatory education*; and, *disutility* resulting from human development disabilities. Future research may also look into monetising health impacts from other environmental contaminants than lead, from overall e-waste exposure or from certain exposure routes. For instance the monetisation of health impacts from air pollution resulting from informal e-waste recycling could be looked into further.

As to a review of gaps in the analysis and the research needs in relation to monetising economic impacts, we refer to the previous section – i.e. the section on the quantitative analysis. As indicated earlier in the report, the quantitative analysis of the economic impacts (in particular the loss of profits to the EU e-waste recycling industry and the lost economic value to the EU) was in fact a monetary analysis.

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Annex A: Chinese soil quality standards

The Chinese soil quality standards, also called GB1568-1995 standards, were put into effect in 1996 with the main aim to prevent soil pollution and to protect soil functions, eco-environment, agricultural and forestry production and human health.

The soil quality standards include the following three classifications (Wang G. & Shan Y. 2013):

1. Class 1 standards:
    - Soil background level
    - Nationally wide background values based
    - Cover natural conservation areas, drinking water source areas etc.
    - Based on more than 400 samples
2. Class II standards:
    - Ecological and environmental effects based
    - Cover farm land, fields of vegetable and tea production, orchard soil
    - Healthy plant growth and safe food quality
    - No potential effects on water bodies
3. Class 3 standards:
    - Soil of higher adsorption capacity/background levels
    - Healthy growth of trees/plants, no hazard to environment
    - Use experimental data based on soil of higher adsorption capacity and artificially contaminated soils

Table 20: The GB1568-1995 Chinese soil quality standards

Contaminants	Class-1	Class-2			Class-3
	Background	<6.5	6.5 - 7.5	>7.5	>6.5
Cd	0.20	0.30	0.30	0.60	1.0
Hg	0.15	0.30	0.50	1.0	1.5
Ni	40	40	50	60	200
As Paddy	15	30	25	20	30
As Dry land	15	40	30	25	40
Cu [a] Agri.	35	50	100	100	400
Cu [a] Frui.	-	150	200	200	400
Pb	35	250	300	350	500
Cr [b] Paddy	90	250	300	350	400
Cr [b] Dry land	90	150	200	250	300
Zn	100	200	250	300	500
HCH [c]	0.05		0.50		1.0
DDT [c]	0.05		0.50		1.0

[a]: 'Agri.' represents agricultural soils, and 'Frui.' represents fruit farm soils.
[b]: In case soil CEC < 5cmol(+) kg^{-1}, the standard values will be half values of the listed.
[c]: HCH (hexachlorocyclohexane), values are the sum of 4 isomers;
[d]. DDT (Dichloro-diphenyl-trichloroethane), values represent the sum of DDT, DDD and DDE.

Source: Wang G. & Shan Y. 2013

Annex B: Guiyu population statistics

Table 21: estimates of inhabitants and number of children born in Guiyu

Year	Estimated number of habitants Guiyu	Crude birth rates for China (per 1000 hab.)[a]	Children born in Guiyu	Crude death rates for China (per 1000 hab.)[a]	People dying in Guiyu	Population proportion under 15 [b]	Population under 15
1995	95,000	17	1,615	7	665	29	27,550
1996	99,000	17	1,683	7	693	28	27,720
1997	103,000	17	1,751	7	721	28	28,840
1998	107,000	16	1,712	7	749	27	28,890
1999	112,000	15	1,680	6	672	26	29,120
2000	116,000	14	1,624	6	696	26	30,160
2001	120,000	13	1,560	6	720	25	30,000
2002	125,000	13	1,625	6	750	24	30,000
2003	**130,000**	12	1,560	6	780	22	28,600
2004	132,000	12	1,584	6	792	21	27,720
2005	134,000	12	1,608	7	938	20	26,800
2006	136,000	12	1,632	7	952	20	27,200
2007	138,000	12	1,656	7	966	19	26,220
2008	140,000	12	1,680	7	980	19	26,600
2009	142,000	12	1,704	7	994	19	26,980
2010	144,000	12	1,728	7	1,008	18	25,920
2011	146,000	12	1,752	7	1,022	18	26,280
2012	148,000	12	1,776	7	1,036	18	26,640
2013	**150,000**	12	1,800	7	1,050	18	27,000
Total			31730		16184		
Average	127,211		1670				27,802

Source: own representation, World Bank, WHO Global Health Observatory Data Repository and Wang et al. 2013. Note: As no population statistics were available for Guiyu to the project team, except for the number of habitants in 2003 and 2013 (Wang et al. 2013), the number of habitants for the other years in the period 1995-2013 have been estimated/calculated starting from these 2003 and 2013 figures, thereby applying urban population growth rates for the period 1995-2013 (assuming Guiyu has been growing very fast in that period) and applying roughly the population growth rates in China as a whole for the period 2003-2013 (assuming that population growth had slowed down somewhat in that period). The number of children born annually in Guyiu have been estimated/calculated by applying the crude birth rates from China (per 1000 habitants) (figures from The World Bank) to the population estimates. [a] Source: The World Bank; [b] Source: WHO Global Health Observatory Data Repository.

Table 22: overall and urban population growth rates in China

Year	Population growth rates China	Urban population growth rates China
1995	1,1	4,2
1996	1	4,1
1997	1	4
1998	1	3,9
1999	0,9	3,8
2000	0,8	3,6
2001	0,7	4,1
2002	0,7	4,2
2003	0,6	4,1
2004	0,6	4
2005	0,6	3,9
2006	0,5	3,7
2007	0,5	3,5
2008	0,5	3,4
2009	0,5	3,3
2010	0,5	3,3
2011	0,5	3,2
2012	0,5	3,1
2013	0,5	2,9

Source: own representation and The World Bank

Evaluation of the Costs and Impacts of Environmental Crime: CITES Trade of the Horsfieldii Tortoise

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LIST OF ABBREVIATIONS

CITES	Convention on International Trade in Endangered Species
IUCN	International Union for the Conservation of Nature
VU	Vulnerable

1 Introduction

1.1 Background Information on Horsfieldii

Testudo horsfieldii is a tortoise and is classified as Vulnerable (VU)[1] on the International Union for the Conservation of Nature (IUCN) Red List following an evaluation of its conservation status in the wild by the Tortoise and Fresh Water Turtle Specialist Group in 1996. For this reason the horsfieldii was categorized to Appendix II of Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) allowing for monitored trade of the species based on quotas self-designated by Parties to the Convention in a manner meant to sustain the species in the wild in its natural range.[2] The horsfieldii is native across Central Asia and can be found in Armenia, Iran, Afghanistan, Pakistan, northwest China, Kazakhstan, Kyrgyzstan, Tajikistan, Turkmenistan and Uzbekistan.[3]

