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The Hidden and External Costs of Pesticide Use — Source link

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Chapter 2
The Hidden and External Costs of Pesticide Use

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Abstract A fair evaluation of the net benefits provided by pesticides is essential to feed the current debate on their benefits and adverse consequences. Pesticides provide many benefits by killing agricultural and human pests. However, they also entail several types of costs, including internal costs due to the purchase and application of pesticides, and various other costs due to the impact of treatments on human health and the environment. Here, we provide a comprehensive review of these costs and their evaluation. We define four categories of costs: regulatory costs, human health costs, environmental costs and defensive expenditures. Those costs are either internal to the market, but hidden to the users, or external to the market and most often paid by a third party. We analysed 61 papers published between 1980 and 2014, and 30 independent dataset. Regulatory costs reached very large values, e.g. US$4 billion yearly in the United States in the 2000s. However, if all regulations were respected, these costs would have jumped to US$22 billion in this country. Health costs studies generally did not take into account fatal cases due to chronic exposure such as fatal outcomes of cancers. Doing so would have increased estimates of health costs by up to tenfold, e.g. from US$1.5 billion to US$15 billion in the United States in 2005.

Most environmental impacts have never been quantified in the literature. Environmental costs were nevertheless estimated to up to US$8 billion in the United States in 1992. Although defensive expenditures have rarely been considered in the literature, they include at least the extra cost of the part of organic food consumption due to aversive behavior linked to pesticide use. This cost reached more than US$6.4 billion worldwide in 2012. Our review thus revealed that the economic costs of pesticide use have been seldom considered in the literature and have undoubtedly

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been strongly underestimated in the past. Despite this underestimation, we found that overall hidden and external costs ranged from US$5.4 million in Niger in 1996 to US$13.6 billion in the United States in 1992. We perform an updated and more complete retrospective evaluation of these costs in the United States and show that they probably reached the value of US$39.5 billion per year at the end of the 1980s-start of the 1990s. We also re-evaluate past benefit-cost ratio of pesticide use in various countries and reveal that the cost of pesticide use might have outreached its benefits, e.g. in the United States at the start of the 1990s. We finally advocate that the key impact to be evaluated is the cost of illnesses and deaths triggered and favored by chronic exposure to pesticides. The benefit-cost ratio of pesticide use may have easily fallen below 1 if this cost had been taken into account. The quantification of this key cost is therefore urgently required for a more accurate evaluation of pesticide use and for regulatory purposes.

Keywords Insecticides • Fungicides • Herbicides • Environmental impact • Cost-of-illness • Defensive expenditures • External costs • Benefit-cost ratio analysis

2.1 Introduction

High levels of agricultural productivity will be required to sustain the world population, given current population growth rates. Between 1960 and 2000, the Green Revolution increased global food production by a factor of two to three (Evenson and Gollin 2003). However, the approaches used to increase production damaged many ecosystems, rendering them more vulnerable to pests. The control of these pests is essential if we are to maintain the high levels of productivity required to meet demand. The growth of the world population has also been accompanied by a desire to improve the length and quality of human life. With people living longer and in better health, food demands have increased, also necessitating the effective control of pests.

Organisms harmful to humans, their environment and production can be controlled in many different ways. Pesticides are one of the most widely used and effective tools for this purpose. Almost two billion people work in agriculture, and most use pesticides to protect their crops or livestock. Pesticides are also widely used in gardens and around the home, in the framework of public health programs. Pesticide sales increased by a factor of 20–30 between the 1960s and 1990s (Oerke 2006). Pesticide use has continued to increase over the last two decades in most developing countries, e.g. Thailand during the 1990s and 2000s (Praneetvatakul et al. 2013) and Pakistan during the 1990s (Khan et al. 2002). Moreover, contrary to what is commonly believed, pesticide use has remained stable in several developed countries, e.g. the United States (Osteen and Fernandez-Cornejo 2013), mostly due to an increase in herbicide use (Schreinemachers and Tipraqsa 2012). Overall pesticide consumption is currently close to two to three million tons per year (United States Environment Protection Agency 2011), 45 % of all pesticides being used in Europe, 25 % in the United States, 4 % in India and 26 % in the rest of the world (De et al. 2014). Total expenditure on pesticides is about US$40 billion per year (Popp et al. 2013).

Despite the high cost of their purchase, the widespread application of pesticides has been favored by the benefits they provide. In particular, they have increased crop and livestock yields and, in some circumstances, have improved human health, e.g. by killing vectors of human pathogens, and quality of life, e.g. by killing troublesome organisms (Cooper and Dobson 2007).

However, the purchase costs are only one of the types of cost associated with pesticide use. Indeed, the spraying of these chemicals has an impact on the environment and health, with potentially serious financial consequences (Fig. 2.1). For instance, in a report published in 1990, the World Health Organization (WHO)

Fig. 2.1 Vietnamese farmer spraying pesticide on rice without protections in Hội An, Quảng Nam, Vietnam. A fair evaluation of the net benefits provided by pesticides requires a thorough estimation of their costs, including those associated with their impact on health and the environment. The purchase costs are only one of the types of cost associated with pesticide use. Indeed, the spraying of these chemicals has an impact on the environment and health, with potentially serious financial consequences. For instance, farmers take safety measures when handling and applying pesticides to their crops, to decrease or prevent direct exposure to these chemicals. The defensive expenditures taken into account include costs associated with precautions taken to reduce direct exposure to pesticides, such as masks, caps, shoes/boots, handkerchiefs, long-sleeved shirts/pants. Spraying is sometimes carried out without protection and even those farmers who do try to protect themselves generally limit this protection to the wearing of long-sleeved shirts and long pants. Low levels of income, awareness and education, the hot and humid climate, cultural taboos, fashion and discomfort are significant factors accounting for the lack of personal protection (Unmodified photography by Garycycles, under Creative Common License CC BY (https://creativecommons.org/licenses/by/2.0/))

indicated that there may be as many as one million unintentional severe acute poisoning incidents annually, resulting in 20,000 deaths (WHO 1990). These serious cases of poisoning account for a minute fraction of the overall impact of pesticides on health. On the basis of a survey of self-reported minor poisoning events in Asia, Jeyaratnam (1990) estimated that as many as 25 million agricultural workers in the developing world annually may suffer a poisoning incident.

A fair evaluation of the net benefits provided by pesticides requires a thorough estimation of their costs, including those associated with their impact on health and the environment. Donald J Epp and coworkers (1977) were probably the first to espouse this idea, with the description of a complete taxonomy of the negative impacts of pesticide use to be taken into account. However, they concluded that the state-of-the-art at the time at which they wrote their report was insufficiently advanced for a monetary evaluation of environmental impacts. David Pimentel performed such an evaluation few years later, providing the first overall estimate of the externalities induced by pesticide use. The articles he published from the late 1970s (Pimentel et al. 1979) onwards (most recently, Pimentel and Burgess 2014) focused on the United States. They inspired a few studies in other countries, but there has never been a synthetic analysis of these studies, their shortcomings, limitations and conclusions. Such a synthesis is essential for the current debate on the benefits and consequences of the use of these chemicals.

This review aims to (i) identify and categorize the various costs triggered by the use of chemical pesticides, (ii) provide a comprehensive overview of the articles estimating – in economic terms – these costs, whether at local, regional or national scale, for a single pesticide or for total pesticide use, (iii) report the costs estimated in these articles. These costs, in US$, have been updated to 2013 values, using annual inflation factors and the 2013 purchasing power parity (PPP) conversion factors obtained from the development indicators of the World Bank (http://data.worldbank.org/indicator/PA.NUS.PPP), (iv) identify the consequences for benefit-cost ratio analyses on pesticide use and (v) provide perspectives concerning the evaluation of these costs.

2.2 Types of Costs Generated by the Deleterious Consequences of Pesticide Use

Pesticides are designed to kill, repel, attract, regulate or stop the growth of living organisms considered to be pests (United States Environmental Protection Agency 2007). A pest is any type of living organism, e.g. mammals, birds, reptiles, fish, amphibians, mollusks, insects, nematodes, weeds and microbes (bacteria and viruses), that competes with our food crops or space, spreads disease or acts as a vector for disease and/or causes us discomfort.

Pesticides include chemicals, biopesticides and biological agents (United States Environmental Protection Agency 2007). We have decided to focus this review on chemical pesticides, for several reasons. First, chemical pesticides account for the vast majority of pesticides used worldwide, e.g. more than 80 % in the United States (United States Environmental Protection Agency 2008). Second, chemical pesticides are probably the most harmful pesticides for the environment and human health. For instance, according to the Stockholm Convention on Persistent Organic Pollutants, nine of the 12 most dangerous and persistent organic pollutants are chemical pesticides (United Nations Environment Programme 2001).

We will also focus mostly on chemicals protecting plants from the damage caused by weeds, plant diseases or animals, notably insects. In fact, the term '*pesticide*' is often exclusively used to refer to plant protection products, although pesticides are also used for non-agricultural purposes. Chemical pesticides are of three main types – herbicides, insecticides and fungicides – but several other types of biocides, such as nematicides and rodenticides, are also used.

Pesticide use has been shown to have a marked positive effect on agriculture (Cooper and Dobson 2007; Gianessi 2009; Gianessi and Reigner 2005, 2007) and human health (Cooper and Dobson 2007). However, pesticides may also have deleterious effects on the environment and human health, generating several types of costs.

For the purposes of this review, we have defined four broad categories of costs (Table 2.1): regulatory costs, human health costs, environmental costs and defensive expenditures. Regulatory costs are all the costs entailed as part of private or public mandatory measures to remove pesticides, to protect the environment or human health from the potential damage caused by pesticides and/or to repair damage already inflicted. For instance, the monitoring and decontamination of tap water can be considered a regulatory cost. Human health costs, often referred to as cost-of-illness, are the expenses associated with acute or chronic pesticide poisoning. These costs are mostly incurred by the farmers applying pesticides, although all citizens can be exposed to pesticides and may, therefore, suffer chronic health effects, in particular. Environmental costs are the costs of both pesticide damage to animals, plants, algae and microorganisms and pest resistance to pesticides. These costs may be incurred by farmers or by society as a whole. Finally, defensive expenditures cover all expenses by farmers and society to prevent pesticide exposure, such as the purchase of organic food or bottled water consumption. These four broad categories of costs include both internal and external costs (Table 2.1).

The internal costs of pesticide use are the costs, to the farmer, of pesticide use within the agricultural production process. These costs are described as "internal" because they determine the price of the final product, i.e. they are internal to the market. We do not review here the "usual" internal costs of pesticide use such as market prices of pesticides, taxes on these products, costs of the application, transport and storage of pesticides, accounting costs, etc., but these costs are taken into account in the re-evaluation of overall costs and of the benefit-cost ratio of pesticide use (see Sect. 2.9). We were particularly interested in the "hidden" costs associated with the impact of pesticides on the environment and human health, regulatory measures and defensive behavior. These additional costs are "hidden" in the sense that farmers are not necessarily aware of them. This is the case for environmental

Table 2.1 Types and categories of costs generated by environmental and health impacts, regulatory actions and defensive behavior

		Hidden costs			External costs	
Category of cost	Pesticide impact	Decrease in benefits[a]	Increase in "usual" internal costs[b]	Generation of other internal costs	Private external costs	External costs *sensu stricto*
Regulatory costs	Public research, communication, expertise on pesticides					X
	Regulations, decrees and laws					X
	Mandatory pesticide handling and disposal			X		
Human health costs	Preventive medicine, annual check-ups			X	X	
	Health issues for farmers	X		X	X	
	Health issues for the public					X
Environmental costs	Pesticide resistance	X	X			X
	Soil degradation	X				X
	Pollination decrease	X				X
	Decrease in natural enemies	X	X			X
	Lower plant production due to herbicide application	X				
	Bee renting			X		
	Degradation of the farm environment				X	
	Livestock health issues				X	
	Degradation of the environment					X
	Domestic animal health issues					X
Defensive expenditure	Purchase of protective clothing, glasses and masks			X		
	Purchase of organic food and bottled water					X

[a]Due to lower yields

[b]Due to an increase in the amount of pesticide applied

impacts increasing pesticide requirements for the production process. These hidden costs increase the "usual" internal costs (Table 2.1). The environmental impact of pesticide use may also decrease production levels. Such "hidden" costs are paid through the achievement of a smaller benefit than would have been achieved by farmers in the absence of a deleterious impact of pesticide use (Table 2.1). Finally, pesticide use generates other internal costs, concerning the purchase of protective equipment, e.g. gloves and masks, the renting of bees for pollination, specific mandatory requirements for pesticide handling and disposal, preventive medicine and annual check-ups for farmers. In addition to the usual internal costs, farmers incur this third class of hidden costs directly (Table 2.1).

Environmental and human health impacts, regulatory actions and defensive behavior triggered by pesticide use also generate external costs (Table 2.1). These costs are described as "external" because they are not included in the farmers' production costs, i.e. they are external to the market. They are mostly paid by a third party, but some, such as those concerning the health of the farmer or degradation of the farm environment, may have a direct impact on farmers. Hence, external costs may be incurred by the farmers themselves ("external private costs", Table 2.1) or by other parties, e.g. consumers, public authorities, people living close to the farm ("external costs sensu stricto", Table 2.1).

Health issues for farmers generate both hidden internal costs and external private costs. The impairment of the farmer's health due to the use of pesticides for a specific type of production, such as crop production, may increase crop production costs, e.g. loss of working hours devoted to crop production, lower yields or the need to pay workers for a larger number of hours of work. Some of the costs of pesticide use relating to health are therefore internal. However, the impairment of farmers' health due to pesticide use may also have economic consequences relating to other types of production, such as livestock production, or lower levels of non-market goods, such as childcare or leisure time. Thus, some pesticide costs relating to health issues are external (Table 2.1). However, it is difficult to determine the proportions of health costs that should be considered internal and external. We will therefore consider all these costs as externalities in this review.

Here, we defined four categories of costs – regulatory costs, human health costs, environmental costs and defensive expenditures – that are commonly not included in the economic evaluation of pesticide use. These costs are either internal to the market, but hidden to the users, or external to the market and most often paid by a third party.

2.3 Literature Surveyed

We carried out a literature review as comprehensive as possible, using Google Scholar and the Web of Science, and screening the references cited by the articles identified relating to this topic. We excluded papers based on contingent valuation methods (see below), resulting in the identification of 61 relevant articles in total

(Table 2.2). These articles were published in peer-reviewed scientific journals (23), scientific journals without peer review (11), books (1), book chapters (10), conference proceedings (2), PhD theses (2) and reports (12) (Table 2.2). These 61 papers are based on only 30 independent datasets, because several papers were based on the same dataset (Table 2.2). These publications have differed in terms of their scientific impact. The 12 articles written by Pimentel and coworkers obtained more than 1500 citations in Google Scholar, the other 49 papers having about 2500 citations between them (Table 2.2). The costs estimated for the United States by David Pimentel et al. are the most widely known, and the corresponding dataset is often considered to be the key dataset when referring to the overall cost of pesticide use. Two other datasets have been widely cited: one relating to the externalities of pesticide use in the United Kingdom (Pretty et al. 2000, 2001, cited about 750 times in total) and the other concerning these externalities in the Philippines (Pingali et al. 1994, 1995; Rola and Pingali 1993, cited about 500 times in total).

The studies identified used different methodologies to estimate costs and these methodological differences partly reflect the heterogeneity of the types of cost considered. Some of the impacts of pesticide use have a value that can be directly estimated from market prices. For instance, mandatory governmental regulations concerning pesticide use may require particular activities, e.g. water monitoring (Pretty et al. 2000, 2001; Waibel et al. 1999), and equipment, e.g. water filters (Pimentel et al. 1992, 1993a, b; Pimentel and Greiner 1997; Pimentel and Hart 2001). Their costs can be determined from market values. Other effects, such as food contamination (e.g. Jungbluth 1996) or the loss of working days if the farmer is ill, have costs based on market price (e.g. Pimentel et al. 1980a, b) that can be evaluated by productivity function methods (Bowles and Webster 1995). The same is true for losses of agricultural production (see the series of papers by Pimentel et al.) due to lower pollination rates, livestock health issues, soil degradation or increases in pesticide use due to the selection of pesticide resistance (e.g. Tegtmeier and Duffy 2004). However, some of the goods affected by pesticide use are non-market goods. For instance, the increase in health risk associated with pesticide use has no directly observable price. In such cases, economists must use non-market evaluation techniques to monetize individual preferences. The monetary values obtained with these techniques reflect the individuals' willingness to pay for a reduction of the risk (Travisi et al. 2006). Revealed willingness to pay is an approach in which the monetary value of a change in risk is derived from individuals' purchasing decisions in existing markets. This approach is often used to estimate the costs of aversive behavior, e.g. wearing protection clothes, drinking bottled or purified water, eating organic food, designed to decrease the risk of human health impairment. Revealed willingness to pay can also be used to estimate the cost of wildlife loss. For instance, the cost linked to human activities, such as bird watching, can be used to estimate bird losses due to pesticides (Pimentel 2005). The contingent valuation method – also referred to as stated willingness to pay – is also often used for the market valuation of non-market goods (Venkatachalam 2004). This method is based on stated preferences in hypothetical market settings. We decided not to use estimates based on stated willingness to pay because the answers

Table 2.2 Characteristics of the papers providing at least one estimate of the external cost of pesticide use

Reference	Source	Language	Peer Reviewed?	Dataset	Category of cost considered[a]				Country of the study	No. of citations in Google Scholar
					Reg.	COI	Env.	DE		
Ajayi (2000)	Report	English	No	1		X			Ivory Coast	50
Ajayi et al. (2002)	Report	English	No	2	X	X	X	X	Mali	23
Athukorala et al. (2012)	Sci Journal	English	Yes	3		X		X	Sri Lanka	3
Atreya (2005)	Sci Journal	English	Yes	4		X		X	Nepal	17
Atreya (2007)	Report	English	No	5		X		X	Nepal	8
Atreya (2008)	Sci Journal	English	Yes	5		X		X	Nepal	33
Atreya et al. (2012)	Sci Journal	English	Yes	6		X		X	Nepal	6
Atreya et al. (2013)	Sci Journal	English	Yes	6		X		X	Nepal	0
Brenna (2001)	Sci Journal	Italian	Yes	7		X			Italy	0
Choi et al. (2012)	Sci Journal	English	Yes	8		X			South Korea	0
Cole and Mera-Orcés (2003)	Chapter book	Spanish	No	9		X			Ecuador	6
Cole et al. (2000)	Sci Journal	English	Yes	9		X			Ecuador	60
Corriols et al. (2008)	Sci Journal	English	Yes	10		X			Nicaragua	18
Crissman et al. (1994)	Sci Journal	English	Yes	9		X			Ecuador	129
Devi (2007)	Report	English	No	11		X			India	16
Dung (2007)	PhD thesis	English	No	12		X			Vietnam	2
Dung and Dung (1999)	Report	English	No	12		X			Vietnam	71
Fleischer (1999)	Conf Proc	English	No	13	X	X	X		Germany	5

(continued)

Table 2.2 (continued)

