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Tributyltin and Booster Biocides

A socio-economic impact assessment of NERC funded research

July 2016

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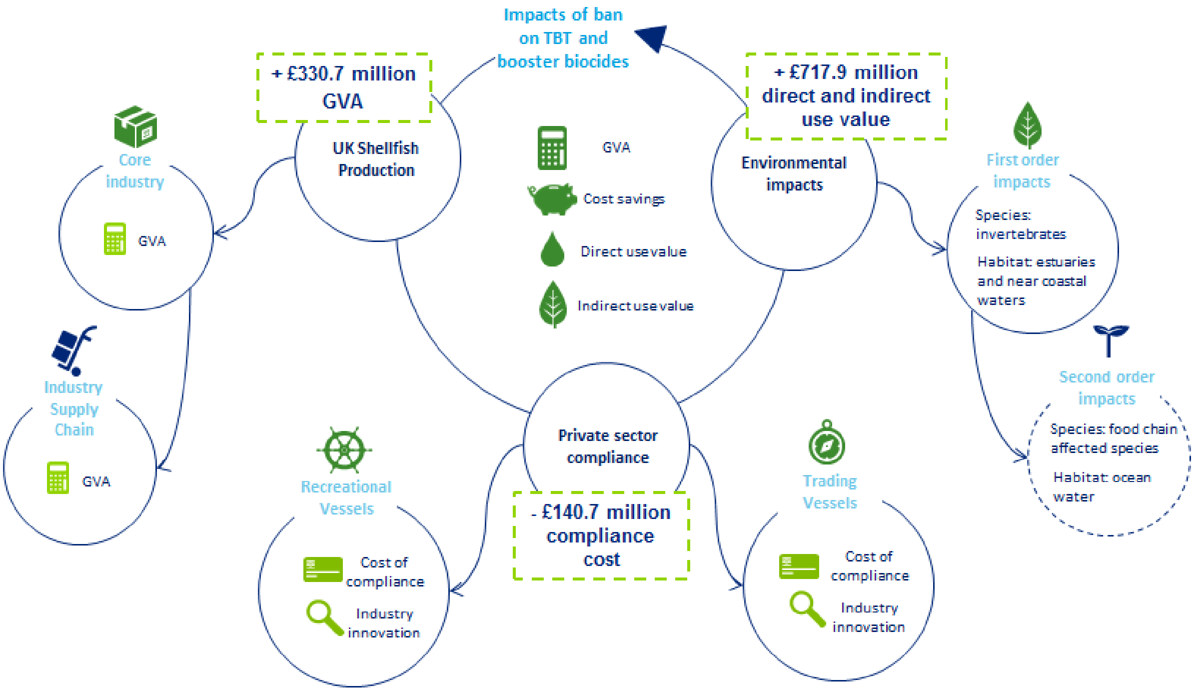
Executive Summary

The Natural Environment Research Council (NERC) has made material scientific contributions in the area of biofouling research, having provided an estimated £3.9 million in funding for related research since 1981. This research was instrumental in developing the evidence base upon which subsequent bans of harmful pollutants such as tributyltin (TBT) were based. The ban of these antifoulants, which were causing severe damage to the marine ecosystem in the United Kingdom and internationally, has led to positive socio-economic impacts.

Total Impact

In total, it is estimated that over the period 1975-2014, the ban on TBT generated positive net impacts worth an estimated £907.9 million in today's prices.

Impact framework, total impacts



Source: Deloitte

The bans on harmful antifouling substances such as TBT have supported market impacts stemming from additional production in the UK shellfish industry as well as the cost to the private sector to comply with the ban. Environmental non-market impacts were also supported through direct and indirect use value of the organisms and habitats protected as a result of the bans.

The impact model developed as part of this study suggests that over the period 1975-2014 the effects are significant:

- **UK shellfish production**: an additional £330.7 million in GVA is estimated to have been supported through direct production of the industry and also its supply chain[1]. Had the ban on TBT and booster biocides not been implemented, UK shellfish production would have been severely constrained due to reproductive failures and shell deformities.
- **Private sector compliance**: a total cost £140.7 million was incurred by recreational and commercial vessels through compliance with the ban. In the short-term, a cost was incurred to

[1] 77% of these impacts are direct, with the remaining 33% stemming from the supply chain impacts (indirect).

use less effective antifoulants. However, in the longer term, the bans spurred innovation, which resulted in new antifoulants which had a greater lifespan than TBT. This 'innovation factor' resulted in positive benefits and lessens the overall cost of compliance.
- **Environmental impacts**: a total of £717.9 million non-market benefits are supported through both the recreational value of the species protected through the ban, as well as the indirect use value that the environmental process of nutrient cycling contributes to the marine ecosystem.