Since the mid 1970s the horsfieldii has been subject to heavy trading for the global pet market.[4] Horsfieldii are either caught in the wild or bred in captivity and exported from its range into the pet trade in predominantly Western countries. The US, Japan and Europe are the main importers of the species.[5] While there are many turtle and tortoise species involved in the pet trade, the horsfieldii is one of the most heavily traded[6] and illegal trade is suspected to take place. Firstly, it is likely that more specimens are traded than are actually reported in CITES trade data.[7] Secondly, it is suspected that the improper use of CITES labels that differentiate between wild and captive bred specimens results in a much higher number of wild caught specimens existing in trade than the data reported would suggest.[8] Finally, tortoise is naturally found in a very vast range which includes countries that are not party to CITES (e.g. Tajikistan),

[1] To further understand the IUCN definitions and terminology used to evaluate a species conservation status see: http://www.iucnredlist.org/static/categories_criteria_2_3

[2] IUCN, "The Horsfieldii Trade - Wildlife Conservation Society," *IUCN Red List of Threatened Species*, 2014, http://www.wcs.org/conservation-challenges/natural-resource-use/hunting-and-wildlife-trade/the-Horsfieldii-trade.aspx.

[3] Katrina Smith and David Lee, "Testudostan: Our Post-Cold War Global Exploitation of a Noble Tortoise," *Bull. Chicago Herp. Soc* 45, no. 1 (2010): 1–9.

[4] European Commission, *Analysis of the Impact of EU Decisions on Trade Patterns. Report 3: Shifts in Sources of Specimens and Purposes of Trade* (Cambridge, 2014).

[5] Smith and Lee, "Testudostan: Our Post-Cold War Global Exploitation of a Noble Tortoise."

[6] UNEP-WCMC, *Review of Species Selected on the Basis of a New or Increased Export Quota in 2008*, 2008, http://ec.europa.eu/environment/cites/pdf/reports/increased_export_quotas_200.

[7] Smith and Lee, "Testudostan: Our Post-Cold War Global Exploitation of a Noble Tortoise."

[8] CITES, *Implementation of the Convention Relating to Capitve-Bred and Ranched Specimens (Decision 16.65)*, CITES, Twenty-Seventh Meting of the Animals Committee (Veracruz, Mexico, May 28, 2014).

which opens the possibility of illegal transport and smuggling through non-party countries or countries with less stringent environmental and enforcement standards.[9]

Despite that certain patterns in trade reveal potential illegality, it is difficult to calculate the rate of extinction of the species or its monetary value as a species because there is a serious lack of reliable data. This is a common reality for many species listed in CITES and therefore is not a justifiable reason for non-inclusion in such evaluations. Unlike the elephant and rhino that are also included in the CITES valuation study of the EFFACE project, the horsfieldii is an example of the omnipresent but invisible nature of illegal wildlife trade. This case is representative of many species that face the threat of unsustainable harvest but that are less emblematic and therefore risk drifting into population decline and possible extinction without notice. The commercial harvest of horsfieldii for the pet trade is regarded by conservation biologists as the foremost threat to its survival and existence in the wild.

For data, this study relied on publicly available trade data provided by CITES that indicate quotas and information on the quantity of exports and imports. Despite the many infrequencies that exist when using such data, it is possible to show how illegal trade may take place and at what scale. A significant challenge to understanding the impact of commercial trade on turtle and tortoise conservation status and that of the horsfieldii specifically, is the fact that there remains little data collection or information about their population status in the wild, which results from the fact that they are difficult to count in their vast range and also considered of lesser importance or reputation than some other flagship species (e.g. panda, rhino, elephant).[10] The data limitation on population and replacement rate was a significant obstacle in the calculation efforts of this study to determine the rate of extinction and sustainability of the current trade.

1.2 Main Threats

Since the 1970s, the main threat to the horsfieldii in the wild is the pet trade for which specimens are collected and exported from their range for commercial purposes.[11] The pet tortoise and turtle industry is global and threatens numerous different chelonian species. Of the 266 known turtle and tortoise species in the world, more than one third are facing extinction.[12] The Horsfieldii is one of many chelonian species that

[9] Tajikistan and Turkmenistan are not Parties and therefore do not report levels of trade and several other of the countries identified above do not provide self-reported data on exports nor do they set quotas, despite that they are likely involved in the trade as transit countries or collection countries.

[10] George Amato, Rob DeSalle, and Oliver A. Ryder, *Conservation Genetics in the Age of Genomics* (Columbia University Press, 2013); Stephanie A. Zimmer-Shaffer, Jeffrey T. Briggler, and Joshua J. Millspaugh, "Modeling the Effects of Commercial Harvest on Population Growth of River Turtles," *Chelonian Conservation and Biology* 13, no. 2 (December 1, 2014): 227–36, doi:10.2744/CCB-1109.1.

[11] P. Bergmann, "The Natural History of the Central Asian Tortoise," *The Cold Blooded News* 28, no. 10 (2001); M. Anderson-Cohen, "Russian Tortoise, Testudo Horsfieldii," *Tortuga Gazette* 30, no. 11 (1994): 1–4.

[12] Ted Williams, "The Terrible Turtle Trade," *National Audubon Society* 101, no. 2 (1999), http://nytts.org/asia/twilliams.htm.

are involved in the trade and are particularly unlikely to cope in the long term from commercial harvesting.[13]

For the pet trade, horsfieldii are either collected from the wild or bred/farmed in facilities using a stock-crop of tortoises collected from the wild. Currently, Uzbekistan is the main exporter of both wild caught and farmed tortoises, with Ukraine making up a smaller but not insignificant portion of the commercial trade.[14] As a species listed in Appendix II, the horsfieldii are legally allowed to be traded and the volume of trade is usually established by each Party (member state). The responsibility for establishing a sustainable quota, therefore, remains within the authority of each Party member. Accordingly, CITES requires that the Scientific Authority of member states, "must be satisfied and advise that the proposed export will not be detrimental to the survival of the species (the so called 'non-detriment finding in Article III, Paragraph 2 (a), and Article IV, paragraph 2 (a)., of the convention)."[15] Thus it is also important to have an independent reference of the conservation status of species other than the Scientific Authority of Parties.

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[13] Smith and Lee, "Testudostan: Our Post-Cold War Global Exploitation of a Noble Tortoise."