Reference	Source	Language	Peer Reviewed?	Dataset	Category of cost considered[a]				Country of the study	No. of citations in Google Scholar
					Reg.	COI	Env.	DE		
Houndekon and De Groote (1998)	Conf Proc	English	No	14		X	X		Niger	11
Houndekon et al. (2006)	Sci Journal	English	No	14	X	X	X		Niger	5
Huang et al. (2000)	Report	English	No	15		X			China	49
James (1995)	Sci Journal	English	Yes	16			X		Canada	11
Jansen et al. (1998)	Report	English	No	17		X			Costa Rica	4
Jungbluth (1996)	Report	English	No	18	X	X	X		Thailand	63
Khan et al. (2002)	Sci Journal	English	Yes	19	X	X	X		Pakistan	25
Maumbe and Swinton (2003)	Sci Journal	English	Yes	20		X			Zimbabwe	103
Ngowi et al. (2007)	Sci Journal	English	Yes	21		X			Tanzania	64
Pimentel (2005)	Sci Journal	English	Yes	22	X	X	X		United States	272
Pimentel (2009)	Book chapter	English	No	22	X	X	X		United States	[b]
Pimentel and Burgess (2014)	Book chapter	English	No	22	X	X	X		United States	1
Pimentel and Greiner (1997)	Book chapter	English	No	22	X	X	X		United States	6
Pimentel and Hart (2001)	Book chapter	English	No	22	X	X	X		United States	11
Pimentel et al. (1980a)	Sci Journal	English	Yes	22	X	X	X		United States	181
Pimentel et al. (1980b)	Book chapter	English	No	22	X	X	X		United States	11

Pimentel et al. (1991a)	Book chapter	English	No	22	X	X	X		United States	[a]
Pimentel et al. (1991b)	Sci Journal	English	Yes	22	X	X	X		United States	136
Pimentel et al. (1992)	Sci Journal	English	Yes	22	X	X	X		United States	632
Pimentel et al. (1993a)	Book chapter	English	No	22	X	X	X		United States	133
Pimentel et al. (1993b)	Sci Journal	English	Yes	22	X	X	X		United States	93
Pingali et al. (1994)	Sci Journal	English	Yes	23		X			Philippines	154
Pingali et al. (1995)	Book chapter	English	No	23		X			Philippines	31
Praneetvatakul et al. (2013)	Sci Journal	English	Yes	24	X	X	X		Thailand	7
Pretty et al. (2000)	Sci Journal	English	Yes	25	X	X	X		United Kingdom	537
Pretty et al. (2001)	Sci Journal	English	Yes	25	X	X	X		United Kingdom	225
Rola and Pingali (1993)	Book	English	No	23		X			Philippines	322
Soares and Porto (2009)	Sci Journal	English	Yes	26		X			Brazil	36
Soares and Porto (2012)	Sci Journal	English	Yes	26		X			Brazil	3
Soares et al. (2002)	Sci Journal	English	Yes	27		X			Brazil	14
Steiner et al. (1995)	Book chapter	English	No	28	X	X	X		United States	49
Tegtmeier and Duffy (2004)	Sci Journal	English	Yes	29	X	X	X		United States	179

(continued)

Table 2.2 (continued)

Reference	Source	Language	Peer Reviewed?	Dataset	Category of cost considered[a]				Country of the study	No. of citations in Google Scholar
					Reg.	COI	Env.	DE		
Waibel and Fleischer (1998)	Report	German	No	13	X	X	X		Germany	[a]
Waibel et al. (1999)	Sci Journal	English	Yes	13	X	X	X		Germany	27
Wilson (1999a)	PhD thesis	English	No	30		X		X	Sri Lanka	25
Wilson (1999b)	Report	English	No	30				X	Sri Lanka	0
Wilson (2000a)	Sci Journal	English	Yes	30		X			Sri Lanka	33
Wilson (2000b)	Report	English	No	30		X		X	Sri Lanka	2
Wilson (2002a)	Sci Journal	English	Yes	30		X			Sri Lanka	8
Wilson (2002b)	Book chapter	English	No	30				X	Sri Lanka	15
Wilson (2003)	Sci Journal	English	Yes	30		X		X	Sri Lanka	11
Wilson (2005)	Sci Journal	English	Yes	30				X	Sri Lanka	5
Yanggen et al. (2003)	Report	English	No	9		X			Ecuador	8

[a]*Reg., Env.*, COI and DE correspond to regulatory costs, environmental costs, cost-of-illness and defensive expenditures, respectively
[b]Does not appear in Google Scholar

given by respondents may be highly dependent on the way in which contextual information is presented (see Florax et al. 2005).

The literature on hidden internal and external costs of pesticide use thus consists of 61 papers published between 1980 and 2014, these papers being based on 30 independent datasets. The costs were evaluated using both market and non-market methods. Among these latter we chose to exclude studies based on stated willingness to pay.

2.4 Regulatory Costs

Regulations concerning pesticide use are laid down by government bodies and concern (i) mandatory actions that must be undertaken by users and consumers, (ii) governmental actions to organize and check compliance with mandatory actions, and (iii) the activity of governmental agencies associated with pesticide use, such as research agencies. These regulations entail monetary costs. In general, these costs are not included in the market price of the pesticides. They must therefore be paid subsequently, as externalities, by public authorities (hence by consumers and citizens), producers or users.

2.4.1 A Small Number of Studies

Regulatory costs were taken into account in 24 articles (Table 2.2): eight book chapters (7 written by Pimentel and coworkers), four reports (2 from the Hannover Pesticide Policy project), one non-reviewed journal article and 11 articles published in peer-review scientific journals. However, the estimates given in several articles were partly or fully based on the same dataset. This was the case of all papers written by Pimentel and coworkers. It was also the case for Praneetvatakul et al. (2013), who actualized some of the costs originally estimated by Jungbluth (1996). We identified 15 different estimates, but only nine fully independent datasets (Table 2.3).

2.4.2 A High Diversity of Costs

Both external and internal costs are associated with the testing and registration, production, distribution – including importation, transport and sales – use and disposal of pesticides. The external costs are the economic burden to the public authorities responsible for organizing controls and checks on the compliance of stakeholders, e.g. public authorities, consumers, sellers and producers, with the regulations. The internal costs are the monetary subsidiaries paid by pesticide

Table 2.3 Economic costs due to regulations governing pesticide use

Reference	Country	Year	Fully independent dataset[a]	Overall costs (million US$ 2013 per year)
Houndekon and De Groote (1998); Houndekon et al. (2006)	Niger	1996	A	0.15
Ajayi et al. (2002)	Mali	1999	B	1.58
Khan et al. (2002)	Pakistan	2002	C	9.71
Fleischer (1999); Waibel and Fleischer (1998); Waibel et al. (1999)	Germany	1996	D	168.26
Pretty et al. (2000, 2001)	United Kingdom	1996	E	318.51
Praneetvatakul et al. (2013)	Thailand	2010	F	357.28
Pimentel et al. (1980a, b)	United Sates	1980	G	491.96
Jungbluth (1996)	Thailand	1995	F	558.33
Pimentel et al. (1991a, b)	United States	1991	G	2372.34
Steiner et al. (1995)	United States	1991	H	3203.00
Pimentel and Hart (2001)	United States	2001	G	3451.19
Pimentel and Greiner (1997)	United States	1997	G	3751.06
Pimentel (2005, 2009); Pimentel and Burgess (2014)	United States	2005	G	4229.13
Pimentel et al. (1992, 1993a, b)	United States	1992	G	4319.01
Tegtmeier and Duffy (2004)	United States	2002	I	4988.69

[a]The same letter indicates a partial dependence of cost estimates

handlers, e.g. users, sellers and producers, when they have to comply with mandatory regulations (Ajayi et al. 2002).

The various types of regulatory costs considered in the 24 articles investigating those costs are given in Table 2.4. The sources of these costs were highly diverse, including campaigns to raise public awareness of the impact of pesticides, monitoring and control, and public research on pesticides. The considerable diversity of these items may go some way to explaining why none of the studies considered the entire set of costs and heterogeneity in the costs considered by the various studies. Several articles listed a large number of qualitatively different regulatory costs, but estimates were frequently lacking. For instance, Ajayi et al. (2002) mentioned extension services as one of the externalities of pesticide use, but they provided no estimate of the costs involved. Waibel et al. (1999) also considered several costs, including the costs of removing contaminated products from the market and the cost of administrative activities, e.g. laws and decrees, and researches, but these

Table 2.4 Types of regulatory costs for pesticide use

Type of cost	Ajayi et al. (2002)	Houndekon et al. (2006)	Jungbluth (1996)	Praneetvatakul et al. (2013)	Khan et al. (2002)	Steiner et al. (1995)	Tegtmeier and Duffy (2004)	Pimentel et al. (1980a, b)	Pimentel et al. (1991a, b)	Pimentel et al. (1992, 1993a, b)	Pimentel and Greiner (1997)	Pimentel and Hart (2001)	Pimentel (2005); Pimentel and Burgess (2014)	Pretty et al. (2000, 2001)	Fleischer (1999); Waibel and Fleischer (1998); Waibel et al. (1999)
Pesticide registration, regulation and market monitoring	X		X	X		X									X
Public awareness campaigns on pesticide impact					X									X	X
Disposal of obsolete and leftover pesticides	X	X				X									
Farm work safety						X									
Control & monitoring															
Crop and/or food										X	X	X	X	X	X
Water (surface, underground and/or wells)						X	X		X	X	X	X	X	X	X
Livestock														X	
Wildlife								X	X						
Undefined			X	X				X	X	X	X	X	X		
Water decontamination					X					X	X	X	X	X	X
Public research on pesticides			X	X											
Extension services			X												X
Economic shortfall															
Crop			X	X	X			X	X	X	X	X	X		
Water	X														X
Livestock								X	X						
Milk								X	X						
Fishing								X	X						

costs were not quantified. Differences in public regulations between countries also underlie the considerable differences in the items considered between papers. For instance, Khan et al. (2002) pointed out that there was no monitoring program in Pakistan in 2002.

Most papers took into account the economic shortfall of crops exceeding the maximum residue limit or the costs of controls and monitoring (Table 2.4). Water decontamination, the regulation of pesticide registration and market monitoring costs were estimated in a small number of papers (Table 2.4) (Fig. 2.2). Other costs, such as those associated with governmental public information campaigns, economic shortfalls for water exceeding the maximum residue limit and public research on pesticides, were considered and estimated even less frequently (Table 2.4). However, these costs may account for a large proportion of the external costs of pesticides. For instance, public information campaigns accounted for about 10 % of the total external costs estimated by Khan et al. (2002) in the Pakistan, and public research costs were estimated at about 10 % of the total external costs by Praneetvatakul et al. (2013) in Thailand.

Finally, some costs, such as the time and money spent establishing regulations, have never been estimated. This is unfortunate, because it has been acknowledged that such costs may be high, due to the need for research and development, expert advice and a number of official tests (Ajayi et al. 2002; Waibel et al. 1999).

2.4.3 Estimated Costs

Estimates of total annual regulatory costs vary considerably, from US$150,000 (2013) in Niger (Houndekon and De Groote 1998; Houndekon et al. 2006) to US$5 billion (2013) in the United States (Tegtmeier and Duffy 2004) (Table 2.3). We did not carry out a meta-analysis to find the cause of this variation. However, as a first approximation, we can consider this variation to be due to the differences in the categories of costs considered, the detailed composition of each category and the geographic scale of the study. The costs of commonly considered categories were particularly variable and depended strongly on the subcategories included. For instance, monitoring and control costs were frequently considered, but different aspects of these costs were covered. The estimates obtained thus differed considerably between papers, depending, in particular, on whether or not they considered the control of underground water. For instance, Pimentel and coworkers began to consider the costs of monitoring underground water and wells in their papers published in 1991. The consideration of these costs led to an immediate increase in their estimates of the overall cost of pesticide regulations of more than 300 %, with these costs accounting for 90 % of total regulatory costs for pesticide use (Pimentel et al. 1991a, b). Water decontamination and economic shortfalls due to crop contamination have

Fig. 2.2 Peace River Manasota Regional Water Supply Authority Water Treatment Plant facility. Water decontamination corresponds to one of the several regulatory costs induced by pesticide use. Estimates of regulatory costs differed considerably between studies, depending, in particular, on whether or not they considered the control of underground water. For instance, in the United States, Pimentel and coworkers began to consider the costs of monitoring underground water and wells in their papers published in 1991. The consideration of these costs led to an immediate increase in their estimates of the overall cost of pesticide regulations of more than 300 %, with these costs accounting for 90 % of total regulatory costs for pesticide use (Pimentel et al. 1991a, b). Moreover Pimentel (2005, 2009) and Pimentel and Burgess (2014) estimated that the current monitoring of wells in the United States (about US$2 billion per year) would have reached US$17 billion per year if all the wells in the United States were monitored (Unmodified photography by Florida Water Daily, under Creative Common License CC BY (https://creativecommons.org/licenses/by/2.0/))

been taken into account by Pimentel et al. since 1992. These costs accounted for about 40 % of the externalities associated with pesticide use.

2.4.4 Actual Versus Theoretical Costs

Most estimates of regulatory costs were based on the actual expenditure of various stakeholders, including public authorities, manufacturers, distributors, sellers and farmers. No attempt was made to estimate non-monetary values. Due to the '*regulatory*' nature of these costs, estimates were generally based on the official budget reports of public agencies.

However, current costs may be much lower than the theoretical value. For instance, Pimentel (2005, 2009) and Pimentel and Burgess (2014) estimated the current monitoring of wells in the United States at about US$2 billion per year, but indicated that this cost would have reached US$17 billion per year if all the wells in the United States were monitored. Including these theoretical costs made a large difference, increasing the overall regulatory costs estimated by Pimentel (2005, 2009) and Pimentel and Burgess (2014) from US$4.2 billion to almost US$22 billion. Similarly, Jungbluth (1996) noted that costs related to pesticide residues in food in Thailand were difficult to estimate and were based on hypothetical scenarios rather than on real situations. In the absence of pesticide residue control for most food products, Jungbluth (1996) had to extrapolate the proportion of products exceeding the maximum residue limit from scarce data. Assuming that 10 % of all fruits and vegetables were above the maximum residue limit and assuming that these products would be unsaleable according to regulations, Jungbluth (1996) obtained a cost of about five billion Baht in 1996. He considered this value – corresponding to almost 90 % of the regulatory costs – as an upper limit for the costs truly paid by the corresponding stakeholders. Conversely, Jungbluth (1996) noted that if the maximum residue limit was not reached, then only the cost of control and monitoring should be taken into account, corresponding to 48.5 million Baht in 1996. This value should be taken as the lower limit of estimates. Along the same lines, Khan et al. (2002) distinguished between actual and potential costs. The potential costs they considered included the cost of establishing laboratories for pesticide residue analyses, residue monitoring programs, and training programs on the safe use of pesticides. These costs were largely theoretical, because there were no such activities in the region covered by their study in 2002, like in many developing countries (Ecobichon 1999). They reported the existence of regulations, but a lack of enforcement. They pointed out, in particular, that there was no comprehensive national monitoring system, and this may remain the case.

2.4.5 Conclusions

Regulatory costs, in particular, have been underestimated. We will see that this is also true for the other categories of "hidden" and external costs, but this underestimation may be particularly marked for regulatory costs. First, only 24 of the 61 articles assessing the external cost of pesticides included regulatory costs, and these 24 articles were actually based on only nine fully independent datasets. Second, each of these articles considered only a small number of regulatory costs. Finally, current costs are probably much lower than the costs that would have to be paid if the complete control, monitoring and decontamination of pesticide residues were to be undertaken and if all products exceeding the legal maximum residue limit had to be withdrawn from the market.

Although underestimated, regulatory costs could reach very large values such as US$4 billion (2013) yearly in the United States in the 2000s. Our analysis shows that if all regulations were respected, these costs would have jumped to US$22 billion (2013).

2.5 Human Health Costs

Despite strict regulations on the registration and use of pesticides, there are major concerns about their direct impact on human health following occupational exposure and the indirect exposure of non-occupationally exposed populations. Agricultural workers in fields and greenhouses are often occupationally exposed to pesticides, as they are responsible for preparing, mixing and loading pesticide preparations, spraying pesticides, sowing pesticide-treated seeds, harvesting sprayed crops, and cleaning and disposing of pesticide containers. Similarly, workers in the pesticide industry are also likely to experience occupational exposure. The families of farmers and other people living in rural areas in which pesticides are intensively used may also be indirectly exposed to these chemicals, through off-target pesticide drift from agricultural applications in particular (Lee et al. 2011) (Fig. 2.3). Finally, the overall population is also indirectly exposed to pesticides, through the consumption of food and drinking water contaminated with pesticide residues. Many pesticides can damage human health (Damalas and Eleftherohorinos 2011) and, for this reason, high doses over short periods (acute poisoning) and lower doses over longer periods of time (chronic exposure) may have an impact on human health. Karabelas et al. (2009) found that 84 of the 276 active substances authorized as plant protection products in Europe at the end of 2008 – 32 of the 76 fungicides, 25 of the 87 herbicides and 24 of the 66 insecticides – had at least one deleterious effect on health following acute and/or chronic exposure. These effects included acute toxicity, carcinogenicity, reproductive and neurodevelopmental disorders and endocrine disruption. Worldwide, pesticide use has resulted in thousands of cases of acute and chronic poisoning, with effects of varying severity on human health, from mild effects to death. In this section, we review the studies providing estimates of the economic consequences of human health impairment, from benign health damage to death, due to pesticide use.

2.5.1 Several Studies Based on a Limited Number of Datasets

We identified 57 articles providing monetary costs of the impact on health of pesticide exposure. These studies were published in diverse forms, including articles in scientific peer-reviewed journals (e.g. Choi et al. 2012), book chapters (e.g. Cole and Mera-Orcés 2003), PhD dissertations (e.g. Dung 2007), conference proceedings (e.g. Yanggen et al. 2003) and specific reports (e.g. Devi 2007). Some datasets were used as the basis of several publications. For instance, the dataset from the pioneering study by David Pimentel in the United States has been used in several

Fig. 2.3 Pesticides are sprayed in crop fields to protect them against agricultural pests. During these spray applications, these chemicals may disperse by drifting. They may therefore reach non-target crops in neighbouring fields, weakening these plants and reducing yields. Such crop injuries have been reported, in particular, for aerial applications of glyphosate (e.g. Ding et al. 2011; Reddy et al. 2010). Families of farmers and other people living in rural areas in which pesticides are intensively used may also be indirectly exposed to these pesticides, through this off-target pesticide drift from agricultural applications. After spraying, pesticides can also seep into the soil (Gil and Sinfort 2005; Pimentel 1995). Once in the soil, some soluble pesticides may be washed out in runoff water and during soil erosion, resulting in leaching into rivers and lakes (Chopra et al. 2011) (Unmodified photography by Santiago Nicolau, under Creative Common License CC BY-SA (https://creativecommons.org/licenses/by-sa/2.0/))

publications reporting either the same estimates (Pimentel and Greiner 1997; Pimentel and Hart 2001) or providing new estimates (Pimentel et al. 1992; Pimentel and Greiner 1997; Pimentel 2005, 2009) but describing the same types of cost. Similarly, the original dataset of Clevo Wilson (1999a) has been used in several articles in scientific journals and in several book chapters (e.g. Wilson 1999b, 2000a, b, 2002a, b, 2003, 2005). These 57 articles thus actually correspond to 29 independent cost-of-illness studies, starting with two papers by Pimentel et al. published in 1980 and ending with a book chapter written by Pimentel and Burgess and published in 2014 (Table 2.5). All 29 datasets involved cost-of-illness analyses, but they were produced by different methodologies (Table 2.5). Some focused on occupational exposures, notably those of the individuals spraying pesticides, whereas others focused on the pesticide exposure of the whole population. Some authors provided direct estimates of the various health costs, whereas other inferred health costs indirectly, by complex statistical modeling (Table 2.5).