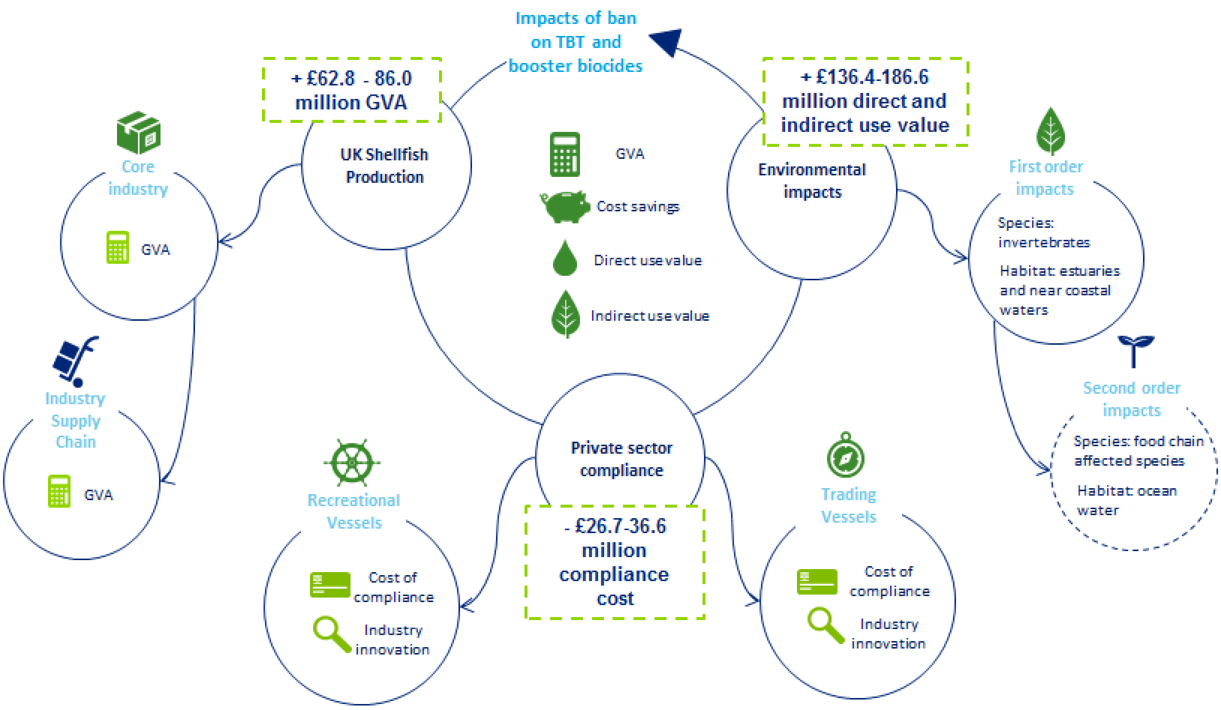
These impacts demonstrate that the investment in NERC science is generating positive net benefits to the UK economy and society. The non-market benefits supported highlight the value of the environment and emphasize the importance of wider benefits generated through environmental science research.

NERC Impact

The total impacts stated above have been apportioned to NERC in order to estimate the proportion of net impacts that are directly attributable to NERC science. Attribution of impacts to NERC science is based on the proportion of scientific papers that have been acknowledged as having made a contribution to TBT legislation in two third party evaluations of the history of TBT. This yields an attribution range of 19% - 26%.

It is estimated that over the period 1975-2014, NERC funded research on TBT supported positive net impacts worth £172.5 million - £236.0 million in today's prices and attributable specifically to NERC science.