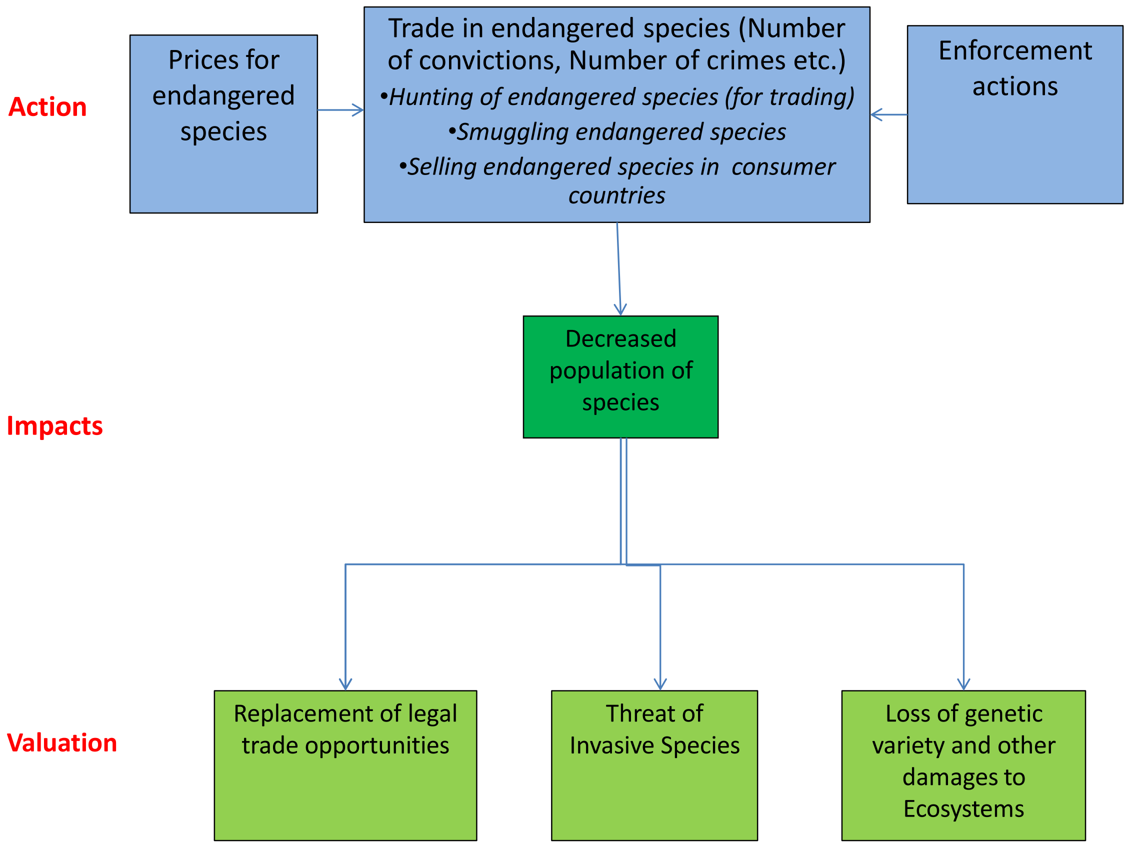
[14] European Commission, *Analysis of the Impact of EU Decisions on Trade Patterns. Report 3: Shifts in Sources of Specimens and Purposes of Trade*.

[15] CITES, "The CITES Export Quotas," 2015, http://www.cites.org/eng/resources/quotas/index.php.

2 Methodology

The following methodology sets out to explain how the impacts of illegal trade in certain CITES protected species could be assessed and which data sources and methods will be deployed to achieve thorough estimates for those impacts. The methodology is thereby based on the impact chain illustrated in Figure 1, which was developed in task one of WP3.

Figure 1: Impact chain



The methodology has two distinct steps:

- In the first step the impact of illegal trade on population numbers of the species protected by CITES is assessed based on existing statistics on quotas, trade and population.
- In the second step the economic and environmental impacts will be assessed. For some of the impacts valuations will be undertaken.

2.1 Estimating the Impact of Illegal Trade on Population Numbers

The most direct effect of illegal trade in endangered species is the impact it has on the conservation status of the species in its natural habitat and then the ripple effect this has on ecosystem services and global biodiversity. For many species, some level of data can be obtained on the conservation status of certain populations and the volume of trade.

For our analysis we have chosen the horsfieldii tortoise, where it is possible to use available information to make estimates about the proportion of illegal trade as compared to the legal trade. We identified three potential avenues of illegal trade.

1) It is likely that many more species are involved in the trade than are actually reported in CITES data.[16]

2) It is suspected that the improper use of CITES labels that differentiate between wild and captive bred specimens, results in a much higher number of wild caught specimens existing in trade than the data reported would suggest.[17] (See Figure 2)

3) Transport, smuggling and trade through countries that are not Party to CITES or countries with less stringent environmental and enforcement standards.[18]

In addition to illegal trade, this study also attempts to make the point that export quotas while legal, are not necessarily by any means sustainable. Thus, with the horsfieldii there is a parallel problem of a highly exploitative legal trade that operates within the rules and obligations of CITES but that nevertheless does not seem to accurately ensure the sustainable exploitation of the species.

For the horsfieldii, the available information on population and trade quotas is assessed to identify the causality between estimated illegal trade numbers and population figures. It is important to emphasize the fact that different datasets will have certain biases and are not complete. Therefore, we do not suggest to undertake a sophisticated statistical analysis, as the information on population developments is so poor that a broad range of statistics need to be used.

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[16] Smith and Lee, "Testudostan: Our Post-Cold War Global Exploitation of a Noble Tortoise."

[17] CITES, *Implementation of the Convention Relating to Captive-Bred and Ranched Specimens (Decision 16.65)*.

[18] Tajikistan and Turkmenistan are not Parties and therefore do not report levels of trade and several other of the countries identified above do not provide self-reported data on exports nor do they set quotas, despite that they are likely involved in the trade as transit countries or collection countries.

2.2 Estimating and Valuing Impacts of Decreasing Population Numbers

To value the population loss, an understanding of both the value of the species in the pet trade and its value within the local ecosystem is required, which makes this analysis limited by two important factors.

The value/price of the species on the market is artificially low due to the overexploitation and the low barriers to circumvent the legislation. Currently, source countries could not build a significant income in a legal way as prices for the animals are negligible.

It is also worth noting that the value of the species for the overall ecosystem cannot be valued properly. The horsfieldii tortoise is not a keystone species and its particular existence on the Central Asian steppes is not well understood in terms of its role within this ecosystem. Compared to a shark or elephant, the loss of such a species in this particular ecosystem is not gauged at the same level, however, we are also limited in our evaluation of its role within the ecosystem. We did look into existing estimates for the value of the turtles and tortoises to ecosystems more generally but found there to be few studies applicable to the case.