2.5.2 Estimated Costs

The economic impact on human health has been evaluated per case, per farmer (or household), per rural establishment and at regional or national levels. The detailed costs reported in the 29 independent studies are shown in Table 2.6.

The costs of pesticide poisoning were evaluated at between about US$30 in Thailand and US$600 in Costa Rica (2013) per case, with each farmer/household using pesticides incurring annual costs of US$3 in China to US$187 in Sri-Lanka (2013) per year. In Central America, several authors have reported annual costs of US$32 to US$100 (2013) (see Vaughan (1993) and Villagrán (1976) cited by García (1998) and Castillo and Appel (1990) and Alvarado et al. (1998) cited by Cole et al. (2000)). These costs may be as high as US$850 (2013) per year for a rural establishment. At national level, health costs due to pesticide exposure have been estimated at US$1.1 million in Italy to about US$1.5 billion in the United States (2013) (Table 2.6).

These costs cannot be considered comparable, because they are influenced by several parameters, e.g. the type of pesticide used, the number of treatments applied, the degree to which farm staff spraying pesticides are protected etc., that may differ considerably between countries, with particularly marked differences between developed and developing countries. Moreover, in any given country, these costs have probably decreased over time, for two reasons. First, farmers have certainly become more aware of the effects of pesticide use on health and, therefore, probably protect themselves better against pesticide drifts. Second, some of the most dangerous pesticides have been withdrawn in many countries. Hence, on the one hand, costs actualized to 2013 values in US$ could easily be considered overestimates of current costs. On the other hand, human health costs were probably greatly underestimated at the time at which these reports were published, for three reasons. First, the frequencies of illness and death triggered by chronic exposure to pesticides have rarely been evaluated (see Sect. 2.5.5). Second, acute poisoning events generate various types of costs, and none of the studies performed to date has taken all these costs fully into account (see Sect. 2.5.3). Third, not all pesticide-poisoning events are recorded in databases or reported by farmers, particularly in developing countries (e.g. Lekei et al. 2014; Shetty et al. 2011). Indeed, some of the individuals carrying out pesticide spraying consider the symptoms of poisoning to be '*normal*' and do not, therefore, pay much attention to them.

Table 2.5 Cost-of-illness studies on pesticide exposure

				Type			Estimates from		Type of costs		
Reference	Country	Year	Method[a]	Acute	Chronic	Who?	Interviews or surveys	Databases	Treatment	Loss of work time	Death
Ajayi (2000)	Ivory Coast	1996/1997	DC	X		Farmers	X		X	X	
Ajayi et al. (2002)	Mali	2000	DC	X		Overall	X	X	X	X	X
All Pimentel and Pimentel et al. papers	United States	1980–2014	DC	X	X	Overall		X	X	X	X
Athukorala et al. (2012)	Sri Lanka	2007/2008	DC	X		Farmers	X		X	X	
Atreya (2005)	Nepal	2004	DC	X		Farmers	X		X	X	
Atreya (2007, 2008)	Nepal	2005	SI	X		Farmers	X		X	X	
Atreya et al. (2012, 2013)	Nepal	2008	SI	X		Farmers	X		X	X	
Brenna (2001)	Italy	1999	DC	X		Overall		X	X		
Choi et al. (2012)	South Korea	2009	DC	X		Overall		X	X	X	X
Crissman et al. (1994); Cole et al. (2000); Cole and Mera-Orcés (2003); Yanggen et al. (2003)	Ecuador	1991/1992	DC	X		Overall	X	X	X	X	
Corriols et al. (2008)	Nicaragua	2001	DC	X		Overall	X		X		

Devi (2007)	India	2004/2005	SI	X		Farmers	X		X	X	
Dung and Dung (1999); Dung (2007)	Vietnam	1996/1997	SI	X		Farmers	X		X	X	
Houndekon and De Groote (1998); Houndekon et al. (2006)	Niger	1996	SI	X	X	Farmers	X		X	X	
Huang et al. (2000)	China	2000	SI	X		Farmers	X		X		
Jansen et al. (1998)	Costa Rica	1993	DC	X		Farmers		X	X	X	
Jungbluth (1996)	Thailand	1995/1996	DC	X		Farmers		X	X	X	
Khan et al. (2002)	Pakistan	2002	DC	X		Farmers and industrial workers		X	X	X	X
Maumbe and Swinton (2003)	Zimbabwe	1998/1999	DC	X		Farmers	X		X	X	
Ngowi et al. (2007)	Tanzania	2005	DC	X		Farmers	X		X	X	
Pingali et al. (1994, 1995); Rola and Pingali (1993)	Philippines	1989–1991	SI	X	X	Farmers	X		X	X	
Praneetvatakul et al. (2013)	Thailand	2010	DC	X		Overall		X	X	X	
Pretty et al. (2000, 2001)	United Kingdom	1996	DC	X		Farmers	X		X	X	
Soares et al. (2002)	Brazil	1991–2000	Si	X		Rural workers	X		X	X	
Soares and Porto (2009, 2012)	Brazil	1998/1999	SI	X		Farmers		X	X	X	

(continued)

Table 2.5 (continued)

Reference	Country	Year	Method[a]	Type		Who?	Estimates from		Type of costs		
				Acute	Chronic		Interviews or surveys	Databases	Treatment	Loss of work time	Death
Steiner et al. (1995)	United States	1988	DC	X	X	Overall		X			
Tegtmeier and Duffy (2004)	United States	2002	DC	X	X	Overall		X	X	X	X
Fleischer (1999); Waibel and Fleischer (1998); Waibel et al. (1999)	Germany	1996	DC	X		Overall		X	X	X	X
Wilson (1999a, 2000a, b, 2002a, 2003)	Sri Lanka	1995/1996	DC	X	X	Farmers	X	X	X	X	

[a] *DC* direct cost and *SI* statistically inferred

Table 2.6 Estimated costs of the impact of pesticide exposure on health

Estimate	Country	Year	US$ 2013 (per year)	Reference
Per case	Ecuador	1992	44.71	Crissman et al. (1994); Cole et al. (2000); Cole and Mera-Orcés (2003); Yanggen et al. (2003)
	Thailand	1995	32.30	Jungbluth (1996)
	Nicaragua	2001	134.93	Corriols et al. (2008)
	Tanzania	2005	<1 to 400	Ngowi et al. (2007)
	Costa Rica	1993	591.18	Jansen et al. (1998)
Per farmer/household per year	China	2000	2.63	Huang et al. (2000)
	Nepal	2005	3.10	Atreya (2007, 2008)
	Ivory Coast	1996	11.88	Ajayi (2000)
	Nepal	2008	16.24	Atreya et al. (2012, 2013)
	Vietnam	1996	25.53	Dung and Dung (1999); Dung (2007)
	Sri Lanka	2007	15.66	Athukorala et al. (2012)
	Nepal	2004	35.06	Atreya (2005)
	Philippines	1989	82.37	Pingali et al. (1994, 1995); Rola and Pingali (1993)
	India	2004	104.52	Devi (2007)
	Brazil	2000	168.37	Soares et al. (2002)
	Sri Lanka	1995	187.36	Wilson (1999a, 2000a, b, 2002a, 2003)
	Zimbabwe	1998	9.42	Maumbe and Swinton (2003)
Per rural establishment per year	Brazil	1998	5.05–1072.09	Soares and Porto (2009, 2012)

(continued)

Table 2.6 (continued)

Estimate	Country	Year	US$ 2013 (per year)	Reference
At national level per year	Italy	1999	1,100,000	Brenna (2001)
	Thailand	1995	1,260,000	Jungbluth (1996)
	United Kingdom	1996	2,297,000	Pretty et al. (2000, 2001)
	Mali	1999	3,714,000	Ajayi et al. (2002)
	Thailand	2010	2,990,000	Praneetvatakul et al. (2013)
	Niger	1996	4,397,000	Houndekon and De Groote (1998); Houndekon et al. (2006)
	Nicaragua	2001	4,058,000	Corriols et al. (2008)
	South Korea	2009	7,792,000	Choi et al. (2012)
	Germany	1996	17,986,000	Fleischer (1999); Waibel and Fleischer (1998); Waibel et al. (1999)
	Sri Lanka	2007	35,885,000	Athukorala et al. (2012)
	Pakistan	2002	78,064,000	Khan et al. (2002)
	Sri Lanka	1995	21,636,000–130,044,000	Wilson (1999a, 2000a, b, 2002a, 2003)
	United States	1988	246,290,000–486,510,000	Steiner et al. (1995)
	United States	1991	439,320,000	Pimentel et al. (1991a, b)
	United States	1980	581,180,000	Pimentel et al. (1980a, b)
	United States	2002	1,308,890,000	Tegtmeier and Duffy (2004)
	United States	1992	1,326,450,000	Pimentel et al. (1992, 1993a, b)
	United States	1997	1,367,040,000	Pimentel and Greiner (1997); Pimentel and Hart (2001)
	United States	2005	1,492,960,000	Pimentel (2005, 2009); Pimentel and Burgess (2014)

2.5.3 Non-fatal Cases of Acute Poisoning

Acute poisoning, leading to respiratory, gastrointestinal, allergic, and neurologic disorders, is commonly reported by farmers, and particularly by those carrying out pesticide applications (e.g. Hudson et al. 2014; Kishi et al. 1995). For instance, in a broad survey performed in 2010, Lee et al. (2012) found that 25 % of South Korean male farmers had suffered acute occupational pesticide poisoning, suggesting that there may be more than 200,000 cases per year across South Korea. About 12 % of these pesticide-poisoning cases led to the consultation of a medical doctor or hospitalization (Lee et al. 2012). In the United States, the incidence of pesticide poisoning events requiring medical care among the 3,380,000 agricultural workers is thought to be between 10 and 600/100,000 (Calvert et al. 2008 and references therein), corresponding to about 300–20,000 cases annually.

All the cost-of-illness studies took acute poisoning events into account, but they considered very different types of costs associated with such poisoning events. Both indirect and direct costs were incurred. Direct costs are paid either by the farmers themselves or by the society, if, for example, hospital admission is free of charge. Indirect costs correspond to the working time lost by poisoned individuals and their families during and after the poisoning event. This time, which many farmers may not have considered – 90 % in the study by Athukorala et al. (2012) –, can be converted into wage loss and, therefore, into a monetary cost. All cost-of-illness studies took the cost of hospitalization and/or doctor fees into account (Table 2.7). By contrast, the costs of medication and of transport to and from hospital visits and medical consultation were explicitly included in only two thirds and one third, respectively, of the studies (Table 2.7). The economic burden due to the number of days taken off work to recover from poisoning events is the indirect cost classically identified in cost-of-illness studies. Almost all studies included this cost, paid by farmers, and some found that it outweighed, by far, the direct cost of acute poisoning (e.g. Wilson 1999a, 2000a, b, 2003)

However, absence from work to recover from illness is only one of the various indirect costs associated with pesticide poisoning. Indeed, Wilson (1999a, 2000a, b, 2002a, 2003), who generated what is probably the most comprehensive and complete list of indirect costs to date, also identified (i) a decrease in productivity for farmers not taking time off from work to recover and just after their return to work, (ii) impaired decision-making and (iii) a loss of leisure time (Table 2.7). However, he recognized that it would be difficult to estimate the number of leisure hours lost and the decrease in working efficiency. Leisure hours were defined as '*any time spent at home after work, such as time spent reading a newspaper, watching television, listening to the radio, playing a game or practicing a hobby, or time spent with the family*'. As suggested by Becker (1965), Wilson evaluated leisure time costs on the basis of the hourly wage, given that any loss of leisure time would be likely to affect productivity at work.

Decreases in productivity at work and in decision-making abilities were estimated in a few other cost-of-illness studies (Table 2.7). However, none of these

Table 2.7 Types of costs taken into account for acute pesticide poisoning in cost-of-illness studies

Type of costs	Atreya (2005)	Atreya (2007, 2008)	Atreya et al. (2012, 2013)	Wilson (1999a, 2000a, b, 2002a, 2003)	Ajayi (2000)	Crissman et al. (1994); Cole et al. (2000); Cole and Mera-Orcés (2003); Yanggen et al. (2003)	Choi et al. (2012)	Devi (2007)	Ngowi et al. (2007)	Ajayi et al. (2002)	Dung and Dung (1999); Dung (2007)	Fleischer (1999); Waibel and Fleischer (1998); Waibel et al. (1999)	Houndekon and De Groote (1998); Houndekon et al. (2006)	Jansen et al. (1998)	Athukorala et al. (2012)	Pingali et al. (1994, 1995); Rola and Pingali (1993)	Maumbe and Swinton (2003)	Khan et al. (2002)	All Pimentel and Pimentel et al. articles	Tegtmeier and Duffy (2004)	Jungbluth (1996)	Praneetvatakul et al. (2013)	Pretty et al. (2000, 2001)	Soares et al. (2002)	Soares and Porto (2009, 2012)	Brenna (2001)	Corriols et al. (2008)	Huang et al. (2000)	Steiner et al. (1995)
Direct costs																													
Hospitalization and/or doctors' fees	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	[c]	[d]	[e]
Treatment (drugs, local healing, local cure)	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X												
Transport to and from hospital/doctor	X	X	X	X	X	X	X	X	X									X											
Indirect costs																													
Loss of working days whilst recovering	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	[a]	X	X	X	X	X	X	X	X				

Lower productivity if remaining at work and on return to work	X	X	X	X	X					X	X																		
Impairment of decision-making abilities	X	X	X	X																									
Time taken by family members to nurse the victim of illness	X	X	X			X																							
Childcare												X																	
Loss of leisure time				X																									
Cost of fatal poisoning incidents																													
Loss of work time/productivity							X			X																			
Cost of life												X						[b]	X	X									

[a]These costs were estimated in terms of the number of days lost, but the corresponding economic value was not estimated
[b]These authors took the cost of deaths into account. However, they did not specify which costs they took into account. They merely wrote '*Substantial amounts of 24 million in terms of treatment, 18 million for work loss during ailments and 224 million rupees for accidental deaths are estimated as the externality costs*'
[c]These authors provide the mean out-of-pocket costs (without details) for a single acute pesticide poisoning episode – they also indicate that poisoned farmers loss about 20 days of productive working time, without quantifying this cost
[d]No details are given, except that they took into account only the treatment costs for visible impairments (excluding chronic poisoning). However, it is not clear whether their costs include the loss of work. However, they clearly do not include the cost of life
[e]Unknown, because no details are provided about the different type of costs included in the estimates

other studies evaluated the loss of leisure time as in the study by Wilson. However, Wilson did not estimate all the indirect costs due to pesticide poisoning and recognized that '*the costs to the family were not taken into account*'. These costs, including the time taken by family members to nurse the victim of illness, were investigated in cost-of-illness studies performed in Nepal (Atreya 2005, 2007, 2008; Atreya et al. 2012, 2013) and Ecuador (Cole et al. 2000; Cole and Mera-Orcés 2003; Crissman et al. 1994; Yanggen et al. 2003). The cost of childcare, which was estimated by Fleischer and coworkers (Table 2.7), is another indirect cost that was not considered by Wilson. Finally, an additional indirect cost, identified but not estimated by Devi (2007), is the time spent traveling to seek medical help. Thus, none of the cost-of-illness studies performed to date fully took into account all the various costs associated with acute pesticide poisoning.

2.5.4 Fatal Cases of Acute Poisoning

Suicide accounts for most of the fatal cases of acute poisoning. Gunnell et al. (2007) estimated that 250,000 people die from voluntary pesticide ingestion each year, accounting for 30 % of all suicides. The costs associated with such deaths cannot be considered an externality of pesticide use. Nevertheless, accidental pesticide poisoning, mostly in the occupational setting, may be fatal in some cases and the costs associated with such deaths can be treated as external costs. Fatal accidents due to occupational pesticide poisoning are very rare in some countries, such as the United States (1 case recorded from 1998 to 2005, Calvert et al. 2008), but may concern several tens or hundreds of workers per year in other countries with higher levels of pesticide use or in which workers are less well equipped with personal protection equipment (Fig. 2.1). For instance, Santana et al. (2013) reported that 2052 deaths, excluding homicides and suicides, were recorded as due to pesticide poisoning in Brazil, between 2000 and 2009. Half of these deaths concerned agricultural workers and most of them were caused by poisoning with organophosphate and carbamate pesticides.

The cost of fatal cases of accidental poisoning was estimated in only six sets of cost-of-illness studies: Ajayi et al. (2002), Choi et al. (2012), Khan et al. (2002), Tegtmeier and Duffy (2004), Pimentel and coworkers and Fleischer and coworkers (Table 2.7). Fatal cases have generally been ignored, mostly due to the type of cost-of-illness studies performed. Indeed, several of these studies involved interviews with a sample of farmers about the costs they incurred during pesticide poisoning incidents (Table 2.5). By definition, studies of this type cannot take deaths into account and, therefore, did not assess the cost of fatal poisoning events.

Two studies estimated the cost of these deaths, by evaluating the corresponding loss of work time. Ajayi et al. (2002) economically quantified the loss of life as the decrease in agricultural gross domestic product per habitant during the mean duration of an economically active life in agriculture set, in their study, at 50 % of 30 years. Similarly, Choi et al. (2012) estimated the loss of productivity loss due to premature death. Age- and sex-specific mean wages and employment rates were used as surrogates for per capita productivity for each sex and age group. Like Ajayi et al. (2002), Khan et al. (2002) included fatal injuries in their overall estimate of health costs. They attributed an overall cost of 224 million Rupees (US$15.1 million (2013)) to such injuries, but provided no details about how this cost was estimated.

David Pimentel and coworkers also considered the cost of fatal cases of pesticide poisoning. They used different sources for their estimates, based on the reasoning that no-one can place a precise monetary value on a human life. In their first estimate, Pimentel et al. (1980a, b) estimated the value of an individual human life at about US$1 million (about US$3.2 million (2013)). This value was considered to be the amount of money that industry and government might reasonably spend to prevent a death, but Pimentel et al. (1980a, b) wrote that '*obviously it is much less than the true value of a human life*'. In their article published in 1992, Pimentel et al. used the monetary ranges computed by the insurance industry and used an estimate of US$2 million (about US$3.4 million (2013)), which they considered to be conservative. Pimentel and Greiner (1997) and Pimentel and Hart (2001) used an estimate of US$2.2 million (about US$3.2 million (2013)) per human life, corresponding to the mean value of the damages paid to the surviving spouses of slain policemen in New York City, which they again considered to be a conservative estimate. Finally, Pimentel (2005, 2009) and Pimentel and Burgess (2014), in their most recent re-evaluation of pesticide externalities, used the United States Environmental Protection Agency standard of US$3.7 million (about US$4.7 million (2013)) per human life. Finally, Fleischer and coworkers estimated the cost of acute fatal poisoning events in Germany, using the estimate of US$2 million per life taken by Pimentel et al. (1993a) (see Waibel and Fleischer 1998).