Impact framework, NERC impacts



Source: Deloitte

The three categories of impacts can be examined individually in terms of their apportionment to NERC science:
- **UK shellfish production**: of the £330.7 million in GVA is estimated to have been supported through direct production of the industry and its supply chain, £62.8-86.0 million of this value is attributable to NERC science.
- **Private sector compliance**: of the £140.7 million compliance cost incurred by recreational and commercial vessels, £26.7-36.6 million of these costs can be attributed to NERC.

- **Environmental impacts**: a total of £717.9 million non-market benefits are supported through both the recreational value of the species protected through the ban as well as the indirect use value. £136.4-186.6 million of this is attributable to NERC science.

Over the appraisal period, NERC spend on TBT research is estimated to be £3.9 million. This input spend can be used with the apportioned benefits and costs to calculate two benefit cost ratios (BCRs). The first BCR assesses the value for money of NERC research specifically and is calculated as: Net Impacts Apportioned to NERC / NERC spend on TBT research. This yields a range of 44.1 – 60.3.

The second assesses the value for money of the TBT regulation and is calculated as: Net Benefits Apportioned to NERC / (Costs Apportioned to NERC + NERC spend on TBT research). This yields a BCR ranging from 6.5 – 6.7.

1 Introduction

1.1 Study scope and objectives

The Natural Environment Research Council (NERC) is the leading funder of environmental science research, training and innovation in the UK. The organisation is responsible for promoting and supporting basic, strategic and applied environmental research to advance scientific knowledge and technology in the UK, as well as improving public awareness of environmental issues. NERC science strives to understand how we can benefit from natural resources, build resilience to environmental hazards and manage environmental change, while ensuring the future prosperity and wellbeing of both the UK and global community.

NERC draws on both public and private funding sources to support its researchers, with public funds primarily sourced from the Department for Business, Innovation and Skills (BIS). It is important for NERC to demonstrate accountability to both BIS as well as HM Treasury for the impact of the public funds it receives and distributes, since its activities and funding decisions are independent of government.

It is within this context that NERC commissioned Deloitte in spring 2016 to undertake an independent assessment of the socio-economic and environmental impacts of NERC science in two specific research areas: biofouling and water as a resource. This report presents the findings and methodology relating to NERC's research on biofouling, the use of antifoulants and their impact on the marine ecosystem. A separate report has been produced which presents the findings relating to water as a resource.

1.2 Study Method

A discrete impact analysis has been undertaken by Deloitte in each of the two topic areas set out above. Each study is independent in that it does not consider impacts outside the remit of, or the interaction between, each topic area.

Evidence on each area's economic and wider impacts is based on an analysis of individual Research Excellence Framework (REF) submissions and case studies, reports by NERC, secondary research papers and interviews with industry experts. The collected evidence was then applied to a theory of change logic framework in order to map research outputs to outcomes and impact. In doing so, the focus has been on identifying the 'final good' which was produced by actors in the economy using NERC science. Impacts have been assessed with respect to whether they accrue to the public sector, private sector and environment. The specific assumptions and methodology underpinning the impact assessment is provided in the subsequent chapters.

The impact analysis undertaken and tools and techniques applied have drawn on the principles of appraisal and evaluation as set out in the Government's Green and Magenta Books. To this end, the analysis makes considerations for factors such as net benefits and additionality. This assessment is therefore in line with and methodologically consistent with the Green and Magenta books. Nevertheless, it should be noted that the purpose of this report is different from both the Green and Magenta Book purposes, as it is neither a business case for funding nor an end-to-end evaluation which evaluates likely impacts ex-ante before the investment is made, and post-ante once the investment programme has completed. The evaluation of impacts relating to science are also different given the fact that the outcomes of scientific research are uncertain and unpredictable. Such research is exploratory and scientists are judged not against a pre-agreed set of deliverables, but rather the quality of their research methodology. To this end, it is not possible to judge scientific research funding against a predetermined set of outputs, which therefore prevents the use of an end-to-end evaluation. Further considerations that must be made when evaluating the impacts of scientific research are detailed in Section 4.

1.3 Acknowledgements

Deloitte would like to acknowledge and thank the following people for their contributions to this report:

- John Peasland, Department for Business Innovation and Skills;
- Dr. Massimiliano Volpi, NERC;
- Alex Duffey, NERC.

1.4 Limitations of the research

The scope of this analysis is limited to the thematic area agreed between NERC and Deloitte. As a result, impacts associated with other areas of NERC research are not considered and those impacts presented cannot be interpreted as being the total contribution of NERC science.