3 Population Trends and Trade

3.1 Known Population Estimates

The first challenge to evaluating the impact of legal and illegal trade of horsfieldii is the fact that information on their population density is limited and outdated. This can be attributed on the one hand to the species' extensive range and reclusive habits and on the other to its perceived insignificance compared to a flagship or keystone species such as a panda, rhino or elephant. The most comprehensive population evaluation was done by the Tortoise and Freshwater Horsfieldii Specialist Group in 1996 on behalf of the IUCN, declaring the species vulnerable.[19] In the European Commission 2008 Report on heavily traded species, the horsfieldii was noted as having once been abundant throughout Central Asia but cited that in more recent studies populations are found to be declining rapidly in all range states.[20]

Independent studies on population data of horsfieldii have been assessed in some countries, however, they are scattered both in terms of the time period they were under taken and the geographic location covered. While these studies are far from conclusive, they do provide insight of population figures before the ascent of the international pet trade and they also indicate a clear decline in population density since the pet trade.[21]

Kazakhstan: Population figures for Kazakhstan are the most comprehensive with estimates in the 1950s of 5-72 individuals per hectare (Paraskiv, 1956 as cited in Lee and Smith) and a similar study conducted between 1975-1979 found 0.2 to 29 individuals per hectare (Kubykin 1982 as cited in Lee and Smith). The last census in 2000 recorded densities in the same region of 3.9 to 10.3 tortoises per hectare (Kuzman 2002 cited in Lee and Smith 2010).[22] Field studies undertaken in the 1980s in Kazakhstan indicate that harvest from the wild can lead to a complete population collapse from large areas cited in Traffic report (2000).[23]

Uzbekistan: Unpublished studies indicate densities of 0.5 to 43 tortoises per hectare with a total population estimate of 20 million individuals in Uzbekistan. This has been provided by the Uzbekistani government in 1997 and again in 2011.[24] However, this study remains unpublished and was produced by a commercial exporter and may be subject to biased information to justify continued exploitation of the species.[25]

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[19] IUCN, "The Horsfieldii Trade - Wildlife Conservation Society."

[20] Bonin, F., Devaux, B. & Dupré, A. 2006. Horsfieldiis of the World. English translation by P.C.H. Pritchard. Johns Hopkins University Press, 416 pp

[21] Smith and Lee, "Testudostan: Our Post-Cold War Global Exploitation of a Noble Tortoise."

[22] Ibid.

[23] TRAFFIC Europe, *Ranching and Breeding of Horsfield Tortoises (Testudo Horsfieldii) in Uzbekistan*, 2000.

[24] Ibid.

[25] Smith and Lee, "Testudostan: Our Post-Cold War Global Exploitation of a Noble Tortoise."

China: Studies undertaken in China where the species largely no longer exists reported the distribution area and population density of horsfieldii were observed in an area of 500km^2 with 4136 ± 2162 individuals/km^2 in the earlier 1960s; in an area of 270km^2 and 61.5±31 individuals/km^2 in the earlier 1980s; in an area of 180km^2 an 6.04 ind./km^2 in the earlier 1990s.[26] In China, populations of horsfieldii have collapsed and densities are 1% what they were in the 1950s (Luxmoore, Groombridge and Broad, 1988) cited in TRAFFIC Europe 2000).[27]

3.2 Replacement Rates

Like many turtle and tortoise species, horsfieldii mature slowly and have modest reproduction capacities. For these reasons, horsfieldii are poor candidates for legal commercial trade and are easily susceptible to collapse in the presence of over harvesting which can happen when illegal trade takes place.[28] Adult females reach sexual maturity after 10 years but are not considered fully mature until 20 or 30 years of age.[29] A female will produce a clutch of two to three eggs and two to three clutches year. The hatchlings have a 70-90% predation rate during their first year.

Their slow generation means that specimens removed from the wild can seriously skew populations and have an impact on their ability to sustain population levels. Moreover, it is estimated that 95% of horsfieldii that enter the pet trade die within a year, thus harvest rates may be significantly higher than those corresponding to that which is documented in CITES trade data.[30]

3.3 Potential Illegal Trade: Ukraine's Use of Various Source Codes for Export of Horsfeildii

A specific and noticeable example of potential illegal trade took place after the EU implemented a trade ban on wild caught horsfieldii from 1999 to 2006. This trade ban resulted in an unexpected supply of captive-bred specimens exported from the Ukraine to the EU. The dramatic shift in trade using various captive bred source codes to justify legal export and import from a non-range country with no prior evidence of hatching or breeding facilities flags the likelihood of illegal activity.[31] A total of 83,293 non-wild

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[26] Shi, Hai-tao. (1998). „Studies on Ecology of Testudo Horsfieldii Gray and Status of its Conservation," in Sichuan Journal of Zoology. Available at: http://www.scdwzz.com/viewmulu_en.aspx?qi_id=103&mid=4342

[27] TRAFFIC Europe, *Ranching and Breeding of Horsfield Tortoises (Testudo Horsfieldii) in Uzbekistan*.

[28] Smith and Lee, "Testudostan: Our Post-Cold War Global Exploitation of a Noble Tortoise."

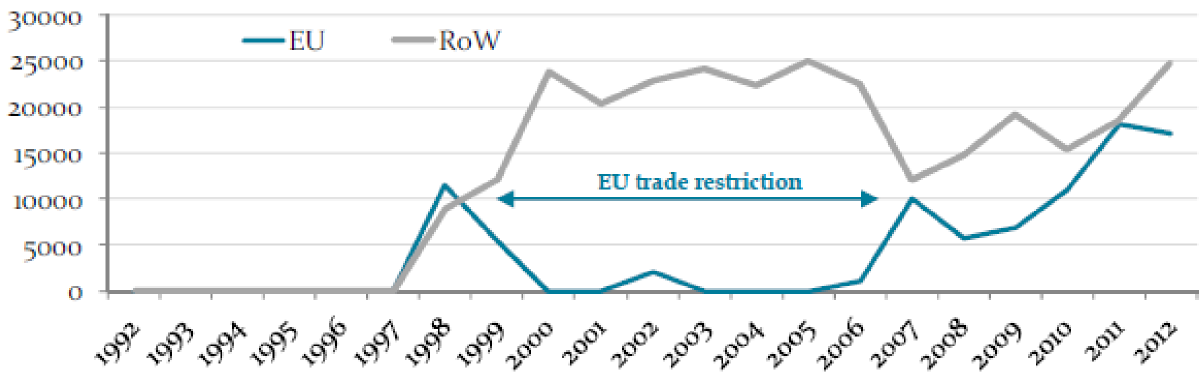
[29] Szczerbak, N.N. (2003) Guide to the reptiles of the eastern Palearctic. Krieger Publishing Company, Malabar, Florida.

[30] John Behler, "Troubled Times for Horsfieldiis," in *IUCN/SSC Tortoise and Freshwater Horsfieldii Specialist Group* (Conservation, Restoration, and Management of Tortoises and Horsfieldiis- An International Conference, Purchase, New York, 1993).

[31] European Commission, *Analysis of the Impact of EU Decisions on Trade Patterns. Report 3: Shifts in Sources of Specimens and Purposes of Trade*.

specimens were imported into the EU between 2000 and 2006 of which more than 90% of them originated from Ukraine using the source code C (Captive Bred) and F (Farmed).[32] The EU trade ban also had the unexpected consequence of increasing the overall specimens in trade.[33] During the 1999-2006 period, the number of specimens in trade nearly doubled, as Uzbekistan continued to export wild caught species to countries other than the EU. In addition, the "captive-bred" specimens coming from the Ukraine met the demands of the EU market for horsfieldii, thereby unintentionally doubling the total number of specimens in trade from Uzbekistan.