2.5.5 *The (Almost) Uncounted Costs of Chronic Exposure*

The most striking feature of cost-of-illness studies on pesticide use is the lack of data concerning the long-term effects of chronic exposure. Several studies have highlighted the possible occurrence of severe health impairment, e.g. cancers, diabetes, depression, neurological deficits, respiratory diseases, fertility problems, cutaneous effects, effects on the unborn embryo, blindness, polyneuropathy, associated with chronic exposure to these chemicals. However, only six estimated the monetary costs of such impairment (Table 2.5). The other studies mostly stated that it was not possible to estimate costs due to chronic exposure because the corresponding illnesses, such as cancers, are multifactorial, making it difficult to estimate the number of cases directly due to pesticide exposure.

The six studies including the costs of health impairment due to chronic exposure provided very rough and incomplete estimates. Steiner et al. (1995) merely considered the cost of chronic illnesses to be as high as that associated with acute poisoning. Pimentel and coworkers based their estimates of the costs of chronic pesticide exposure on a rough estimate of the number of cancers per year. This number varied from 0.5 % of all cancers (Pimentel et al. 1980a, b, 1991a) to 6000 (Pimentel et al. 1991b), <10,000 (Pimentel et al. 1992, 1993a, b), <12,000 (Pimentel and Greiner 1997), 10,000 (Pimentel and Hart 2001) and between 10,000 and 15,000 cases (Pimentel 2005, 2009; Pimentel and Burgess 2014). All but one of these estimates were based on a personal communication from David Schottenfeld indicating that '*US cases of cancer associated with pesticides in human are less than 1 % of the nation's total cancer cases*' (see Pimentel et al. 1980a, 1992). Tegtmeier and Duffy (2004) did not provide another estimate for the United States: they incorporated the estimate of Pimentel et al. (1992) into their overall externalities of pesticide use. Houndekon and De Groot (1998) and Houndekon et al. (2006) took chronic exposure into account to some extent in their estimates, but it is impossible to determine to what extent. Indeed, they asked farmers how much money they spent on medication and medical consultations and how many working days per year they lost to illness, without specifying the type of health effect (acute or chronic and, for chronic effects, the illnesses concerned). Similarly, Pingali et al. (1994, 1995) and Rola and Pingali (1993) performed medical tests, providing an assessment of the ailments of each farmer or respondent and their seriousness. Such ailments may or may not be related to chronic exposure to pesticides. Finally, Wilson (1999a, 2000a, b, 2003) considered long-term illness diagnosed by a physician as arising from pesticide exposure. Given the small number of farmers examined ($n=203$), long-term illnesses were probably underdetected.

This lack of counts is certainly the major flaw of all cost-of-illness studies performed to date. Indeed, there are good reasons to think that the costs of chronic exposure may be not only as high as those of acute poisoning, as stated by Steiner et al. (1995), but probably higher. One reason for this is that sufferers of irreversible illnesses, e.g. blindness, not only undergo short-term treatments, but may also incur long-term costs over a number of years, sometimes until they die. In their most recent re-evaluation of externalities, Pimentel (2005, 2009) and Pimentel and Burgess (2014) estimated the costs of chronic exposure to pesticides, restricted to cancers, reached US$1 billion, a value four times that estimated for the cost of acute poisoning events. However, this estimate did not include the loss of working days and the cost of death. By taking a death rate of 20 % for people suffering from cancers (Siegel et al. 2014) and a rather conservative estimated 3 months of absence from work for cancer treatment and recovery, and using the same costs of death as for acute poisoning, the costs of chronic exposure estimated by Pimentel and coworkers would have reached US$10.2 billion per year in 2005, 45 times the cost of acute poisoning.

2.5.6 Conclusions

The cost-of-illness studies reviewed here clearly show that the external costs relating to human health associated with pesticide use have always been strongly underestimated. First, most studies considered only the costs associated with short-term effects following acute poisoning events. This resulted in a considerably lower estimate of the overall costs, because severe illnesses, e.g. cancers, diabetes, depression, blindness, potentially triggered by chronic pesticide exposure are probably associated with much higher costs than acute poisoning incidents. The few studies to have taken serious illnesses into account yielded only partial and very crude estimates, for only one of the multiple possible illnesses, cancers, and only some of the costs concerned. Moreover, the cost-of-illness studies generally ignored several direct and indirect costs due to acute poisoning.

Another major flaw in cost estimates to date is the lack of consideration of fatal cases of pesticide exposure. Pesticide exposure-related deaths have sometimes been counted for assessments of accidental acute poisoning incidents, but deaths due to chronic pesticide exposure have been completely ignored. Indeed, even though some authors, such as Pimentel et al. estimated the number of cancers, they did not estimate the corresponding number of deaths. In addition, the value of life has probably been underestimated in the past. Pimentel and coworkers increased the estimate of this cost from US$1 to 3.7 million between 1980 and 2005, but, surprisingly, they retained this value (the value provided by the United States Environmental Protection Agency in the early 2000s) in their reassessments published in 2009 (Pimentel 2009) and 2014 (Pimentel and Burgess 2014). There is no standard concept or tool for placing a precise monetary value on a human life, but the reviews and meta-analyses of Kniesner et al. (2012), Lindhjem et al. (2011), Viscusi and Aldy (2003), and Viscusi et al. (2014) converged on a mean of US$9 to 10 million in 2013, which would correspond to a value of US$7.4 million in 2005. The human health costs estimated by Pimentel (2005, 2009) and Pimentel and Burgess (2014) should therefore be re-evaluated. If we use the re-evaluation of the estimated cost of chronic pesticide exposure of Pimentel (2005) proposed above, then overall human health costs in the article published by Pimentel in 2005 would have reached US$15.65 billion (2005), rather than US$1.23 billion (2005) as originally estimated.

Our review shows that health costs studies generally did not take into account fatal cases due to chronic exposure such as fatal outcomes of cancers. Doing so would increase those health costs by up to tenfold, e.g. US$15 billion instead of US$1.5 billion (2013) in the United States in 2005.

2.6 Environmental Costs

We found 26 articles providing 15 different monetary estimates of environmental impacts of pesticide use (Table 2.8). These studies, based on 11 fully independent datasets, either focused on a particular impact or attempted to provide a complete valuation of these impacts. Not only are there only a limited number of studies on this topic, but most were carried out in the 1990s. We found only five studies based on data recorded after 2000 and only one article published since 2006 (Table 2.8).

Table 2.8 Costs of the environmental impact of pesticide use

Reference	Country	Year	Fully independent dataset[a]	Overall costs (million US$ 2013 per year)
James (1995)	Canada	1993	A	0.27–30.73
Houndekon and De Groote (1998); Houndekon et al. (2006)	Niger	1996	B	0.89
Jungbluth (1996)	Thailand	1995	C	5.58
Fleischer (1999); Waibel and Fleischer (1998); Waibel et al. (1999)	Germany	1996	D	9.31
Praneetvatakul et al. (2013)	Thailand	2010	E	16.88
Ajayi et al. (2002)	Mali	1999	F	38.11
Pretty et al. (2000, 2001)	United Kingdom	1996	G	62.74
Steiner et al. (1995)	United States	1991	H/J	203.85–4029.46
Khan et al. (2002)	Pakistan	2002	I	815.12
Pimentel et al. (1991a, b)	United States	1991	J	948.94
Tegtmeier and Duffy (2004)	United States	2002	K/J	1469.74–1507.62
Pimentel et al. (1980a, b)	United States	1980	J	1621.17
Pimentel (2005, 2009); Pimentel and Burgess (2014)	United States	2005	J	5973.50
Pimentel and Greiner (1997); Pimentel and Hart (2001)	United States	1997	J	6993.99
Pimentel et al. (1992, 1993a, b)	United States	1992	J	7967.84

[a]The same letter indicates a partial dependence of cost estimates

2.6.1 Various Types of Environmental Impact

Several types of environmental impact have been considered, but there have been few attempts to classify these impacts into a particular framework (but see Khan et al. 2002). In addition, the costs of these environmental impacts were poorly differentiated from regulatory costs. For instance, several authors considered water monitoring costs and the costs of water decontamination to be costs associated with environmental impact (Pimentel et al. 1980a, b, 1991a, b, 1992, 1993a, b; Pimentel and Greiner 1997; Pimentel and Hart 2001; Pimentel 2005, 2009; Pimentel and Burgess 2014). In this review, we have considered the impact of pesticide use on surface and underground waters as regulatory costs, because these controls and decontamination processes are, in most countries, mandatory. Similarly, the costs of crops and livestock (meat, milk, eggs etc.) contaminated with pesticides to levels exceeding the maximum residue limit, resulting in their mandatory withdrawal from the market and destruction, are considered here as regulatory rather than environmental costs. Finally, we found that environmental impacts could be classified into two main categories: (i) damage to animals (vertebrates and invertebrates), plants, algae and microorganisms and (ii) pest resistance to pesticides (Table 2.9).

2.6.1.1 Damage to Animals, Plants, Algae and Microorganisms

The main environmental impact of pesticides is probably the direct or indirect damage they cause to animals, plants and microorganisms, varying from minor injuries to death. This impact is not restricted to the area in and around fields. Indeed, during applications, pesticides drift away in the air and seep into the soil (Gil and Sinfort 2005; Pimentel 1995). Once in the soil, some soluble pesticides may be washed out in runoff water and during soil erosion, resulting in leaching into rivers and lakes (Chopra et al. 2011).

Damage to Vertebrates

Pesticide use has two main unintentional effects on vertebrate (mammals, birds, fish, reptiles and amphibians) wildlife: (i) deaths due to direct or indirect, e.g. feeding on contaminated plants and/or prey, exposure to high doses and (ii) poorer survival, growth and reproduction due to exposure to sublethal doses and a decline in or the elimination of habitats and food sources due to pesticides (Gibbons et al. 2014; Guitart et al. 2010; Sánchez-Bayo 2011).

Pesticides have a particularly strong impact on birds (Mitra et al. 2011), through direct deaths and the reduction or elimination of habitats and food sources. The indirect effects of insecticides, herbicides and fungicides have been identified as one of the main factors contributing to the decline of farmland birds in several European countries (Geiger et al. 2010). For example, herbicides and insecticides, together

Table 2.9 Environmental impacts of pesticide use and associated costs

Environmental impact	Associated economic costs	Pimentel et al. (1980a, b)	Pimentel et al. (1991a, b)	Pimentel et al. (1992, 1993a, b)	Pimentel and Greiner (1997); Pimentel and Hart (2001)	Pimentel (2005, 2009); Pimentel and Burgess (2014)	Tegtmeier and Duffy (2004)	Steiner et al. (1995)	Khan et al. (2002)	Fleischer (1999); Waibel and Fleischer (1998); Waibel et al. (1999)	Pretty et al. (2000, 2001)	Ajayi et al. (2002)	Jungbluth (1996)	Praneetvatakul et al. (2013)	Houndekon and De Groote (1998); Houndekon et al. (2006)	James (1995)
Damage to animals, plants and microorganisms																
Crops/cultivated plants/trees	Decrease in yields	X	X	X	X	X										
	Insurance of the person applying the pesticide	X	X	X	X	X										
Wild plants (other than weeds)	Loss of biodiversity									X						
Domestic animals and livestock	Illnesses due to poisoning (veterinary costs)	X	X	X	X	X			X							
	Mortality due to poisoning	X	X	X	X	X			X						X	
	Loss of productivity in weakened animals								X							
Fish	Loss for fish	X	X	X	X	X	X	X								
	Decrease in recreational value for fishermen	X	X													

Birds	Loss of birds (average cost per bird)			X	X	X	X	X								
	Loss of pest control ensured by birds															X
	Monitoring of birds	X	X													
	Re-establishment programs	X	X	X	X											
Arthropods (other than bees)	Increase in pesticide use due to a loss of natural enemies	X	X	X	X	X	X		X			X	X	X		
	Decrease in yields due to the resurgence of pests or the emergence of secondary pests			X	X	X	X									
Microorganisms	Increase in pesticide use due to a loss of natural enemies			X	X	X	X									
Bees	Colony losses	X	X	X	X	X	X	X	X	X	X					
	Honey and wax losses	X	X	X	X	X	X	X	X			X				
	Loss of potential honey production	X	X	X	X	X	X	X								
	Pollination losses	X	X	X	X	X	X	X	X							
	Bee rental for pollination	X	X	X	X	X	X	X								
Pest resistance to pesticides	Increase in insecticide use	X	X	X	X	X			X			X	X			
	Increase in pesticide (other than insecticide) use			X	X	X			X							
	Yield losses due to incomplete control of resistant pests			X	X	X			X							

with certain agricultural practices, decrease levels of cereal grains, weed seeds and arthropods, thereby potentially contributing to the decline of bird species dependent on these resources for survival, e.g. Wilson et al. (1999) for granivorous birds and Hallmann et al. (2014) for insectivorous birds. In North America, the decline of several grassland birds, including songbirds in particular, is thought to be mostly due to a direct impact of insecticides (Mineau (2002) and Mineau et al. (2005) for Canada; Mineau and Whiteside (2006, 2013) for the United States). Birds are particularly susceptible to cholinesterase-inhibiting pesticides, e.g. organophosphates and carbamates, mostly because, unlike mammals, they have low levels of anticholinesterase detoxifying enzymes (Walker 1983). The extensive use of carbofuran, a carbamate, through a granular form resembling plant grains in North America has been reported to lead to the death of millions of birds annually (Mineau et al. 2012) (Fig. 2.4). Other birds, such as those predating on rodents, e.g. owls and other birds of prey, are also directly or indirectly poisoned by rodenticides in many developed countries (Christensen et al. 2012; Elliott et al. 2014; Langford et al. 2013; Thomas et al. 2011).

Fig. 2.4 The extensive use of carbofuran, a carbamate, through a granular form resembling plant grains in North America has been reported to lead to the death of millions of birds – like the horned lark *Eremophila alpestris* – annually (Mineau et al. 2012). The ban on these granular formulations of carbofuran introduced in 1991 (Heier 1991) and effective by 1994, in particular, probably had a considerable beneficial effect on bird survival in farmland. The estimate of 17–91 million birds killed per year during the 1980s was therefore almost certainly, as stated by Mineau (2005), the "worst-case" impact of pesticides on birds in an agricultural setting'. The current impact of pesticide use on birds is probably much lower (Unmodified photography by Kelly Colgan Azar, under Creative Common License CC BY-ND (https://creativecommons.org/licenses/by-nd/2.0/))

Many studies have documented direct and indirect effects of both high and sublethal doses of pesticides on several wild vertebrates other than birds. Herbicide treatments can be lethal for amphibians. For instance, one of the surfactants added to glyphosate, the most widely used herbicide worldwide, has been shown to be highly toxic to several species of amphibians in North America (Relyea 2005). Recent reviews and meta-analyses have confirmed that several pesticides decrease amphibian survival (Baker et al. 2013; Egea-Serrano et al. 2012). It has also been shown that pesticides have indirect and sublethal effects on this class of vertebrates, reducing their growth (Baker et al. 2013; Egea-Serrano et al. 2012) and increasing the frequency of abnormalities (Egea-Serrano et al. 2012). For instance, the herbicide atrazine, one of the most commonly used pesticides worldwide, adversely affects amphibians by disrupting metamorphosis, reducing antipredator behavior, decreasing immune function and increasing the frequency of infection (Rohr and McCoy 2010). The endocrine disruptor activities of atrazine, which decreases both time to metamorphosis and size at metamorphosis, can be enhanced by the presence of insecticides and fungicides. The effects of such mixtures of pesticides have probably played a major role in the global decline of amphibians (Hayes et al. 2006). Atrazine also disrupts several life history traits in fish (Rohr and McCoy 2010). Several pesticides, including atrazine, have been shown to have immunotoxic effects (Dunier and Siwicki 1993) and to cause oxidative stress (Slaninova et al. 2009) in fish, and these compounds can also interfere with olfaction in these organisms (Tierney et al. 2010).

Finally, pesticides also injure wild and domestic mammals. Rodenticides, particularly second–generation compounds, kill not only target pests, but many non-target rodent species (Elliott et al. 2014; Fournier-Chambrillon et al. 2004). Species abundance and diversity in rodent communities can also be altered by herbicides, particularly in situations in which these chemicals are used to convert bushwood to grassland (Freemark and Boutin 1995). Pesticides can also poison several domestic mammals (Wang et al. 2007; Berny et al. 2010). In the United States, and probably also in many European countries, the incidence of poisoning is highest in cats and dogs (Berny et al. 2010). These animals often wander freely around homes and farms. They are therefore much more likely to come into contact with pesticides than other domesticated animals. The presence of sprayed chemicals on fodder or of pesticide residues in feed for livestock may lead to fatal poisoning events in domestic farm animals, particularly in developing countries (Ajayi et al. 2002).

Damage to Invertebrates

Insecticide treatments controlling pests also have damaging effects on many non-target terrestrial arthropods in agroecosystems, including the natural enemies (predators, parasites and parasitoids) of agricultural pests (Croft and Brown 1975). Damage to these species may be greater than initially thought, because such damage can occur even at low non-lethal doses of insecticides (Desneux et al. 2007). For instance, sublethal doses of neonicotinoids (a new generation of insecticides) have clearly been shown to affect the foraging success, survival, colony growth, and queen production of honey and bumble bees (Henry et al. 2012; Schneider et al. 2012; Whitehorn et al. 2012) (Fig. 2.5). Beneficial arthropods are also affected by herbicides. This impact may be direct (Norris and Kogan 2000), but it is generally indirect. By killing weeds and non-target plants, herbicides reduce the fitness of many of the arthropods developing or resting on weeds, thereby decreasing the growth of their populations (Freemark and Boutin 1995; Norris and Kogan 2005). Even if herbicides do not actually kill non-target plants, they may still suppress flower formation in some species (Schmitz et al. 2014a), or markedly delay flowering time and decrease flower production in many other species (Boutin et al. 2014). As a consequence, herbicide treatments may indirectly decrease the fitness of pollinating insects in non-crop habitats during periods in which crop plants are unavailable for pollination. Egan et al. (2014) showed that changes in the structure and function of arthropod communities depend on species composition, crop rotation patterns and the timing of herbicide exposure.

Pesticides can also have an impact on aquatic invertebrates (Rasmussen et al. 2013), particularly during pulses of contamination triggered by surface runoff and

Fig. 2.5 Honey bee on apple blossom in Bedfordshire, United Kingdom. Damage to non-target terrestrial arthropods in agroecosystems may be greater than initially thought, because such damage can occur even at low non-lethal doses of insecticides (Desneux et al. 2007). Sublethal doses of neonicotinoids (a new generation of insecticides) have clearly been shown to affect the foraging success, survival, colony growth, and queen production of honey and bumble bees (Henry et al. 2012; Schneider et al. 2012; Whitehorn et al. 2012) (Unmodified photography by Orangeaurochs, under Creative Common License CC BY (https://creativecommons.org/licenses/by/2.0/))

through tile drains during heavy rain. Invertebrates may also be injured during short pulses of contamination due to pesticide desorption from suspended solids or sediment particles. Finally, they can be poisoned via the ingestion of "polluted" particles. Several studies have found associations between pesticide concentrations and decreases in the numbers and abundances of taxa and changes to invertebrate community structure (e.g. Friberg et al. 2003; Liess and von der Ohe 2005; Schäfer et al. 2007, 2011, 2012). These studies were performed at many sites in Europe, Siberia and Australia, and the authors concluded that there was little doubt that pesticides were responsible for the observed changes in aquatic invertebrate communities. Liess and von der Ohe (2005) and Schäfer et al. (2007) showed that the number and abundance of aquatic invertebrate taxa could be compensated, probably through recolonization from undisturbed sections of the stream. Nevertheless, Beketov et al. (2013) found that pesticides had significant effects on regional species and family richness in Germany, France and Australia, with up to 42 % of the taxa from the recorded taxonomic pools lost. Furthermore, in Europe, effects were detected at concentrations considered environmentally benign in current legislation (Beketov et al. 2013).