Figure 2 Direct Global Imports of Testudo Horsfieldii from Uzbekistan



Source: European Commission, *Analysis of the Impact of EU Decisions on Trade Patterns. Report 3: Shifts in Sources of Specimens and Purposes of Trade* (Cambridge, 2014). P12

Ukraine maintained its status as an export country of horsfieldii after the EU trade ban was removed. From 2008 to 2012, an additional 50,347 live Horsfieldii tortoises were imported by countries from the Ukraine and declared as F (farmed).[34] Importing countries reported an additional 21,365 individual Horsfieldii as being re-exported by Ukraine and declared W (wild) between 2008 and 2010. The fact that Ukraine could harvest and breed horsfieldii in such a quantity as its exports indicate is unlikely given that there were so few imports of live specimens. Ukraine had a one-time import of 5000 wild horsfieldii specimens in 2001 from Uzbekistan and did not report additional imports of wild horsfieldii until 2008 when it reported importing 14,000 specimens from Tajikistan.[35]

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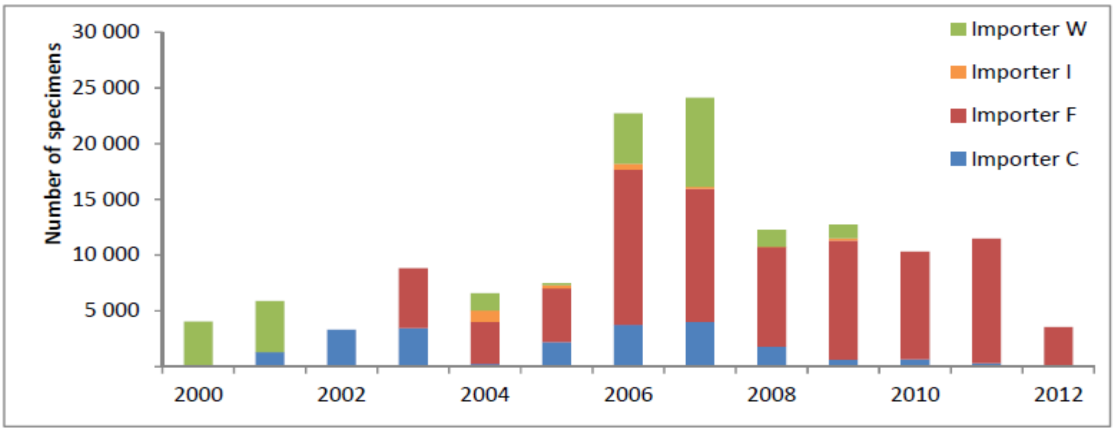
[32] TRAFFIC, *Captive-Bred...or Wild-Taken? Examples of Possible Illegal Trade in Wild Animals through Fraudulent Claims of Captive-Breeding*.

[33] European Commission, *Analysis of the Impact of EU Decisions on Trade Patterns. Report 3: Shifts in Sources of Specimens and Purposes of Trade*.

[34] CITES, *Implementation of the Convention Relating to Captive-Bred and Ranched Specimens (Decision 16.65)*.

[35] Ibid.

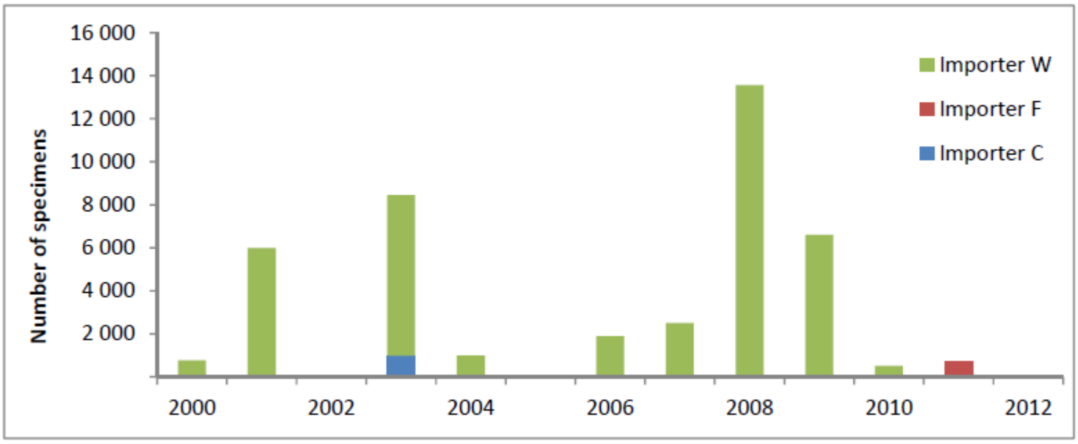
Figure 3 Exports of live Horsfieldii Tortoises from Ukraine as Reported by Importers (2000-2012)



Does not include re-exports. Data from some years may not be complete.

Source: http://cites.org/sites/default/files/eng/com/ac/27/E-AC27-17.pdf

Figure 4 Re-exports of Live Horsfieldii Tortoises from Ukraine as Reported by Importers (2000-2012)



Available data for 2012 may not be complete.

Source : http://cites.org/sites/default/files/eng/com/ac/27/E-AC27-17.pdf

It is therefore likely that illegal activity took place whereby specimens labelled ´captive` or ´ranched` from Ukraine were collected from the wild from range states and re-exported to the EU via Ukraine with incorrect source codes. This example of potential illegal trade encompasses two possible routes of illegality: 1) the manipulation of source codes wild caught versus captive-bred to meet export and import requirements 2) the smuggling and transportation of specimens through third party countries with less stringent enforcement or non-party CITES status.

3.4 Unsustainable Legal Trade: Uzbekistan and Self-Set Annual Quotas

As the main exporter of horsfieldii and a member of CITES, Uzbekistan has provided export quotas on a near annual basis for horsfieldii since 1999, and has also included information on the number of wild caught versus captive bred specimens in most circumstances.[36] Self-set quotas for Uzbekistan have increased substantially in the last decade from a total of 25,000 (wild and captive not distinguished) in 1998 to 100,000 (50,000 wild and 50,000 captive) in 2014. It is also possible to observe that in recent years export quotas have increased substantially from year to year and in some cases by over 30%. For example, in 2011 the quotas for wild specimens were increased to 40,000 when in 2010 a quota for 29,000 existed for live specimens.[37] Captive bred exports from 2011 were also increased from 22,000 in 2010 to 30,000 in 2011. These increases represent a 38% increase for assumed wild caught exports and 36% increase for ranched exports.[38]