Damage to Plants, Algae and Corals

Pesticides can accidentally injure crops. First, the crops protected by the pesticide may be damaged by it. In particular, some pesticides may disrupt photosynthesis, thereby decreasing both growth and yield. Such an effect has been shown for several fungicides, on many crops (Petit et al. 2012), and for some herbicides, on cotton (Reddy et al. 1990) and soybean (Hagood et al. 1980). Similarly, insecticide treatments may also lower yields when applied to lettuce (Toscano et al. 1982) and cotton (Youngman et al. 1990). Second, pesticides may disperse by drifting during spray applications. They may reach non-target crops in neighboring fields, weakening these plants and reducing yields. Such crop injuries have been reported, in particular, for aerial applications of glyphosate (e.g. Ding et al. 2011; Reddy et al. 2010). Third, as some herbicides persist in the soil, other crops (notably vegetables) in the rotation may be affected and display lower yields (e.g. Felix et al. 2007; Mahmoudi et al. 2011). These carryover injuries may be accentuated in fields previously treated with several herbicides. For instance, the addition of atrazine to mesotrione treatments in the year before planting has been shown to increase injury rates by 3–55 % in broccoli, carrot, cucumber, onion, and potato (Robinson 2008).

In some agroecosystems, field margins and boundaries (e.g. hedgerows, woodlots, etc.) are the only remaining habitats for many wild plant species, some of which are beneficial, considered of heritage value or protected (Türe and Böcük 2008). The long-term maintenance of their populations, particularly close to edges of crop fields, may be jeopardized by the drift of herbicide treatments. Several studies have shown that non-target plants are affected by herbicides (e.g. Freemark and Boutin 1995; Gove et al. 2007; Schmitz et al. 2014a), leading to short- and long-term changes in the richness and/or structure of plant communities (e.g. Egan et al.

2014; Gove et al. 2007; Schmitz et al. 2014b). Changes also occur among weed communities within crop fields (e.g. Andreasen and Streibig 2011). These changes in the composition of weed plant communities may reflect lower rates of reproduction in the species most affected by herbicides, as demonstrated by Boutin et al. (2014).

Aquatic plants, algae and coral species may also be affected by pesticide use. The large distances between sprayed fields and bodies of fresh and inshore waters should theoretically provide some protection, through the adsorption of some of the drift by bank vegetation and, probably, also through the dilution of the herbicides in water. In some ecosystems, aquatic and algal species are, indeed, considered to be not necessarily at risk (e.g. Cedergreen and Streibig 2005). However, there may be a major impact on aquatic species in bodies of water subject to intense agricultural runoff (Fabricius 2005). A textbook example is provided by the inshore waters of the Australian Great Barrier Reef. This lagoon has World Heritage status, but is widely contaminated with insecticides and herbicides (Haynes et al. 2000; Lewis et al. 2009; Packett et al. 2009). Kroon et al. (2012) estimated that >30,000 kg of herbicides enter the Great Barrier Reef lagoon each year. Despite their dilution in the water, concentrations exceeding 1 μg L^{-1} have been reported for some herbicides within the lagoon (Lewis et al. 2009). These concentrations may be high enough (Lewis et al. 2012) to have deleterious effects on corals (Cantin et al. 2007; Jones et al. 2003; Negri et al. 2011), seagrasses (Flores et al. 2013), foraminifera (van Dam et al. 2012), benthic microalgae (Magnusson et al. 2008, 2010, 2012) and coralline algae (Negri et al. 2011). The Great Barrier Reef is probably the most widely studied ecosystem threatened by pesticides, but other species in several other coastal water systems are also threatened by the effects of pesticide runoff. The ecosystems concerned include Chesapeake Bay in the United States (Hartwell 2011), the Seto Inland Sea (Balakrishnan et al. 2012) and two lagoons (Yamamuro 2012) in Japan.

Damage to the Soil Community

The effects of pesticides on earthworms (Yasmin and D'Souza 2010), microarthropods (Adamski et al. 2009), nematodes (Zhao et al. 2013), fungi (Morjan et al. 2002) and microorganisms (viruses, protozoa and bacteria) (Imfeld and Vuilleumier 2012; Lo 2010) within the soil may have major environmental consequences. The soil community plays a critical role in crop production and crop protection (Barrios 2007). These small organisms are essential to the functioning of all ecosystems, because they break down waste, thereby recycling the chemical elements required for life. Bacteria and fungi make nitrogen and other elements available to plants (Bonfante and Anca 2009) and, like nematodes, some soil-borne fungi are natural enemies of pest insects (Kaya and Gaugler 1993; Klingen and Haukeland 2006). Earthworms, which are widely recognized as 'ecosystem engineers', contribute to several ecosystem services through pedogenesis, the development of soil structure, water regulation, nutrient cycling, primary production, climate regulation, the remediation of pollution and cultural services (Blouin et al. 2013).

Damage Due to Interactions Between Species and Between Stressors

Species are not isolated from their environment or from other interconnected species. Pesticide exposure may, therefore, have indirect effects on biotic interactions, such as host-parasite relationships (Köhler and Triebskorn 2013). For instance, Rohr et al. (2008) showed that atrazine use was the best predictor of the abundance of larval trematodes (parasitic flatworms) in the declining northern leopard frog *Rana pipiens*. Pesticides can also increase the frequency of deformities associated with trematode infection in amphibians (Kiesecker 2002). More generally, interactions between pesticides and other environmental stressors may play a key role in the decline of amphibian populations (Mann et al. 2009). Synergistic effects of pesticides and natural stressors, such as heat, desiccation, oxygen depletion and pathogens, have already been documented in many other classes of animals (Holmstrup et al. 2010). Pesticides can also affect food webs and competition between species (Köhler and Triebskorn 2013). For instance, benomyl, a widely used fungicide, suppresses populations of arbuscular mycorrhizal fungi in grasslands, altering floral display at the patch level. Such changes have been shown to induce a shift in the community of floral visitors, from large-bodied bees to small-bodied bees and flies, and to decrease the total number of visits to flowers (Cahill et al. 2008).

2.6.1.2 Pest Resistance to Pesticides

The second main environmental consequence of pesticide use is the selection of pesticide resistance. The impact of such resistance is well documented, for all classes of pests targeted and for almost all types of insecticides, herbicides and fungicides (REX Consortium 2013). More than 10,000 cases of resistance to 300 insecticide compounds have been reported in about 600 species of arthropods (Arthropod Pesticide Resistance Database; www.pesticideresistance.com). Similarly, 300 cases of field resistance to 30 fungicides have been reported in 250 species of phytopathogenic fungi (Fungicide Resistance Action Committee database; http://www.frac.info). The International Survey of Herbicide-Resistant Weeds (http://www.weedscience.com) has suggested that there are currently about 429 biotypes resistant to 153 herbicides in 234 weed species.

2.6.2 Economic Consequences Considered to Date

The environmental impacts described above are obviously costly, in many ways. The various economic consequences considered in the 15 sets of studies are shown in Table 2.9.

Pimentel et al. (1980a, b, 1991a, b, 1992, 1993a, b), Pimentel and Greiner (1997), Pimentel and Hart (2001), Pimentel (2005), Pimentel and Burgess (2014), followed by Steiner et al. (1995), Khan et al. (2002) and Tegtmeier and Duffy (2004), tried to carry out a complete evaluation of the economic consequences of pesticide exposure in bees (Table 2.9). They evaluated colony losses, but also considered (i) losses of honey and wax due to bee colonies being either seriously weakened by pesticides or suffering losses when moved by beekeepers to minimize the risk of pesticide damage, (ii) losses of potential honey production because heavy pesticide applications on some crops may result in beekeepers being excluded from sites otherwise suitable for beekeeping, (iii) the lack of pollination due to losses of bee colonies and (iv) bee rental to compensate for this lack of pollination. Pollination losses were the greatest loss by far, accounting for more than 60 % of the total economic impact of pesticide exposure in bees.

A thorough analysis, such as that performed for bees, has never been undertaken for plants, microorganisms or animals other than bees. Considerations of the economic consequence of arthropod and microorganism depletion have focused on the loss of natural enemies of agricultural pests (Table 2.9). This loss of beneficial arthropods, fungi, bacteria and viruses increases pest pressure on crops. First, such losses allow the primary pests themselves to occur at higher densities. Several outbreaks of primary pests have been accounted for by the depletion of their natural enemies by pesticides (Bommarco et al. 2011; Hardin et al. 1995; Wilson et al. 1998). Second, many secondary pests, i.e. species that were once minor or unimportant crop pests, may become major pests if no longer controlled by their natural enemies (Hardin et al. 1995; Eveleens et al. 1973). Primary and secondary pest outbreaks due to the depletion of natural enemies have two main economic consequences: they increase pesticide use and decrease yields.

Pesticide resistance increases the amount of pesticide used, because higher doses are required to kill resistant pests. The use of alternative pesticides to which the resistant pests are still susceptible, or of a mixture of pesticides, which may be more expensive, may prove necessary. Resistance also decreases yields, because some pests become so resistant that they can no longer be fully controlled by pesticides or because the larger amounts of pesticides required to control resistant pests damage the crops treated.

The annual cost of mortality in birds and fish has been evaluated by multiplying the number of individuals actually killed due to direct or indirect exposure to pesticides by the estimated mean price of the individuals concerned. For birds, two additional types of environmental costs have been considered: the monitoring of species threatened by pesticide exposure and the re-establishment of endangered species, e.g. the bald eagle, *Haliaeetus leucocephalus*, affected by pesticides (Table 2.9).

Three economic consequences have been associated with damage to domesticated animals: the cost of illness, e.g. veterinary fees, the cost of dead livestock and the loss of productivity of animals weakened by poisoning, with affected individuals producing less milk, meat or eggs, for example (Table 2.9).

Yield loss is the principal economic consequence of accidental injury to crops from pesticide use. Contractors applying pesticides can be sued for damage to the crop during or after treatment. In many states of the United States, contractors applying pesticides must provide evidence of financial responsibility before spraying. Most are insured, to protect themselves against expensive lawsuits, and this increases the environmental cost of pesticide use.

2.6.3 Counting Environmental Costs: From Specific to Overall Costs

Some studies have focused on a particular impact. For instance, James (1995) specifically estimated the cost of bird losses in Canada. Some studies have been devoted to a specific crop in a specific area, such as the Punjabi cotton zones in Pakistan (Khan et al. 2002). Others have focused on externalities *sensu stricto*: Steiner et al. (1995) and Tegtmeier and Duffy (2004) in the United States and Pretty et al. (2000, 2001) for the United Kingdom. Steiner et al. (1995) therefore chose to ignore the costs associated with pesticide resistances and the loss of natural enemies, because these costs are mostly met by users (see also Pearce and Tinch 1998). Finally, Pimentel et al. (1980a, b, 1991a, b, 1992, 1993a, b), Pimentel and Greiner (1997), Pimentel and Hart (2001), Pimentel (2005) and Pimentel and Burgess (2014) in the United States, Jungbluth (1996) in Thailand and Ajayi et al. (2002) in Mali assessed the total environmental costs associated with pesticide use on all crops at the national level.

Estimates of economic costs due to environmental damages are therefore highly variable, from US$270,000 (2013) for the birds killed in Canada (James 1995) to about US$8 billion (2013) for total environmental impact in the United Sates (Pimentel et al. 1992, 1993a, b) (Table 2.8).

The two main environmental costs considered stemmed from the increase of pesticide use due to pest resistance and the number of birds killed by pesticide exposure. In the study by Pimentel et al. (1992), these two categories accounted for 35 % and 40 % of the total environmental costs. However, it is particularly difficult to assess the costs associated with bird losses. Pimentel et al. (1992), Pimentel and Greiner (1997), Pimentel (2005) and Pimentel and Burgess (2014) reported that the cost of a bird's life in the United States could be estimated at $0.40, $216 or $800. As pointed out by Bowles and Webster (1995), the techniques used to evaluate this cost were not described. In fact, these values correspond, to the cost per bird for bird watching, bird hunting and for rearing and releasing a bird of an affected species in the wild, respectively. In 1992, Pimentel et al. decided to take an average cost of $30 per bird (Table 2.10). Surprisingly, this cost of $30 was never updated and has remained constant in all the papers since published by Pimentel and coworkers. This resulted in a decrease in the estimated annual cost of bird losses from US$3.37 billion (2013) in Pimentel et al. (1992) to US$2.55 billion (2013) in Pimentel and Burgess (2014) (Table 2.10). Based on the estimate of Pimentel et al. (1992), Tegtmeier and Duffy (2004) decided to take the lowest monetary value assigned per bird (US$0.40 in 1992 – re-evaluated to US$0.51 in 2004). This resulted in estimated costs of US$45 million (2013) per year, almost two orders of magnitude lower than the estimates provided by Pimentel et al. in 1992 (Table 2.10). Finally, Steiner et al. (1995) indicated that the cost of a bird may vary between the lower limit of US$0.40 to the mean value of US$30 chosen by Pimentel and coworkers, resulting in annual costs of about US$47 million to US$3.5 billion (2013) (Table 2.10).

Table 2.10 Annual costs for fish and birds killed by pesticides in the United States

Wildlife	Reference	Year of evaluation	Cost per individual (US$)	Killed individuals per year (in million)	Total cost per year (million US$)	Updated total cost per year (million US$ 2013)
Fish	Pimentel et al. (1980a, b)	1980	0.40	2.00	0.80	2.53
	Pimentel et al. (1991a, b)	ND[a]	–	–	–	
	Pimentel et al. (1992, 1993a, b)	1992	1.70	6–14	10.00–24.00	16.86–40.47
	Pimentel and Greiner (1997); Pimentel and Hart (2001)	1997	4.00	6–14	24.00–56.00	35.15–82.02
	Pimentel (2005)	2005	10.00	6–14	60.00–140.00	72.92–170.14
	Tegtmeier and Duffy (2004)	2002	3.65	6–14	21.90–51.10	28.41–66.29
	Steiner et al. (1995)	1991	1.70–10	1.74–2.10	2.89–21.00	5.08–36.90
Birds	Pimentel et al. (1980a, b)	NQ[b]	–	–	–	–
	Pimentel et al. (1991a, b)	NQ[b]	–	–	–	–
	Pimentel et al. (1992, 1993a, b)	1992	30.00	67.00	2000.00	3372.92
	Pimentel and Greiner (1997); Pimentel and Hart (2001)	1997	30.00	67.00	2000.00	2929.38
	Pimentel (2005, 2009); Pimentel and Burgess (2014)	2005	30.00	72.00	2100.00	2552.06
	Tegtmeier and Duffy (2004)	2002	0.51	67.00	34.50	44.75
	Steiner et al. (1995)	1991	0.40–30	67.00	27.00–2000.00	47.45–3514.58

[a]*ND* no details (they provide an overall cost for fishery losses using a re-evaluation of the 1980 figures without providing any details)
[b]*NQ* not quantified

Similar variations were observed in estimates of fish losses in the United States. The cost of a fish varied from US$0.40 to US$10 between papers, resulting in estimates of the annual cost of fishery losses of between US$2.53 million (2013) in 1980 and US$170 million (2013) in 2005, 2009 and 2014, in studies by the same authors (Table 2.10).

2.6.4 Underestimated and Uncounted Costs

The costs provided by these studies are probably far from the actual costs. There are, indeed, several reasons for thinking that the counted costs were underestimated. In addition, several types of environmental damage have yet to be assessed.

2.6.4.1 Most Costs Were Probably Underestimated

Pimentel et al. (1980a) considered their estimate of the cost of domesticated animal poisoning to be low because it was based only on poisoning cases reported to veterinary surgeons. They indicated that in cases of poisoning in which little can be done for the animal, veterinary surgeons are rarely called.

They also considered their estimates of fish deaths to be low, for many reasons. They indicated that 20 % of the reported fish kills gave no estimate of the number of fish killed and that fish kills often cannot be investigated quickly enough to determine whether they result from pesticide exposure. Furthermore, the fast-moving water in rivers dilutes pollutants, making it difficult to identify the chemical involved, and washes away the poisoned fish. Finally, many dead fish sink to the bottom or are eaten by other fish and therefore cannot be counted. Perhaps most importantly, unlike direct kills, few, if any, of the widespread, low-level pesticide poisoning events result in dramatic manifestations and these events are, therefore, not recognized or reported.

The total numbers of birds killed by pesticides is difficult to determine because, like most vertebrate species, they are often secretive, camouflaged, highly mobile and, as pointed by Pimentel et al. (1980a), they do not conspicuously '*float to the surface*' as fish do. They often live in dense grass, shrubs, and trees. Dead birds disappear quickly, well before they can be found and counted (Mineau and Collins 1988). Scavengers have been shown to remove >90 % of bird carcasses placed in farmland overnight (Prosser et al. 2008). Furthermore, field studies seldom account for birds dying outside the treated areas, but birds often hide and die in inconspicuous locations. Estimates of bird mortality do not include birds that die due to the death of one of their parents or the deaths of the nestlings. They do not include nestlings killed because they were fed contaminated arthropods and other foods either. Mineau (2005) considered the estimate of Pimentel et al. (1992) – a mortality of 67 million birds per year in the United States – to be too conservative. Indeed, he estimated that, at the start of the 1980s, 17–91 million songbirds were dying annually in the United States Corn Belt, purely due to the use of a granular formulation of carbofuran in corn. However, Mineau (2005) felt that this figure was still too conservative, because it did not include birds dying in other crops treated with granular carbofuran, such as soybean, sorghum, groundnut, tobacco, cotton or sunflower, or the lethal impact of all the other pesticides, including rodenticides, on birds. Based on the analysis of Mineau (2005), there were probably more than 100 million birds lost annually in the United States between 1978 and 1985.

Crop losses due to pesticides are also probably underestimated because, for many losses, the parties involved come to an out-of-court settlement, and the losses are therefore never reported to the state and federal agencies (Pimentel et al. 1993a). In addition, pesticide damage to target crops due to the application of larger doses to kill pesticide-resistant pests, has probably been underestimated.

2.6.4.2 Several Costs Have Never Been Evaluated

Production and storage sites may be particularly polluted (Elfvendahl et al. 2004; Jit et al. 2010), but this pollution has never been taken into account. Half a million tons of obsolete pesticides are stored throughout the developing world (Food and Agriculture Organization 2011a), often outdoors, in leaky containers, resulting in particularly high levels of pollution of the surrounding soil and water (Ahad et al. 2010; Dvorská et al. 2012). Similarly, the sites at which pesticides are prepared and loaded into sprayers and at which tractors and sprayers are washed may be highly polluted (Helweg et al. 2002). Some costs are covered by the chemical companies themselves. However, this pollution generates externalities *sensu stricto*, through decreases in the price of land, houses and recreational activities close to the sites concerned (Epp et al. 1977).

The cost of damage to wildlife has been counted only for birds and fishes. However, as indicated in Sect. 2.6.1.1, many other non-human vertebrates are also damaged by pesticide use. Similarly, the monetary cost of pesticide impact on aquatic invertebrates, plants, algae and the soil community has never been estimated.