Due to the large number of specimens in trade, CITES has investigated and requested information on population figures of wild horsfieldii in Uzbekistan from the Uzbekistan government and scientific authority in 2008. During this consultation, the Uzbekistan commercial supplier reported to CITES that population estimates for the country were approximately 20 million tortoises.[39] If this number is correct, the legal trade in Uzbekistan would account for less than 1% of the total population of the species and therefore would be unlikely to pose a threat to its conservation status in the wild.[40] However, it is difficult to accept the accuracy of this population estimate. Firstly, the estimate has been provided by a single supplier, a company called the OOO ZooComplex that conducts its own research and publications.[41] Moreover, the figure, 20 million has been provided three times over a fifteen year period in 2000, 2007 and 2013. The fact that the figure remains somewhat vague and is re-used as a population reference over thirteen years in spite of dramatically increased harvest rates over the same period makes this figure

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[36] See Appendix 1 for a full list of exports of horsfieldii. Other countries involved in the trade such as Kazakhstan and Tajikistan have provided export quotas sporadically or inconsistently. Several countries that are likely to be involved in the trade have not provided quotas at all and some others are not members of CITES and therefore not required to provide quotas. For example, Tajikistan and Turkmenistan are not members of CITES. A complete list of the reported export quotas are available on the Species + database.

37 The Uzbekistan export quota for horsfieldii in 2010 is 29000 live and 22000 live/ranched. The first figure does not distinguish whether the specimens were wild caught or bred.

[38] UNEP-WCMC, *Analysis of 2011 CITES Export Quotas (version Edited for Public Release)*, Prepared for the European Commission (Cambridge, 2011), http://ec.europa.eu/environment/cites/pdf/reports/analysis_export_quotas_2011.pdf.

[39] For a full list of Export Quotas see Annex 1

[40] DG Environment, *Analysis of 2013 CITES Export Quotas*, UNEP World Conservation Monitoring Centre. Available at: http://ec.europa.eu/environment/cites/pdf/reports/SRG%2064%20Analysis%20of%202013%20CITES%20export%20quotas.pdf

[41] For more information see: http://www.zoocomplex.com/science/

questionable, which is unlikely given the high levels of harvest that has taken place annually over several decades.

3.5 Summary

The last independent evaluation of the population available on the IUCN Red List website of the Horsfieldii is from 1996, an evaluation that deemed the species vulnerable to exploitation and placed it in Appendix II of CITES. Export quotas at the end of the late 1990s were set in the range of 20,000 specimens from Uzbekistan and 20,000 for the entire Russian Federation. Now in 2014, the number of exported specimens is more than double those figures. From Uzbekistan, quotas for traded horsfieldii are around 100,000 with 50,000 labelled wild caught and 50,000 labelled bred in captivity. It is likely that a large proportion of those labelled as captive bred are actually wild caught. There are several indications:

1. Many adult specimens exist in trade;
2. It is unlikely that commercial facilities are able to produce annually that many specimens given the species slow reproductive functions and problems in captivity;
3. The responsible authority in Uzbekistan, the Customs and Biological State Control Agencies, estimated in the year 2000 that the annual illegal export was around 7,000 tortoises from Uzbekistan, 25,000 from Kazakhstan and 40,000 in total from Central Asian Countries (it is not clear how this information was obtained). However, the Uzbek government gave a much higher figure in 2007 of 35,000 Horsfieldii not accounted for in the trade statistics.[42]

There is only one known Horsfieldii farm in Uzbekistan it is relatively safe to assume that at least 50-75% of the horsfieldii labelled as captive bred are actually illegally wild caught. Such estimates would mean that 20,000 to 30,000 specimens are illegally provided each year.[43]

In addition to potential illegal trade, the level of the legal trade of wild caught tortoises in Uzbekistan has increased substantially in recent years without any evidence of an increased population. So if the quota of 22,000 (wild caught) was sustainable in 2006 it is not quite understandable how a quota of 50,000 (wild caught) can be sustainable for the same population in 2014.

It is therefore likely that the overexploitation of the tortoise populations runs to at least 20,000 to 30,000 animals a year in Uzbekistan and if the real replacement rate is close to the old quota of 22,000 as was reported in 2006 that would mean an exploitation of double the sustainable rate leading clearly towards a population decline.

Poor and unreliable data makes it not possible to gauge the rate of extinction, however, this situation of data availability is relevant for many traded CITES species.

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[42] UNEP-WCMC, *Review of Species Selected on the Basis of a New or Increased Export Quota in 2008*.

[43] UNEP-WCMC, *Review of Species Selected on the Basis of a New or Increased Export Quota in 2008*.

Figure 5 Number of Horsfieldii Traded and Quotas (CITES Data)

	2006	2007	2008	2009	2010	2011	2012	2013	2014
Traded animals – wild caught	62,964	38,674	58,850	45,467	31,157	40,387	42,794	45,606	
Traded animals bred	39,559	36,010	31,812	30,608	35,844	38,374	36,427	46,064	
Total traded animals	102,523	74,684	90,662	76,075	67,001	78,761	79,221	91,670	
Wild Quota Uzbekistan	22,000	22,000	22,000	29,000	Quota not given	40,000	42,100	45,000	50,000
Wild Quota Tajikistan	Quota not given	17,000	17,000	17,000	Quota not given	Quota not given	Quota not given	Quota not given	Quota not given
Wild Quota total	22.000	39,000	39,000	46,000		40,000	42,100	45,000	50,000

4 Quantitative/Qualitative impacts

The numbers above show that it is likely, although impossible to prove, that the current rate of exploitation does lead to a decrease in population. This decline could be valued using the market value of the horsfieldii if sold legally and indirectly by the importance of the horsfieldii for the ecosystem and the value of the ecosystems as a whole. Additionally some qualitative information on the environmental impacts in the import countries is provided.

4.1 Market Value of Tortoises

A *Horsfieldii* tortoise is sold as a pet for between $25 and $100 USD, depending on geographic selling location and season (because of demand).[44] The price paid to exporters/collectors in source countries was estimated in 1997 at €0.45 per individual.[45] When comparing the final sale price to that of the wholesale price, it becomes clear that the majority of the earnings stay with the importing country and pet dealership.

From this information it is quite clear that the overall value of the pet trade for the source countries is negligible. Overall the countries are exporting around 80,000 live animals and this provides an overall value of less than €40,000. It is likely that in the case of properly regulated market and a sustainable rate of exploitation the value for exporting countries might increase due to greater scarcity but it is unlikely that this could ever grow into a significant income stream for the exporting countries.

4.2 Environmental Impacts

The main environmental impact is the decrease in overall tortoise population numbers and eventual digression of the population in the wild towards extinction. Due to the fact that there are not concrete population figures on the horsfieldii it is useful to compare the impact of wild harvesting on other tortoise and turtle species. There are currently 317 recognized species of turtles and tortoises in the world. Of those that have been assessed by the IUCN Red List, 63% are considered threatened, and 10% are critically endangered. 42% of all known turtle species threatened.[46] Turtle and tortoise diversity is particularly affected by commercial trading and it has been proven that many populations that experience a continual level of exploitation result in collapse.[47] Conservation biologists have cited examples of tortoise and turtle

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[44] Big Apple Pet Supply, "Russian Tortoise (4' - 5')," March 13, 2015, http://www.bigappleherp.com/Russian-TortoiseIg A.

[45] TRAFFIC Europe, *Ranching and Breeding of Horsfield Tortoises (Testudo Horsfieldii) in Uzbekistan*.

[46] Kurt Buhlmann et al., "A Global Analysis of Tortoise and Freshwater Turtle Distributions with Identificaiton of Priority Conservation Areas," *Chelonian Conservation and Biology* 8, no. 2 (2009): 116–49.

[47] Frank Biermann et al., "The Fragmentation of Global Governance Architectures: A Framework for Analysis," *Global Environmental Politics* 9, no. 4 (2009), https://www.dropbox.com/sh/70r734b7iv9d8p9/f4qsp1YJTr/Biermann%20et%20al.%202009_The%20Fragmentation%20of%20Global%20Governance%20Architectures.pdf; Smith and Lee, "Testudostan: Our Post-Cold War Global Exploitation of a Noble Tortoise."

species experiencing unsustainable harvests in their range. For example, Asian Box turtle, Roofed tortoise, and Asian Softshell turtles have experienced a precipitous decline as they have been hunted for markets, both pet and medicinal.[48] The Tortoise and Freshwater Turtle Specialist Group explained in a 2011 petition to CITES from the Centre for Biological Diversity that "natural populations of horsfieldiis are characterized by a suite of life history characteristics that may predispose these populations to rapid declines in the face of anthropogenic harvest."[49]

For the horsfieldii, It is not possible to gauge the rate of extinction because current population figures are extremely out dated and incomplete. Moreover, there is little information available regarding the role of the horsfieldii tortoise in the Central Asian steppe ecosystems. The horsfieldii is not for instance a keystone species or a predator and its excessive inactivity (9 months of the year) illustrates the tortoises unique ability to survive in a harsh and desolate environment more than it illustrates its inherent purpose within this environment. Thus, the Horsfieldii is an example where the ecosystem value of the species is of lesser consequence and efforts for its conservation are based mostly on its intrinsic value. Overall it needs to be concluded that there is no evidence that the ecosystem as a whole and its ecosystem services would suffer if the horsfieldii tortoise population would decrease further[50].

In attempting to value its role in the ecosystem, we looked in the TEEB and EVRI databases but did not find relevant or similar studies that attempted to gauge the economic value of the existence of a similar species. While there were some examples of sea turtles and their value to island and coastal communities,[51] these studies were not deemed comparable due to the sea turtles inhabitation of a very different marine ecosystem and its strong role within ecotourism for which cannot be attributed to the horsfieldii tortoise in Central Asia.

4.3 Environmental Impact in Consumer Countries

The main environmental side effect of illegal and legal wildlife trade of the horsfieldii tortoise relate to the risks associated with invasive species and pathogen pollution. Many horsfieldii that end up as pets in importing countries are at one point or another "let go" or abandoned by their owners, in particular, because of their long lifespan. At the point of release, many Horsfieldii are sick with disease from poor caretaking by pet owners and because of the dramatically different climatic conditions in host countries.[52]

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[48] Ted Williams, "The Terrible Turtle Trade."

[49] Centre for Biological Diversity, *Re: Species Proposals for Consideration at CITES CoP16*, Petition, (2011), http://www.biologicaldiversity.org/campaigns/southern_and_midwestern_freshwater_Horsfieldiis/pdfs/Freshwater_Horsfieldiis-CITES_petition_Aug_8.pdf.

[50] Databases containing valuation studies of ecosystems and biodiversity were consulted to see if any similar studies attempted to put an economic value on conservation of a tortoise or turtle species, however, information was extremely limited and comparable studies were not found.

[51] See: Clem Tisdell and Clevo Wilson. 2002 Economic, Educational and Conservation Benefits of Sea Turtle Based Ecotourism: A Study Focused on Mon Repos. Wildlife Tourism Research Report Series: No 20. Cooperative Research Centre for Sustainable Tourism.

[52] Smith and Lee, "Testudostan: Our Post-Cold War Global Exploitation of a Noble Tortoise."

When released into a non-native habitat they pose the twofold impact of introducing a non-native species that could compete with local turtle and tortoise species and also pose the risk of introducing disease that affects local wildlife and potentially human beings.

There are several documented cases of imports of turtles and tortoises negatively affecting flora and fauna in importing countries. The red slider turtle is a common example of an invasive turtle species that is now found outside of its natural range in southern Europe, Africa, Asia and the US. Recognized as harming local species, the red slider was banned in 1997 by the EU for import because of the damage they brought on local European fresh water turtle populations.[53] According to a 2014 publication by Nature on invasive species, horsfieldii are in the first stage of invasive species introduction in the United States.[54]

Invasive turtle species also pose health threats. Several examples of imported turtle species carrying salmonella with the potential to pass on to human populations has been documented in Spain.[55] While there is not a specific example of the Horsfieldii tortoise spreading disease, tracing such links are difficult and often studied after the fact on a case by case basis. In general, sick and imported tortoises transported in unsanitary and poor conditions carry a certain likelihood of spreading disease. The introduction of a non-native species and potential diseases effect both importing and exporting countries.[56] However, it is important to note that some of these impacts would occur through the release of tortoises which were farmed and imported legally as well as those which were traded illegally.

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[53] O. Kopecý, L. Kalous, and J. Patoka, *Establishment Risk from Pet-Trade Freshwater Horsfieldiis in the European Union* (Suchdol, Czech Republic: Czech University of Life Sciences Prague, Facultural of Agriculture, Food and Natural Resources, 2013).

[54] Reuben P. Keller, Marc W. Cadotte, and Glenn Sandiford, *Invasive Species in a Globalized World: Ecological, Social, and Legal Perspectives on Policy* (University of Chicago Press, 2014).

[55] J. Hidalgo-Vila et al., *Salmonella in Free-Living Exotic and Native Horsfieldiis and in Pet Exotic Horsfieldiis from SW Spain* (Madrid, Spain: Laboratorio Central de Veterinaria, Ministerio de Agricultura, Pesca y Alimentación, 2008).