The direct costs of bird and fish losses have been estimated, but several indirect costs associated with these losses have yet to be analyzed. Indeed, birds and fish provide several ecosystem services. Birds make a significant contribution to the four principal types of ecosystem services defined by the United Nations Millennium Ecosystem Assessment: provisioning, regulating, cultural and supporting services. In agricultural ecosystems, they control pests, by eating arthropods, rodents and weeds (Whelan et al. 2008). Interestingly, James (1995) estimated the cost of bird losses in Canada, by setting the cost of an individual bird at the cost of achieving the same level of insect control with insecticides, if the birds were absent. This clearly corresponds to only part of the economic advantage birds provide to humans. Indeed, in addition to their contribution to pest control, birds also play significant roles in pollination, seed dispersal, and scavenging (Whelan et al. 2008).

Arthropods also provide substantial ecosystem services. However, the studies performed to date have considered only the lack of pest control provided by natural enemies killed by pesticides. However, like bees, '*wild*' insects provide other services in addition to pest control, including pollination, dung burial and food for wildlife (Losey and Vaughan 2006).

2.6.5 Conclusions

The cost of the environmental impact of pesticides has been poorly investigated to date. Only 15 sets of studies have evaluated these costs, and these studies were actually based on only 11 independent datasets. Only six studies provided an overall cost assessment at national level. The pioneering work of David Pimentel in the United States remains the key reference, but this work dates from the 1980s and 1990s, with a partial update published in 2005, 2009 and 2014. Although Pimentel and coworkers provided the most complete evaluation of environmental impairment available, we have shown that this assessment was probably highly incomplete, with a strong underestimation of costs.

It should be borne in mind that the current environmental impact of pesticide use is probably very different from that during the 1980s and 1990s (see Sect. 2.8.4). In North American and European countries, the most dangerous and persistent pesticides (e.g. DDT, carbofuran) have been banned and partly replaced by less toxic and less persistent compounds, strongly decreasing the impact on birds and fish. However, other countries, such as India and China, are still producing, exporting and using DDT (van den Berg et al. 2012). Moreover, pesticide resistance has steadily increased over the last 30 years (Rex Consortium 2013). The doses of pesticides applied to many crops are, therefore, almost certainly higher than in the past, resulting in a greater impact on the environment.

To conclude on environmental costs of pesticide use, we show that they suffered large underestimation and most of them were never considered in the literature. They were nevertheless estimated to up to US$8 billion (2013) in the United States in 1992.

2.7 Defensive Expenditures

The aversive behavior approach estimates the amount that someone is willing to pay to reduce their environmental exposure to hazardous chemicals, such as pesticides (Dickie 2003). This expenditure can be seen as an investment, to protect against both short- and long-term illnesses. As for the cost-of-illness approach, different names have been given to the costs due to aversive behavior: averting costs, precautionary costs, mitigating costs, revealed willingness to pay for safety and defensive expenditures (Wilson 1999a). In this review, we will use the term "defensive expenditures". Defensive expenditures can be either private if incurred by the farmers themselves or external if incurred by consumers (Pearce and Tinch 1998) (Table 2.1). Defensive expenditures may be incurred due to several types of aversive behavior, such as wearing protective clothes when applying pesticides for farmers, monitoring and removing pesticides from drinking water for consumers, and eating organic food to avoid, or at least reduce the levels of pesticide residues on food for consumers.

2.7.1 Defensive Expenditures for Pesticide Handling and Spraying

Farmers take safety measures when handling and applying pesticides to their crops, to decrease or prevent direct exposure to these chemicals. The defensive expenditures taken into account include costs associated with precautions taken to reduce direct exposure to pesticides, such as masks, caps, shoes/boots, handkerchiefs, long-sleeved shirts/pants (Table 2.11). These products may have multiple uses, but only products purchased specifically for the use and handling of pesticides are considered and their costs are generally annualized according to the expected lifespan of the product (e.g. Atreya 2008). Wilson (1999a, 2000b, 2003, 2005) considered the hiring of personnel to spray pesticides as a defensive activity, and therefore included this expense as defensive expenditures.

Only 13 articles have estimated the cost of defensive expenditures, and these estimates were based on only seven independent datasets (Table 2.1). This small number of studies considering defensive expenditures may be accounted for by defensive expenditures not being an externality *sensu stricto*. These costs are paid by farmers, which accounts for their lack of inclusion in studies focusing on the external costs of pesticide use such as those performed by Pimentel and coworkers.

Two groups of authors, in particular, have explored the defensive expenditures of farmers: Clevo Wilson (Wilson 1999a, b, 2000b, 2002b, 2003, 2005) and Athukorala et al. (2012) in Sri Lanka and Kishor Atreya (Atreya 2005, 2007, 2008 and Atreya et al. 2012, 2013) in Nepal. We were able to identify only one other studying exploring defensive expenditures, by Ajayi et al. (2002), in Mali.

In Nepal and Sri Lanka, farmers were found to spend a mean of between US$6 and US$32 (2013) per year on defensive expenditures (Table 2.11). Ajayi et al. (2002) estimated that farmers in Mali would need to spend US$30 to US$60 (2013) per year on equipment to ensure that they were protected against pesticide exposure. Wilson (Wilson 1999a, 2000b, 2003, 2005) and Athukorala et al. (2012) used data obtained directly from farmers to estimate the annual cost for the whole of Sri Lanka. They estimated these costs at between US$1 million (2013) if only 5 % of the farmers used pesticides and US$10 million (2013) if 20 % of the farmers used pesticides (Table 2.11).

Table 2.11 Costs of aversive protection (defensive expenditures) against pesticide use

	Atreya et al. (2012)	Atreya et al. (2012, 2013)	Atreya (2005)	Atreya (2007, 2008)	Athukorala et al. (2012)	Wilson (1999a, b, 2000b, 2002b, 2003, 2005)	Ajayi et al. (2002)
Country	Nepal	Nepal	Nepal	Nepal	Sri Lanka	Sri Lanka	Mali
Year	2008	2008	2004	2005	2007	1995	1999
Items							
Boots/Closed shoes	X	X	X	X	X	X	X
Shawls/Face shield	X	X	X	X			X
Gloves	X	X	X	X	X	X	X
Mask	X	X	X	X	X	X	X
Cap/Hat	X	X	X	X			X
Handkerchief	X	X	X	X			
Long-sleeved shirts	X	X	X	X	X	X	X
Full-length pants	X	X	X	X	X	X	X
Spectacles	X	X	X	X			
Building of special storage units						X	
Hired labor						X	
Cost per year (US$ 2013)							
Average cost per farmer	7.09	11.67	6.75	4.88	16.52	34.1	39.26–78.85
Overall cost at national level	–	–	–	–	971,116–3,937,820	1,728,000–10,379,000	–

In Nepal, defensive expenditures accounted for about 15 % of the total cost of pesticide use and 27 % of pesticide expenditure, i.e. the amount spent on purchasing pesticides in a year. Defensive expenditures were slightly higher (Atreya 2008) or slightly lower (Atreya et al. 2012, 2013) than the cost-of-illness, but essentially of a similar magnitude. In Sri Lanka, Athukorala et al. (2012) found these costs to be one quarter those for medical expenditure and one seventh the loss of earnings; cost-of-illness was thus 11 times higher than defensive expenditures (Wilson 1999a). Nevertheless, in this country, annual defensive expenditures corresponded to 12 % of the monthly income of a farmer (Athukorala et al. 2012; Wilson 1999a, 2000b, 2003, 2005). These costs, although low, could be a significant burden to farmers, whose incomes fluctuate greatly, due to adverse biotic, e.g. pest and disease damage, and abiotic, e.g. weather conditions, crop price fluctuations, conditions.

Several types of defensive expenditures have not been considered, probably due to data, time and financial constraints. The elements not analyzed include the purchase of more expensive sprayers less likely to malfunction and place the user at risk of exposure. They also include the time spent purchasing, cleaning and fixing defensive/protective equipment, and reading '*warnings and instructions*'. Precautionary drug treatment to protect against pesticide exposure and leisure time given up in favor of aversive behavior should also be taken into account. The estimates to date therefore almost certainly constitute the lower limit of the range of actual defensive expenditures paid by farmers to reduce their exposure to pesticides.

Moreover, in developing countries, these costs could probably be increased to levels much higher than those currently observed, as pesticide users often adopt few protective measures (Food and Agriculture Organization 2011b). Spraying is sometimes carried out without protection and even those farmers who do try to protect themselves generally limit this protection to the wearing of long-sleeved shirts and long pants. Low levels of income, awareness and education, the hot and humid climate, cultural taboos, fashion and discomfort are significant factors accounting for the lack of personal protection (Atreya et al. 2013) (Fig. 2.1).

Sivayoganathan et al. (1995) reported that some Sri Lankan farmers were keen to use protective measures but did not do so due to cultural taboos, such as wearing shoes in the field. The field is seen as a sort of "temple" because the land within it produces food. Another cultural taboo mentioned concerned the wearing of long pants during pesticide applications, which many farmers, especially the elderly, were reluctant to do, due to their low socioeconomic status.

Finally, not only might farmers be unable to afford adequate precautionary/defensive measures, but the protective gear required may be unavailable as it may not be sold by any shop to which the farmer has access. Hence, defensive expenditures have never been correctly counted, both because the actual expenses were not fully estimated and because they could potentially be much higher than they currently are, particularly in developing countries.

2.7.2 Defensive Expenditures for Safe Drinking Water

The presence of pesticides in tap water may be one of the key reasons for consumers buying bottled water or drinking purified or filtered water. These sources of water are much more expensive for the consumer than tap water. The excess costs of purified or bottled water over tap water could be considered as both a private cost borne by farmers if they drink such water and as an external costs to non-farming consumers buying such water. The production and transportation of bottled water also require the consumption of massive amounts of fossil fuels (Gleick and Cooley 2009). Finally, the bottles degrade slowly, and their incineration can produce toxic byproducts. Bottled water thus has an environmental impact between 90 and 1000 times greater than that of tap water (Jungbluth 2005). The resulting pollution can be considered as a negative externality for society as a whole. However, if the production, transportation and purchase of bottled water and all devices for water purification or filtration are to be considered as defensive expenditures, and hence as external costs, these expenditures should be made specifically to protect against pesticide residues. This relationship is anything but simple.

Consumers choose to drink bottled, purified or filtered water for two main reasons: because they think this water tastes better and/or is safer than tap water (Doria 2006; Doria et al. 2009; Dupont et al. 2010). Several factors are known to influence the public perception of drinking water quality: organoleptic properties, risk perception, attitude towards water chemicals, past problems attributed to water quality, trust in water companies, information from the mass media and family members (Doria 2010). Hence, the presence of pesticides, whether real or imagined, in tap water may be only one of a number of factors pushing people to buy bottled water and/or to drink purified or filtered water. Unfortunately, we were able to identify no study specifically exploring this question. Studies on factors influencing drinking behavior have considered chemical pollutants either as a general entity, i.e. with no specification of the type of chemical substance (e.g. Auslander and Langlois 1993), or have concentrated on lead, chlorine and/or water hardness, e.g. the survey of Statistics Canada (2009), which specifically mentioned chlorine. Pesticides, like other chemical substances including fluoride, nitrates, heavy metals and industrial chemicals, are sometimes specified, but, according to Doria (2010), their relevance to the perception of drinking water safety appears to be very limited or restricted to specific locations.

No specific data are available for pesticides, but several studies have explored the influence of chemicals on the water-drinking behavior of consumers, notably in Canada. In Toronto, 73 % of those questioned felt that tap water contained "some" or "a lot" of chemical pollutants, but half the households overall rated this source of water as "good" or "very good" (Auslander and Langlois 1993). In a more recent national survey of a representative sample of 1633 Canadians, 62 % felt that tap water posed no problem for health (Dupont et al. 2010). Only 12 % and 3 % believed that this source of water posed moderate or serious problems for health, respectively. In their study focused in one Canadian province, McLeod et al. (2014) also found that no more than 12 % of the 2000 respondents believed tap water to be unsafe to drink. Noteworthy, those respondents who believed tap water to be unsafe appeared more likely to choose bottled water (McLeod et al. 2014). In other countries with reliable supplies, surveys generally indicate that most people perceive the risk associated with drinking tap water to be small (Doria 2006). In low- and medium-income countries, in which tap water quality is often poorer, surveys of the motives for choosing bottled water over tap water have not been performed. However, in such countries, the average per capita consumption of bottled water is low.

In conclusion, the extra cost of drinking bottled, purified and filtered waters, rather than tap water, cannot be firmly attributed to the presence of pesticides. Of course, consumers indirectly pay for the monitoring and elimination of pesticides from the tap water they use, as these costs are passed on by water companies, through the billing process. We decided to count these costs as regulatory rather than as defensive expenditures because, as indicated in Sect. 2.4, monitoring and decontamination processes are mandatory in most countries: see the United States Safe Drinking Water Act (http://water.epa.gov/lawsregs/rulesregs/sdwa/index.cfm), for example.

Pesticides trigger defensive expenditures when they are detected in tap water at levels beyond the threshold considered acceptable, thus causing a decrease in quality. The monitoring of private wells, which are generally not regulated by public authorities, and the use of filtering/purifying devices for detecting and eliminating pesticides from these wells can also be considered as defensive expenditures.

Water quality violations may trigger aversive behavior, such as the purchase of bottled water. When such violations are due to pesticide contamination (e.g. Zaki et al. 1982), the increased in the purchase of bottled water in the area concerned may be considered defensive expenditures. Zivin et al. (2011) estimated that, in 2005, United States citizens spent US$47.15 million (2005) in response to element/chemical violations of water quality. They indicated that this estimate probably constituted the lower limit of the cost of defensive expenditures, because they only considered bottled water consumption and did not include other responses to violations, such as purchasing alternative beverages, e.g. juice, other actions people may have taken, e.g. boiling water, and more permanent responses, e.g. installing water filters. Zivin et al. (2011) did not provide details of the elements/chemicals responsible for the quality violations. We know only that they did not include nitrate, which was counted separately. It is therefore difficult to determine what proportion of the costs corresponded to pesticide contamination. Similarly, Dupont and Jahan (2012) estimated that Canadian households spent almost US$600 (2010) per year on tap water substitutes (purchase of bottled water and devices for filtering/purifying tap water), to decrease the perceived health risks associated with tap water consumption. Unfortunately, the influence of pesticides on this perception was not investigated.

The second type of defensive expenditures concerns the monitoring and decontamination of private wells and small-scale public systems. As indicated above, in the United States, state and federal authorities do not generally regulate these sources of drinking water. The householders concerned therefore pay for the detection of pesticides in these wells and their elimination. In the United States, 15 million households regularly obtain drinking water from their own private wells (United States Environmental Protection Agency 2002) and the groundwater in those wells may be contaminated with pesticides, particularly in rural areas (Toccalino et al. 2014). Pesticides, such as atrazine, deethylatrazine, simazine, metolachlor, and prometon are, indeed, regularly detected in groundwater and wells (Goss et al. 1998; Hallberg 1989; Ritter 1990, 2001; Toccalino et al. 2014). However, pesticide concentrations in North American domestic wells were found to be generally low. In Ontario, for instance, only six of the 1292 water-wells surveyed contained pesticide residues at concentrations above the maximum acceptable value (Goss et al. 1998). Similar findings were reported for the United States: for the 1993–2011 period, pesticide concentrations exceeded human-health benchmarks in only 1.8 % of the 2541 samples collected from 1271 wells in well networks distributed nationwide (Toccalino et al. 2014). However, pesticide contamination rates and concentration may reach higher values in some countries. In the Netherlands, several pesticides were detected in 27 % of groundwater samples taken from 771 monitoring wells. In 11 % of these samples, the concentration exceeded the upper regulatory limit (Schipper et al. 2008).

Worldwide, the most important contaminant of groundwater and private wells, in terms of health concerns, is arsenic (Nordstrom 2002). Arsenic contamination may have diverse sources, some of which are entirely natural, as in Bangladesh (Nickson et al. 1998). However, arsenic contamination may also result from local anthropogenic activities, such as mining (Mukherjee et al. 2006). In Canada and the United States, significant amounts of arsenic contamination result from the use of arsenic-based pesticides (Smedley and Kinniburgh 2002; Wang and Mulligan 2006).

According to the Massachusetts Department of Environmental Protection, testing a well for arsenic costs US$15 to US$30. Treatment systems for removing arsenic (reverse osmosis, activated alumina) cost at least US$400 per year (Sargent-Michaud et al. 2006). In addition to the costs of monitoring and testing, the presence of arsenic may also increase the consumption of bottled water (Jakus et al. 2009). As arsenic comes from diverse sources, which may vary over space and time, it is not easy to evaluate defensive expenditures due to arsenic-based pesticides. However, in the United States, where 15 million households regularly obtain drinking water from their own private wells, this cost might reach several hundred million US$ per year.

2.7.3 Defensive Expenditures to Avoid Pesticide Residues in Food: The Purchase of Organic Food

Consumers choose to purchase organic food for several reasons, some of which are linked to the externalities of pesticides and to a demand for pesticide-free food (Fotopoulos and Krystallis 2002; Misra et al. 1991; Squires et al. 2001; Tsakiridou et al. 2008; Williams and Hammitt 2001) (Fig. 2.6). Most consumers of organic

Fig. 2.6 Consumers choose to purchase organic food for several reasons, but partly as a consequence of the perceived negative risk of pesticides to the environment and to the consumer. The world market for organic food has grown considerably over the last 15 years: it almost tripled between 2000 and 2008 and continued to grow thereafter, from US$50 billion in 2008 to US$64 billion in 2012 (Sahota 2014). Assuming that prices in this market are 20 % higher than those of conventional food and that about 50 % of the reasons for consumers choosing organic food are directly linked to the avoidance of pesticide risk (e.g. Schifferstein and Oude Ophuis 1998), then the added cost of pesticide use may be about US$6.4 billion (2013) worldwide (Unmodified USDA photography courtesy of Sam Jones-Ellard, under creative common license CC BY (https://creativecommons.org/licenses/by/2.0/))

food declare that the main reasons for this choice are connected to personal health and the avoidance of environmental damage (e.g. Huang 1996; Hughner et al. 2007; Magnusson et al. 2003; Saba and Messina 2003; Schifferstein and Oude Ophuis 1998; Schlegelmilch et al. 1996; Squires et al. 2001; Tregear et al. 1994; Wier et al. 2008). In Greece, about 90 % of general consumers consider organic food to be healthier than conventionally farmed food, and 75 % think that it is better for the environment; even higher percentages were recorded among the consumers of organic food (Tsakiridou et al. 2008). Animal well-being, taste or simply fashion are other factors less frequently proposed by consumers to explain their choices (Pearson et al. 2011). Parents of young children and babies are among those most likely to consume organic food, as a proactive measure, to prevent health problems (Pearson et al. 2011). Another reason cited for buying organic food is also linked to health, with some ill individuals choosing to buy organic food because they hope that it will help them to recover more rapidly (Pearson et al. 2011). Health is thus a key motive behind organic food consumption. Another reason often given for purchasing organic food is that it decreases damage to the environment, and this idea is generally supported by scientific evidence (e.g. Mäder et al. 2002; Gomiero et al. 2011). Buying organic food is thus partly a consequence of the perceived negative risk of pesticides to the environment and to the consumer.