[56] Pro Wildlife. 2000. The decline of the Asian Horsfieldii. https://www.prowildlife.de/sites/default/files/Horsfieldii%20report.pdf

5 Conclusions

The Horsfieldii tortoise is a heavily traded species of tortoise and its status in the wild is threatened by both legal CITES trade and illegal trade. This paper outlined several plausible circumstances where illegal trade had or currently was taking place. It identified three potential instances of illegality. 1) It is likely that many more species are involved in the trade than are actually reports.[57] 2) It is suspected that the improper use of CITES labels that differentiate between wild and captive bred specimens, results in a much higher number of wild caught specimens existing in trade than the data reported.[58] 3) It is likely that many individual tortoises are illegally transported and smuggled through non-Party countries or countries with less stringent environmental and enforcement standards.[59] One case of illegal trade that was especially obvious was the Ukraine. It became clear with the example of Ukraine exporting large quantities of 'captive-bred' Horsfieldii after the 1999 EU ban on wild caught specimens, that illegal trade took place, in this case involving both the manipulation of source codes and the smuggling of specimens from transit countries. An interesting and unexpected impact of the EU trade ban on wild caught specimens that was implemented in attempt to protect the species had the negative and unexpected result of actually, was the doubling the overall number of specimens in trade. This resulted from the fact that exports from Uzbekistan continued to non-EU countries (namely the US and Japan), while EU imports were met by Ukrainian exports of mislabelled "captive bred" specimens, which were likely wild caught specimens smuggled from Uzbekistan and other range states.

This report also found that the legal trade in Horsfieldii as dictated through annual CITES quotas was potentially threatening to the conservation status of the species. For Uzbekistan the quota continues to be increased year after year despite that no comprehensive and independent study of the tortoise population has been conducted since 1997. The way that CITES is designed, allows for the decision on the quota or legal harvest to be determined by each sovereign Party. When quotas are established in this way, they depend on the assumed interest and ability of the Party member to establish a sustainable rate of harvest. A species may then be legally over-harvested. Reasons for over harvesting are variable and depend on the specific species and country, but could include factors such as: a) low level of political will to address issue b) relative importance in society and/or awareness of species conservation status c) high profits incurred for the country harvesting and exporting d) inadequate resources to monitor population. These are some examples that could affect the assumption inherently made by CITES that states have an obligation and interest in accurately monitoring and reporting the conservation status of all relevant species and implementing sustainable quotas for harvest. It would be useful and important to have an independent

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[57] Smith and Lee, "Testudostan: Our Post-Cold War Global Exploitation of a Noble Tortoise"; UNEP-WCMC, *Review of Species Selected on the Basis of a New or Increased Export Quota in 2008*.

[58] CITES, *Implementation of the Convention Relating to Captive-Bred and Ranched Specimens (Decision 16.65)*.

[59] Tajikistan and Turkmenistan are not Parties and therefore do not report levels of trade and several other of the countries identified above do not provide self-reported data on exports nor do they set quotas, despite that they are likely involved in the trade as transit countries or collection countries.

census on the status of species in the wild to justify high levels of export.[60] Thus, the approval of increasing CITES quotas for the Horsfieldii, pose a problem for the species because there is inconclusive data on populations. Moreover, while it is not easy to gauge the level of illegal trading, it is acknowledged to exist parallel to the legal trade. The fact that several neighbouring or trade involved countries are not party to CITES or do not hand in their required annual reporting documents (e.g. Ukraine), which indicate potential issue areas for accurately understanding the trade and its impact on the conservation status of the species and biodiversity more generally.

While this study fell short of valuing the illegal Horsfieldii tortoise trade in monetary terms, it did illustrate how illegal and legal trade can lead to the unsustainable exploitation of a species that could, if continued, lead to its eventual extinction. There are three reasons why the valuation of the overexploitation of the Horsfieldii is so challenging:

- Firstly, the legal value of the specimen at the site of collection is monetarily insignificant therefore legal trade is not likely to provide a significant income stream and cannot therefore be used for valuation of the damages of illegal trade.
- Secondly, the value of the specimens to the local ecosystem is either not known or perceived to be small.
- Thirdly, there are not many studies on the value of whole ecosystems, which do not attract tourists or provide other known ecosystem services.[61]

In that respect the case of the Horsfieldii is an ordinary one and representative of many traded species. The silent majority of species traded in CITES are likely to be affected by lack of data, publicity and easily monetised value. For these species, it is difficult to use valuation techniques to support conservation efforts.

[60] Smith and Lee, "Testudostan: Our Post-Cold War Global Exploitation of a Noble Tortoise."

[61] While *The Economics of Ecosystems and Biodiversity* (TEEB) database could be used to illustrate the economic value of a species to a specific ecosystem or ecosystem services, the amount of literature on these topics as they relate to turtles and tortoises in the downloadable TEEB and the EVRI (Environmental Valuation Reference Inventory) databases was extremely limited and comparable studies were not found.

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Annex A CITES EXPORT QUOTAS

YEAR	COUNTRY	QUOTA	NOTES
2014	Uzbekistan	50000	live, ranched
2014	Uzbekistan	50000	live, wild-taken
2013	Uzbekistan	45000	live, ranched
2013	Uzbekistan	45000	live wild-taken
2012	Uzbekistan	30000	live, ranched
2012	Uzbekistan	42100	live wild-taken
2011	Uzbekistan	30000	live, ranched
2011	Uzbekistan	40000	live, wild-taken
2010	Uzbekistan	29000	live
2010	Uzbekistan	22000	live, ranched
2009	Tajikistan	17000	wild-taken
2009	Uzbekistan	5000	eggs
2009	Uzbekistan	29000	live
2009	Uzbekistan	17000	live, ranched
2008	Tajikistan	17000	wild-taken
2008	Uzbekistan	5000	eggs
2008	Uzbekistan	22000	live
2008	Uzbekistan	2000	live, captive-bred
2008	Uzbekistan	17000	live, ranched
2007	Tajikistan	17000	wild-taken
2007	Uzbekistan	5000	eggs

YEAR	COUNTRY	QUOTA	NOTES
2007	Uzbekistan	13000	ranched
2007	Uzbekistan	22000	wild-taken
2006	Uzbekistan	14000	ranched
2006	Uzbekistan	22000	wild-taken
2005	Uzbekistan	13000	ranched
2005	Uzbekistan	22000	wild-taken
2004	Uzbekistan	7000	ranched
2004	Uzbekistan	23000	wild-taken
2003	Uzbekistan	1150	live (confiscated animals)
2003	Uzbekistan	5000	ranched
2003	Uzbekistan	25000	wild-taken
2002	Kazakhstan	40000	live
2002	Uzbekistan	30000	live (wild-taken and ranched)
2001	Kazakhstan	40000	live
2001	Tajikistan	20000	wild-taken
2001	Uzbekistan	30000	live (wild-taken and ranched)
2000	Kazakhstan	39000	live
2000	Uzbekistan	35000	live
1999	Russian Federation	20000	as re-exports from Kazakhstan
1999	Russian Federation	15000	as re-exports from Tajikistan
1999	Uzbekistan	35000	

YEAR	COUNTRY	QUOTA	NOTES
1998	Russian Federation	25000	re-export; origin Uzbekistan
1998	Uzbekistan	25000	
1997	Russian Federation	20000	re-export; origin Uzbekistan