Organic food consumption can thus be considered, at least in part, as an externality of pesticide use if organic food is more expensive than non-organic, conventional foods. Comparisons of the organic and conventional food markets show that organic food is generally more expensive than conventionally produced food (e.g. Bonti-Ankomah and Yiridoe 2006). The excess cost of organic food varies considerably between countries and products (Bonti-Ankomah and Yiridoe 2006) and is dependent on several factors. However, according to several studies, the lower limit for this price premium would lie somewhere between 10 % and 20 % (e.g. Bonti-Ankomah and Yiridoe 2006; Rodríguez et al. 2008), although price premiums of between 50 % and more than 100 % were reported in the United States in 2013 for fruits and vegetables, respectively (see the web page on Organic prices of the United States Department of Agriculture Economic Research Service: http://www.ers.usda.gov/data-products/organic-prices.aspx#.VAmF0mTV_sk). This price premium, paid by the consumers of organic food thus corresponds, at least in part, to the consumers' willingness to pay for avoiding pesticide risks (Onozaka et al. 2006) and, more precisely, to the hedonic estimation of willingness to pay for a reduction of the presence of pesticides in food. The range of values for the mean price premium of organic food has been confirmed by studies of the willingness to pay for organic food carried out with the contingent valuation technique. Consumers were asked to set a value on the premium they would be prepared to pay for organic food rather than conventionally produced food. These studies also highlighted considerably variability in the responses obtained (e.g. Zehnder et al. 2003; reviewed by Bonti-Ankomah and Yiridoe 2006), but they frequently suggested that the minimum value was about 10–20 % (e.g. Bonti-Ankomah and Yiridoe 2006; Gil et al. 2000; Onozaka et al. 2006; Rodríguez et al. 2008).

The worldwide organic food market was of the order of US$64 billion in 2012 (Sahota 2014), equally split between Europe (US$29 billion) (Schaack et al. 2014) and the United States (US$29 billion) (Fitch Haumann 2014). In Europe, the organic food market in 2012 represented about US$9 billion in Germany, US$5 billion in France and US$2.5 billion in the United Kingdom (Schaack et al. 2014). The world market for organic food has grown considerably over the last 15 years: it almost tripled between 2000 and 2008 and continued to grow thereafter, from US$50 billion in 2008 to US$64 billion in 2012 (Sahota 2014).

If we assume that prices in this market are 20 % higher than those of conventional food and that about 50 % of the reasons for consumers choosing organic food are directly linked to the avoidance of pesticide risk (e.g. Schifferstein and Oude Ophuis 1998), then the added cost of pesticide use is about 10 % of the total market value of organic food. This amounts to US$2.9 billion for the United States and

Europe, and about US$0.9 billion for Germany, US$0.5 billion for France, and US$0.25 billion for the United Kingdom. Griffith and Nesheim (2008) used hedonic prices and purchase quantities for 2003 and 2004 in the United Kingdom to estimate the aggregate lower limit of willingness to pay for organic products. They obtained a value of about 22 % of the annual expenditure on organic products, corresponding to about US$0.55 billion, based on the figures obtained for the organic market in the United Kingdom in 2012. Griffith and Nesheim (2008) estimated that about 20 % of the lower limit of the willingness to pay was directly linked to health and environmental concerns – about US$110 million, corresponding to 44 % of our estimate of US$0.25 billion.

2.7.4 Conclusion

Defensive expenditures have rarely been considered among the external and "hidden" costs of pesticide use. For instance, we found no study considering the defensive expenditures of both farmers and consumers. In particular, the consumption of organic food as a defensive action against pesticide residues has never been fully considered as a negative externality of pesticide use. Indeed, all studies to date on the economics and rationale of organic food consumption have been completely disconnected from studies analyzing the benefit-cost ratio of pesticide use.

In general, aversive actions have been little studied and, when considered, they have generally been restricted to the protection of the body and respiratory system by farmers handling or applying pesticides. However, these costs are only part of the costs directly borne by farmers.

Furthermore, aversive actions could be carried out on a much wider scale than is currently the case. This is certainly true for protective clothing, which is rarely worn by farmers in most developing countries, and for the monitoring and decontamination of drinking water. If all owners of private wells carried out monitoring and were equipped with a filter/purifier, or if the consumption of bottled water continues to grow, then defensive expenditures to avoid residues in drinking water could rise exponentially. However, it should be borne in mind that these costs are somewhat linked to cost-of-illness. If tap water contains pesticide residues at levels that may injure human health, then an increase in defensive expenditures should lead to a decrease in cost-of-illness. Put another way, some of the current cost-of-illness could be due to a lack of aversive action. Alternatively, an increase in defensive expenditures might decrease the overall cost of pesticide use if these additional defensive expenditures are overcompensated by the decrease in cost-of-illness they trigger. Similarly, an increase in the consumption of organic food might decrease the cost-of-illness by reducing chronic illness although the relationship between exposure to low pesticide doses and chronic illnesses remains very difficult to quantify.

Here, we show that defensive expenditures have rarely been considered in the literature of pesticide use cost. These costs include at least the extra cost of organic food consumption due to aversive behavior linked to pesticide use. This cost reached more than US$6.4 billion worldwide in 2012.

2.8 Overall Hidden and External Costs

Pesticide use has a marked positive impact on agriculture (Cooper and Dobson 2007; Gianessi 2009; Gianessi and Reigner 2005, 2007) and human health (Cooper and Dobson 2007). However, as highlighted above, it also has a significant negative impact on the environment and on human health, and entails economic costs linked to regulations and defensive actions. It is therefore worthwhile estimating the global cost of pesticide use, for comparison with the economic benefits, with a view to re-evaluating the overall economic balance of pesticide use (see Sects. 2.9.1 and 2.9.2). This is a prerequisite for the evaluation of public policies concerning pesticide use, including the reduction of pesticide use (e.g. Barzman and Dachbrodt-Saaydeh 2011; Löfstedt 2003). Unfortunately, several current policies relating to the reduction of pesticide use are based on estimates that do not consider the global cost of pesticide use, including external costs, but only the benefits in terms of agricultural production, e.g. the Ecophyto 2018 plan of the French government, which aims to halve pesticide use over a 10-year period (Jacquet et al. 2011). In evaluations of the consequences of regulations aiming to decrease pesticide use, very different conclusions may be reached depending on whether the global costs of pesticide use are (Pimentel et al. 1993b; Pimentel 2005; Pimentel and Burgess 2014) or are not (Gianessi 2009; Gianessi and Reigner 2005, 2007; Jacquet et al. 2011) taken into account. This section reviews the few studies that have tried to estimate the overall hidden and external costs at national level. We will see that such costs are underestimated and that the available estimates are out-of-date. By comparing different datasets and estimating the specific costs that were not estimated in previous studies, we tried to perform a more complete evaluation of the hidden and external costs of pesticide use in the United States at the beginning of the 1990s.

2.8.1 A Small Numbers of Estimates

We found only ten independent groups of papers combining estimates of regulatory, environmental and human health costs at the national level. These groups of studies are those of Ajayi et al. (2002) for Mali, Houndekon and De Groot (1998) and Houndekon et al. (2006) for Niger, Jungbluth (1996) and Praneetvatakul et al. (2013) for Thailand, Khan et al. (2002) for Pakistan, Pimentel and coworkers (Pimentel et al. 1980a, b, 1991a, b, 1992, 1993a, b; Pimentel and Greiner 1997; Pimentel and Hart 2001; Pimentel 2005; Pimentel and Burgess 2014), Steiner et al. (1995) and Tegtmeier and Duffy (2004) for the United States, Pretty et al. (2000, 2001) for the United Kingdom, and Fleischer and coworkers (Fleischer 1999; Waibel and Fleischer 1998; Waibel et al. 1999) for Germany.

These articles revealed considerable heterogeneity for overall hidden and external costs, which ranged from US$5.4 million (2013) in Niger in 1996 (Houndekon and De Groote 1998; Houndekon et al. 2006) to US$13.6 billion (2013) in the

United States in 1992 (Pimentel et al. 1992, 1993a, b) (Table 2.12). For the United States, the estimates of Pimentel and coworkers also varied over time. They reported overall hidden and external costs of US$2.7, 3.7, 13.6, 11.8 to 12.1 and 11.7 billion (2013) in 1980, 1991, 1992, 1997 and 2005, respectively (Table 2.12). These differences mostly reflected differences in the types of costs taken into account. Hence, from 1991, Pimentel and coworkers included the cost of monitoring wells and groundwater, accounting for 55 % of the external costs. From 1992, they also estimated the cost of bird losses, accounting for 25 % of the external costs, and re-evaluated the cost of pesticide resistance from about 7–17 % of the external costs.

2.8.2 Overall Costs Are Underestimated

The overall hidden and external costs reported above are underestimated for two reasons. First, none of the available estimates include defensive expenditures (Table 2.13). Second, as shown above, they did not take into account some, or even in some cases most of the specific costs within the other three cost categories, i.e. environmental impact, human health and regulatory actions (Table 2.13). For instance, losses of reptiles, amphibians, soil and aquatic communities and wild vertebrates other than birds and fish have never been evaluated (Table 2.13). Similarly, the costs of the human health impact of pesticide use have not been fully explored. Pimentel et al. estimated the costs of cancer treatment, but they did not calculate the cost of deaths due to these cancers (Table 2.13). Finally, none of the estimates took into account major environmental disasters associated with pesticide production and disposal sites. The dramatic pesticide industry accidents at Bhopal in India (Mishra et al. 2009) (Fig. 2.7) and Seveso in Italy (Consonni et al. 2008), together with less severe incidences, such as the James River kepone disaster in the United States (Huggett and Bender 1980), caused thousands of deaths and long-term disorders in humans, together with damage to the soil, animals and plants that could probably be estimated at several billion of US$.

This bias towards an underestimation of external costs is not related to a lack of rigor on the part of the authors conducting these studies. Instead, it results principally from the difficulties involved in estimation of the economic costs of the unintentional impacts of pesticide use, particularly for goods without market values. Indeed, Pimentel and Greiner (1997) pointed out that the scarcity of data made their assessments of the external costs inaccurate, such that the costs themselves had to be considered incomplete. Hence, as indicated by Waibel et al. (1999), most estimates of external costs performed to date must be considered as minimum costs.

Table 2.12 Overall hidden and external costs of pesticide use. DE corresponds to defensive expenditures

Reference	Country	Year	Costs in million US$ 2013 (per year)				
			Human health impacts	Regulatory actions	Environmental impacts	DE	Overall hidden and external costs
Ajayi et al. (2002)	Mali	1999	3.71	1.58	38.11	NE[a]	43.40
Houndekon and De Groote (1998); Houndekon et al. (2006)	Niger	1996	4.40	0.15	0.89	NE	5.44
Jungbluth (1996)	Thailand	1995	1.26	558.33	5.58	NE	565.17
Khan et al. (2002)	Pakistan	2002	78.06	9.71	815.12	NE	902.89
Pimentel et al. (1980a, b)	United States	1980	581.18	491.96	1621.17	NE	2694.31
Pimentel et al. (1991a, b)	United States	1991	439.32	2372.34	948.94	NE	3760.60
Pimentel et al. (1992, 1993a, b)	United States	1992	1326.45	4319.01	7967.84	NE	13,613.30
Pimentel and Greiner (1997); Pimentel and Hart (2001)	United States	1997	1367.04	3451.19–3751.06	6993.99	NE	11,812.22–12,112.09
Pimentel (2005, 2009); Pimentel and Burgess (2014)	United States	2005	1492.96	4229.13	5973.50	NE	11,695.59
Praneetvatakul et al. (2013)	Thailand	2010	2.99	357.28	16.88	NE	377.15
Pretty et al. (2000, 2001)	United Kingdom	1996	2.30	318.51	62.74	NE	383.55
Steiner et al. (1995)	United States	1986	246.29–486.51	3203.00	203.85–4029.46	NE	3653.14–7718.97
Tegtmeier and Duffy (2004)	United States	2002	1308.89	4988.69	1469.74–1507.62	NE	7767.32–7805.20
Fleischer (1999); Waibel and Fleischer (1998); Waibel et al. (1999)	Germany	1996	17.99	168.26	9.31	NE	195.56

[a]*NE* not estimated

Table 2.13 Cost taken (green) or not (red) into account in the estimates of the overall cost of pesticide use

Category of cost	Ajayi et al. 2002	Houndekon and De Groot 1998; Houndekon et al. 2006 (a)	Jungbluth 1996	Khan et al. 2002	Pimentel et al. 1980a, 1980b	Pimentel et al. 1991a, 1991b	Pimentel et al. 1992, 1993a, 1993b	Pimentel and Greiner 1997; Pimentel and Hart 2001	Pimentel 2005, 2009; Pimentel and Burgess 2014	Praneetvatakul et al. 2013	Pretty et al. 2000, 2001	Steiner et al. 1995	Tegtmeier and Duffy 2004	Fleischer 1999; Waibel and Fleischer 1998; Waibel et al. 1999
Regulatory actions														
Pesticide registration, regulation and market monitoring	G	R	G	R	R	R	R	R	R	R	R	G	R	G
Public awareness campaigns on pesticide impact	R	R	R	G	R	R	R	R	R	R	G	R	R	G
Disposal of obsolete and leftover pesticides	G	G	R	R	R	R	R	R	R	R	R	G	R	R
Farm work, mandatory safety	R	R	R	R	R	R	R	R	R	R	R	G	R	R
Control & monitoring														
Crop and/or food	R	R	R	R	R	R	G	G	G	G	G	R	R	G
Water (surface, underground and/or wells)	R	R	R	R	R	G	G	G	G	G	G	G	G	G
Livestock	R	R	R	R	R	R	R	R	R	R	G	R	R	R
Wildlife	R	R	R	R	G	G	R	R	R	R	R	R	R	R
Undefined	R	R	G	R	R	R	R	R	R	R	R	R	R	R
Water decontamination	R	R	R	R	R	R	G	G	G	G	G	R	R	G
Public research on pesticides	R	R	G	R	R	R	R	R	R	R	R	R	R	R
Extension services	R	R	G	R	R	R	R	R	R	R	R	R	R	G
Economic shortfall														
Crop	R	R	R	R	G	G	G	G	G	G	R	R	R	R
Water	G	R	R	R	R	R	R	R	R	R	R	R	R	G
Livestock	R	R	R	R	G	G	R	R	R	R	R	R	R	R
Milk	R	R	R	R	G	G	R	R	R	R	R	R	R	R
Fishing	R	R	R	R	G	G	R	R	R	R	R	R	R	R
Human health impacts														
Acute poisoning														
Medical care	G	G	G	G	G	G	G	G	G	G	G	G	G	G
Loss of work	G	G	G	G	G	G	G	G	G	G	G	G	G	G
Other indirect costs	G	R	R	R	R	R	R	R	R	R	R	R	R	G
Cost of fatal cases	G	R	R	R	G	G	G	G	G	R	R	R	G	G
Chronic poisoning														
Medical care	R	G	R	R	G	G	G	G	G	R	R	G	G	R
Loss of work	R	G	R	R	R	R	R	R	R	R	R	R	R	R
Other indirect costs	R	R	R	R	R	R	R	R	R	R	R	R	R	R
Cost of fatal cases	R	R	R	R	R	R	R	R	R	R	R	R	R	R
Environmental impact														
Damageto animals, plants, algae and microorganisms														
Crops/cultivated plants/trees	R	R	R	R	G	G	G	G	G	R	R	R	R	R
Wild plants (other than weeds)	R	R	R	R	R	R	R	R	R	R	R	R	R	G
Domestic animals and livestock	R	G	R	G	G	G	G	G	G	R	R	R	R	R
Fish	R	R	R	R	G	G	G	G	G	R	R	G	G	R
Birds	R	R	R	R	G	G	G	G	G	R	R	G	G	R
Wild vertebrates (other than birds and fish)	R	R	R	R	R	R	R	R	R	R	R	R	R	R
Bees	G	R	R	G	G	G	G	G	G	R	G	G	G	G
Natural enemies	G	R	G	G	G	G	G	G	G	G	R	R	G	R
Invertebrates (other than bees and natural enemies)	R	R	R	R	R	R	R	R	R	R	R	R	R	R
Soil community	R	R	R	R	R	R	R	R	R	R	R	R	R	R
Aquatic communities (other than fish)	R	R	R	R	R	R	R	R	R	R	R	R	R	R
Pest resistance to pesticides	G	R	G	G	G	G	G	G	G	R	R	R	R	R
Defensive expenditures (DE)														
DE for pesticide handling and spraying	R	R	R	R	R	R	R	R	R	R	R	R	R	R
DE for safe drinking water	R	R	R	R	R	R	R	R	R	R	R	R	R	R
Purchase of organic food	R	R	R	R	R	R	R	R	R	R	R	R	R	R

[a]Houndekon et al. (2006) took chronic poisoning partly into account in their estimates, but it is impossible to know the extent to which this was done. Indeed, they merely asked farmers to state how much money they spent on medication, consultations and loss of working days during the year, without specifying the type of health effect (acute or chronic, and, for chronic effects, the corresponding illnesses)

Fig. 2.7 The Abandoned Union Carbide Pesticide Plant, Bhopal, India. This production site gave probably the most dramatic pesticide industry accidents of the history (Mishra et al. 2009). This disaster led to the death of several thousands of people and induced long-term disorders in humans, together with damage to the soil, animals and plants that could probably be estimated at several billion of US$ (Unmodified photography by Bhopal Medical Appeal, under Creative Common License CC BY-SA (https://creativecommons.org/licenses/by-sa/2.0/))

2.8.3 A Re-evaluation of the Overall Costs for the United States at the Start of the 1990s

As authors sometimes evaluate different impacts, we felt that it would be possible to perform a more complete evaluation of the external cost of pesticide use in the United States at the start of the 1990s (Table 2.14). For this purpose, we used the estimates of Pimentel et al. (1992), but we (i) actualized some external costs already estimated by these authors, e.g. honeybee and pollination losses, (ii) corrected some of their costs by taking additional data into account, e.g. bird losses, (iii) included several costs that were not evaluated by Pimentel et al. (1992), e.g. deaths due to chronic poisoning, the purchase of organic food, and (iv) removed costs that were theoretical rather than actual, e.g. wells and groundwater monitoring and decontamination, economic shortfall due to crop contamination and the disposal of contaminated crops. We ended up with a cost of US$35.2 billion (2013) (Table 2.14), a value 2.5 times higher than the original value of US$13.6 billion (2013) estimated by Pimentel et al. in 1992 (Table 2.12).

This new estimate is more complete, but it remains conservative because a number of costs, e.g. the loss of reptiles, amphibians, soil and aquatic communities and wild vertebrates other than birds and fish, the costs of acute and chronic poisoning, the purchase of bottled water and purifying devices to protect consumers against pesticide exposure, are still not included. In addition, we decided to remove from the overall sum the costs of monitoring and decontaminating wells and groundwater, and the economic shortfall due to crop contamination (Table 2.14). We removed these economic shortfalls due to crop contamination because they were conditional on the absolute respect of United States regulations, which would be unrealistic (Pimentel 2005). Pimentel et al. (1992) calculated the cost of monitoring and decontaminating all wells and groundwater, even though these activities were not actually carried out. As indicated above, it should be borne in mind that some of the human health costs to society would disappear if all wells and groundwater were effectively cleaned. If we take some of these costs into account, the overall costs would probably have been between US$35.2 billion and US$39.5 billion (2013) at the end of the 1980s/start of the 1990s.

2.8.4 Most, If Not All Overall Costs Are Out-of-Date

The articles reviewed here were retrieved from more than 30 years of studies on the costs of pesticide use. Over this period, there has been a massive, rapid change in pesticide use, as a consequence of changes in governmental legislation, i.e. the establishment of higher standards for pesticide registration, and efficiency issues, i.e. due to the exponential increase in pesticide resistance within pest and pathogen populations. This has led to a change in the panel of active ingredients used, which is currently very different from that employed 10, 20 or 30 years ago. DDT, one of the most noxious pesticides ever used, was one of the first agents to be banned, initially in the United States in 1972, and then in most other countries. In Europe, as in the United States, older pesticides are being reassessed one-by-one, to ensure that they meet the new regulatory standards (Damalas and Eleftherohorinos 2011). This re-registration process has already resulted in a substantial decrease in the number of pesticides available on the market: in an 8 year period (2001–2008), 704 pesticides were banned in Europe, 26 % of which were insecticides, 23 % herbicides and 17 % fungicides (Karabelas et al. 2009). Of the 276 pesticides authorized for use in Europe in 2009, 194 existed before 1993 and 82 had been released onto the market in the last 20 years (Karabelas et al. 2009). However, two factors may limit the benefits expected from prohibition of the most dangerous active ingredients. First, resistance to pesticides has resulted in the need for higher doses to be applied. Second, pesticides are sometimes used after they are banned (Shetty et al. 2011).

In any case, the current impact of pesticides is necessarily different from that in the past. Hence, while reporting the impact of insecticide use on the decline of many grassland birds in the United States, Mineau and Whiteside (2013) wrote that their '*analysis considered bird trends from 1980 to 2003*' and that '*there is evidence that the acute lethal risk to birds was already dropping during the second half of that period*'. Indeed, Mineau and Whiteside (2006) noted that '*the lethal risk to birds*

Table 2.14 Re-evaluation of the overall hidden and external costs of pesticide use in the United States

	In million US$			
	Original estimate	Year of estimate	Updated estimates (2013)	Reference
Cost				
Human health				
Acute health effect (treatment plus loss of work)	61	1988	123	Steiner et al. (1995)[a]
Chronic (treatment of cancer)	707	1992	1192	Pimentel et al. (1992)
(loss of work for the person with cancer)	–	–	87	Own calculations[b]
Death due to acute poisoning	–	–	405	Own calculations[c]
Death due to chronic poisoning	–	–	18,000	Own calculations[d]
Environmental impact				
Domestic animal and livestock death	30	1992	51	Pimentel et al. (1992)
Increase in pesticide use due to the destruction of natural enemies	260	1992	439	Pimentel (2005)
Crop losses due to pesticide resistance	1400	1992	2361	Pimentel et al. (1992)
Colony losses due to pesticides	13	1992	22	Pimentel et al. (1992)
Honey and wax losses	25	1992	43	Pimentel et al. (1992)
Loss of potential honey production	27	1992	46	Pimentel et al. (1992)
Bee rental for pollination	4	1992	7	Pimentel et al. (1992)
Pollination losses	200	1992	337	Pimentel et al. (1992)
Crop losses due to pesticide injury	136	1992	229	Pimentel et al. (1992)
Crop losses due to the destruction of natural enemies	260	1992	439	Pimentel (2005)
Insurance of the person applying the pesticide	245	1992	413	Pimentel et al. (1992)
Fishery losses	100	2005	122	Pimentel (2005)
Bird losses	–	–	5903	Own calculations[e]
Re-establishement of endangered birds	102	1992	172	Pimentel et al. (1992)
Regulatory actions				
Monitoring and decontamination of pesticide-polluted groundwater	1800	1992	3036	Pimentel et al. (1992)[f]

(continued)

Table 2.14 (continued)

	In million US$			
	Original estimate	Year of estimate	Updated estimates (2013)	Reference
Pesticide registration, certification, cancellation, training and farm work safety	757	1991	1330	Steiner et al. (1995)[g]
Government funds for monitoring the pesticide contamination of fruits, vegetables, grains, meat, milk, water, and other items	400	2005	486	Pimentel (2005)
Pesticide monitoring in wildlife	5	1980	16	Pimentel et al. (1980a, b)
Economic shortfalls				
Crops	1000	2005	1215	Pimentel (2005)[h]
Livestock	3	1980	9	Pimentel et al. (1980a, b)
Milk	<1	1980	1	Pimentel et al. (1980a, b)
Fish	5	1980	15	Pimentel et al. (1980a, b)
Defensive expenditure				
Purchase of organic food	2900	2012	2961	Own calculations[i]
Overall cost			35,208	

[a]Cost for 1988, see Table 10.3 of Steiner et al. (1995). For the cost in 2013, we considered the lower limit of 61 million dollars in 1988

[b]Based on 10,000 cases of cancer per year (Pimentel 2005) and 3 months (90 days) of recuperation per person with a cost per day of recuperation = $80 in 2005 (Pimentel 2005)

[c]Based on 45 deaths per year (Pimentel 2005) and a cost of 9 million US$ per life in 2013 (Viscusi et al. 2014)

[d]Based on 10,000 cancers per year (Pimentel 2005), a mortality rate of 20 % amongst individuals with cancer (Siegel et al. 2014) and a cost of life of US$9 million per life in 2013 (Viscusi et al. 2014)

[e]Based on 100 million bird deaths annually (see Mineau 2005), with a cost of 30 dollars per bird (Pimentel et al. 1992). This price relates purely to recreational value. We can add a value of 5 dollars for the protection against insects provided by the birds lost (see James 1995). Hence, the cost in 1992 would be 100 × 35 = US$3.5 billion

[f]Assuming that monitoring and decontamination were actually carried out. Theoretical rather than actual cost. Not included in the overall cost

[g]The original estimate is for 1991, but expressed in 1986 US$ (see Table 10.1 in Steiner et al. 1995)

[h]Assuming that all the crops and crop products exceeding the regulatory thresholds were disposed of. Theoretical rather than actual cost. Not included in the overall cost

[i]Considering that the United States organic food market represented US$29 billion in 2012 and assuming that prices in this market are 20 % higher than the price of conventional food and that about 50 % of the incentives of consumers to buy organic food are directly linked to pesticide risk avoidance (e.g. Schifferstein and Oude Ophuis 1998). See Sect. 2.7.3

has generally declined over the last decade in most crops /…/ The reasons for this improvement vary from crop to crop, but usually entail the replacement of older more hazardous products with newer ones with lower acute toxicity to birds'. The ban on granular formulations of carbofuran introduced in 1991 (Heier 1991) and effective by 1994, in particular, probably had a considerable beneficial effect on bird survival in farmland. The estimate of 17–91 million birds killed per year during the 1980s was therefore almost certainly, as stated by Mineau (2005), the '*'worst-case' impact of pesticides on birds in an agricultural setting*'. The current impact of pesticide use on birds is probably much lower.

The cost of the impact of pesticide use on human health may not have decreased in recent years. The trend towards the use of less dangerous chemicals may have decreased the frequency and severity of acute poisoning events. However, the illnesses resulting from chronic exposure, such as cancers in particular, may take years to appear. As an example, Cohn et al. (2007) showed that DDT exposure in young women during the period of peak DDT use in the United States predicts breast cancer later in their life (Cohn et al. 2007). Cohn et al. (2015) also showed that a larger exposition to DDT in utero is associated with an increased risk of breast cancer in adult women. As the authors stated, these findings are relevant "*even in countries in which DDT is not currently used*". This delayed effect is reinforced by the fact that "*DDT remains a global environmental contaminant, even in places where it has been banned, due to its environmental persistence and semivolatility*". Illnesses due to chronic exposures may therefore occur long after the chemicals that played an active role in triggering them have been banned. This time lag effect may have resulted in such illnesses being more frequent and, thus, more costly now than they were in the past. Similarly, most of the benefits to human health of the current process of pesticide re-registration may not appear for some time.

Our synthesis shows that overall hidden and external costs ranged from US$5.4 million (2013) in Niger in 1996 to US$13.6 billion (2013) in the United States in 1992 and were strongly underestimated. Performing an updated and more complete evaluation of these costs in the United States at the start of the 1990s, we show that overall hidden and external costs probably reached the value of US$39.5 billion (2013) per year.

2.9 Conclusions and Perspectives

2.9.1 Benefit-Cost Ratio Analysis of Pesticide Use: A Necessary…

The use of pesticides is economically justified if the benefit-cost ratio of pesticide use is greater than 1, indicating that the benefits are greater than the costs. The issue of how to measure pesticide productivity has been addressed in a large number of articles within the field of agricultural economics, although most did not consider the externalities of pesticide use. Fernandez-Cornejo et al. (1998) reviewed the estimates of the marginal product of pesticide use (the product obtained from one additional unit of pesticide use expressed in $/$ pesticide expenditure). These estimates, obtained between 1963 and 1991, were highly variable, ranging from less than 1 to more than 10 and tending to decrease over time, with a mean value, since the 1980s, of about 4. All the papers by Pimentel and coworkers were based on these estimates (those of Headley 1968) and took into account a benefit-cost ratio of 4. This value has become the most widely cited benefit-cost ratio for pesticide use. Yancy (2005) proposed a benefit-cost ratio of about 3 for herbicide use. In their highly cited paper published in *Science*, Zilberman et al. (1991) noted that '*a $1 increase in aggregate pesticide expenditures has been estimated to raise gross agricultural output from $3 to $6.50*'. Based on the estimated benefits of pesticide use calculated by Gianessi (2009) and Gianessi and Reigner (2005, 2007), Popp (2011) proposed a benefit-cost ratio of about 6.5.

However, this ratio did not include the external and hidden internal costs of pesticide use reviewed above. Any fair calculation of this ratio must include not only the usual internal costs to farmers (pesticide market costs and application costs), but also the external costs and hidden internal costs corresponding to the "other internal costs" defined in Sect. 2.2 (see also Table 2.1). However, it should exclude the hidden internal costs resulting in either an increase in the usual internal costs, such as costs linked to pesticide resistance, or a decrease in benefits, such as a reduced pollination. Indeed, these last two types of cost are already accounted for in estimates of the usual internal cost of pesticides or the gross value of agricultural production.

Some of the papers estimating the overall costs of pesticide use also provided estimates of the benefits of pesticide use (Khan et al. 2002; Pimentel et al. 1980a, b, 1992, 1993a; Pimentel and Greiner 1997; Pimentel and Hart 2001; Pimentel 2005; Pimentel and Burgess 2014; Waibel and Fleischer 1998). This enabled us to re-evaluate the benefit-cost ratio of pesticide use, by calculating the overall costs to be included in this ratio as the sum of the usual internal costs, the hidden internal costs generating "other internal costs" and external costs. The resulting ratios are given in Table 2.15.

In most cases, the ratio was higher than 1 (Table 2.15), but some of the ratios obtained were close to 1 (Waibel and Fleischer (1998) for Germany, and Pimentel et al. (1992, 1993a) for the United States) and one was below 1 (Khan et al. (2002) for Pakistan), indicating that overall costs have sometimes outweighed the benefits of pesticide use in agriculture. Hence, Pingali et al. (1994) concluded that '*When health costs are explicitly considered for a risk-neutral farmer, the net benefits of insecticides applied are negative. In other words, the positive production benefits of applying insecticides are exceeded by the increased health costs*'. This may have been the case, even in developed countries. Based on our re-evaluation of the overall costs of pesticide use for the United States in Sect. 2.8.3, the benefit-cost ratio in this country at the start of the 1990s was 0.70 (Table 2.15). In 1992, Pimentel *et al.* concluded '*complete long-term cost/benefit analysis of pesticide use would reduce the*

Table 2.15 Benefits versus costs of pesticide use

Reference	Country	Year	Costs or benefits in million US$ (2013) per year				Benefit/cost ratio
			"Usual" internal costs	Other hidden costs and external costs	Total cost	Benefits[a]	
Khan et al. (2002)	Pakistan	2002	532.67	186.38	719.05	610.19	0.85
Fleischer (1999); Waibel and Fleischer (1998); Waibel et al. (1999)	Germany	1996	1309.38	196.46	1505.84	2198.74	1.46
Pimentel et al. (1980a, b)	United States	1980	8855.42	1534.96	10,390.38	34,473.00	3.32
Pimentel et al. (1992, 1993a, b)	United States	1992	6914.00	10,038.01	16,952.01	26,983.00	1.59
Pimentel and Greiner (1997); Pimentel and Hart (2001)	United States	1997	9520.47	8707.08–9006.95	18,227.52–18,527.42	38,080.00	2.06–2.09
Pimentel (2005, 2009); Pimentel and Burgess (2014)	United States	2005	12,152.66	8985.55	21,138.21	48,610.65	2.30
Pimentel et al. (1992) and our calculations[b]	United States	1992	6914.00	31,404.00	38,318.00	26,983.00	0.70

[a]Estimates of the benefits provided in the papers of Pimentel et al. come from a 1:4 ratio of direct costs: benefits (Pimentel et al. 1978; Headley 1968)
[b]The values for "Usual" internal costs and benefits are from Pimentel et al. (1992). The values for the other hidden costs and external costs are from our own calculations (see Table 2.14)

perceived profitability of pesticides'. The re-analysis of their data shows that the profitability of pesticides has, indeed, undoubtedly been overestimated in the past. Hence, pesticide use, at the doses applied, may have entailed costs exceeding the profits generated.

2.9.2 … Yet Difficult Approach

When estimating the benefit-cost ratio of pesticide use, we need to bear in mind the alternative farming system to which pesticide use may be compared. Only benefit or cost items differing between the two types of agriculture should then be considered. For instance, conventional food production with pesticide use is often compared with organic farming, as pesticide use is lower in organic systems. In this review, we decided to include the purchase of organic food in the external cost of pesticide use (see Sect. 2.7.3) because (i) the alternative mode of production is not necessarily organic farming, e.g. it could be farming based on genetically modified crops, and (ii) the price premium of organic food would probably decrease considerably in a totally organic farming system.

The estimates of the benefits used to determine the benefit-cost ratio in the previous section were restricted to internal benefits, i.e. agricultural production. They did not include external benefits, such as reduced morbidity and mortality or a decrease in biological invasions (Felsot 2011). The estimation of external benefits is a difficult task that has been attempted by few authors (but see Felsot 2011). One of the difficulties is that the list of external benefits may, like that of external costs, be very long. For instance, conventional agriculture based on chemical pesticides has a positive effect on the activity of research laboratories in chemistry, the chemical industry, chemical sellers, agricultural advisors specializing in chemical usage, chemical waste disposal and treatment. It even has a positive effect on research into the cost of pesticide use, e.g. such as the analyses on which this review is based and this review itself.

As for costs, the most meaningful way to describe the external benefits of pesticide use is to compare conventional agriculture involving pesticide use with an alternative farming system. Only the benefit items differing between the two types of farming considered should then be compared. For instance, when comparing pesticide use as a tool for integrated pest management or organic farming, food production is often considered to be constant between strategies and is not considered as an adjustment variable. Thus, the external benefits, such as positive health effects linked to sufficient food production, are also common to the different strategies considered. However, other external benefits, such as the positive effects on health of a high sanitary quality of food, side effects on invasion biology, and the positive economic consequences of a developed pesticide industry compared to the developed work force in the field may differ between modes of agricultural production.

2.9.3 Chronic Exposure, Severe Illnesses and Death: The Cornerstones of Externalities

Our literature review provided evidence to suggest that hidden and external costs have been underestimated. The key parameter is probably the cost of illnesses and deaths due to pesticide use, notably due to chronic exposure. The benefit-cost ratio may easily fall below 1 if the costs of chronic illness and acute fatal poisoning events due to pesticide use are taken into account, because human life is clearly of great value. Our re-analysis of the data of Pimentel et al. suggested that each percent of cancers attributable to pesticides was associated with a cost of about 20 billion dollars annually.

Unfortunately, it is very difficult to estimate the cost of chronic diseases. A relationship has been found between exposure to some pesticides over a number of years and several severe illnesses (see Baldi et al. 2013). Several reviews and/or meta-analyses of case-control and/or long-term epidemiological surveys have shown that (i) occupational exposure is associated with an increase in the frequencies of Parkinson's disease (Van Maele-Fabry et al. 2012), amyotrophic lateral sclerosis (Malek et al. 2012), non-Hodgkin lymphoma (Schinasi and Leon 2014), the impairment of several neurobehavioral functions (Mackenzie-Ross et al. 2013), disorders of the reproductive system (notably low sperm concentration and quality) (Martenies and Perry 2013; Mehrpour et al. 2014) and several cancers (Alavanja and Bonner 2012; Alavanja et al. 2013) and (ii) the risks of brain cancer, leukemia and lymphoma in childhood are also significantly associated with parental exposure to pesticides (Vinson et al. 2011; Van Maele-Fabry et al. 2010, 2013).

However, the development of most illnesses, including cancers in particular, is generally multifactorial. Hence, despite the significant association between pesticide exposure and such illnesses, it is difficult to prove a causal effect of pesticides. According to Andersson et al. (2014), the conclusion of Dich et al. (1997) warning that '*few, if any of the associations* (between pesticide exposure and cancers) *can be considered established and causal*' still holds in 2014, for most, if not all long-term human disorders. Even if certain pesticides were clearly proved to be involved in these disorders, their contribution relative to other factors would still be difficult to determine. There may also be a general reluctance of the epidemiologists to compute and publish the health burden attributable to specific factors. Doing so 'takes epidemiologists as impartial scientists and thrusts them more clearly into the political arena of public health' (Steenland and Armstrong 2006). This, together with more technical causes, probably explains why we found no study providing scientifically based estimate of the number of cancers and other severe illnesses that could actually be attributed to pesticide exposure, not only among farmers, but also for the whole population.

In March 2015, the International Agency for Research on Cancer held a meeting in Lyon. This World Health Organization agency concluded that the herbicide glyphosate (Fig. 2.8), the insecticides malathion and diazinon were probably

Fig. 2.8 In March 2015, the herbicide glyphosate – contained in the widely known Roundup herbicide by Monsanto – has been classified "probably carcinogenic to humans" by the International Agency for Research on Cancer (Unmodified photography by Mike Mozart, under Creative Common License CC BY (https://creativecommons.org/licenses/by/2.0/))

carcinogenic to humans and that the insecticides tetrachlorvinphos and parathion were classified as possibly carcinogenic to humans (Guyton et al. 2015).

Based on the increasing body of evidence suggesting a tight association between some cancers and pesticide exposure, attributable risk estimates may be proposed soon. This would make it possible to revise, either upward or downward, the estimate of 0.5–1 % used by David Pimentel and coworkers as the basis of their estimations over the last 35 years. In any case, such data would bring us closer to the actual overall costs of pesticide use and would provide policy makers with tangible elements to guide their decisions.

Meanwhile, our re-evaluation of past benefit-cost ratio of pesticide use in various countries reveals that the costs of pesticide use might have outreached its benefits in the past, e.g. in the United States at the start of the 1990s. We finally advocate that the key impact to be evaluated is the illnesses and deaths due to chronic exposure to pesticides. Taking into account the costs they generate could drastically decrease the benefit-cost ratio of pesticide use. The quantification of this key cost is therefore urgently required for a more accurate evaluation of pesticide use and for regulatory purposes.

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