In addition, primary research and data collection to evaluate impact of individual research projects fell outside the scope of this project. As such, identified impacts and their quantification are based on existing evidence and discussions with stakeholders – data and evidence received has not been validated by Deloitte. The suite of impacts identified and assumptions used are therefore guided by the existing, available evidence, not necessarily comprehensive, but should be treated as indicative.

Equally, it should be noted that while NERC-funded research is making a material contribution to achieving impacts, this study recognises that other research and funding bodies are also contributing to the knowledge stock in these areas. As such, impacts are presented as both total impacts and also NERC attributable impacts. Attribution of impacts to NERC science is based on the proportion of scientific papers that have been acknowledged as having made a contribution to TBT legislation in two third party evaluations of the history of TBT regulation. Further details on the methodology used to apportion impacts are provided in section 2.

On a technical note, where data is available, the quantification of impacts accounts for both costs and benefits and is presented as net impacts.

2 Biofouling and the Use of Antifoulants

2.1 Introduction to biofouling and antifoulants

Biofouling refers to the settlement and accumulation of marine microorganisms, plants, algae or animals on manmade surfaces in water. Biofouling can affect a number of different water-based activities and infrastructure elements; however, its negative impacts are most often associated with the shipping industry.

As a vessel moves through the water, it is subject to drag based on both its shape and the surface roughness on the hull. When biofouling growth occurs on a ship's hull, it increases its surface roughness and thereby increases its drag.[2] Increased drag results in both operational and economic inefficiencies. Operationally, it lowers a vessel's manoeuvrability and speed, while leading to higher costs due to increased fuel consumption, use of manpower, material and dry docking time.[3] Considering that a layer of algal slime only 1mm thick will increase roughness by 80% and cause a 15% loss in ship speed, it is clear how critical antifouling practices are to the shipping industry.[4]

For thousands of years, people have used antifouling methods to protect vessel hulls; the Ancient Greeks used lead sheathing and copper nails, while Arab shipping vessels are thought to have applied a lime and mutton fat coating to vessel hulls.[5] More recently, copper has been applied to vessel hulls in the form of sheathing and paint. Copper-based 'free association' paints were seen as the most effective antifouling agent available on the market up until the 1960s. However, they were limited by the fact that the antifoulant coating was unable to release constant concentrations of biocide from the paint surface for extended periods of time.[6] The function of the biocide in the anti-fouling paint is to prevent the settling of organisms on the hull and to poison the organisms that do. [7] As a result, it would release the biocide rapidly at first, but required frequent reapplication. This limitation led to the incorporation of tributyltin (TBT) into copper based antifouling paints beginning in the early 1960s.

TBT is a type of organotin compound, which has been produced since the late 1940s. Incorporating it with the copper-based 'free association' antifoulant created a self-polishing paint that ensured a more constant release of biocide and reduced repainting frequency.[8] The use of TBT antifoulant paints grew rapidly in the 1970s, due to both its effectiveness as an antifoulant and the longer lifespan it offered between reapplication.[9]

2.1.1 The negative environmental implications of TBT

TBT was highly effective and as a result, became one of the most widely used antifoulants for both commercial and recreational vessels.[10] However, the negative environmental impacts it caused were greatly underestimated. The remainder of this sub-section 2.1.1 provides a narrative

[2] Chambers, L. D. et al. (2006), Modern Approaches to marine antifouling coatings. Surface & Coating Technology v. 201 p. 3642-3652.
[3] Ibid.
[4] Price A. and Readman, J. (2013) Booster biocide antifoulants: is history repeating itself? In: Late lessons from early warnings: science, precaution, innovation. Part B - Emerging lessons from ecosystems. The European Environment Agency. Part 12. 297-310.
[5] Ibid.
[6] Hyder Consulting (2006), Literature Review on the Characteristics and Potential Environmental Impacts of Non-TBT Antifouling Paints.
[7] Bray, S. (2006), Tributyltin pollution on a global scale. An overview of relevant and recent research: impacts and issues. Centre for Environmental Sciences, University of Southampton.
[8] Santillo, D. et al (2001), "Tributyltin (TBT) Antifoulants: A Tale of Ships, Snails and Imposex", European Environmental Agency, Kgs. Nytorv 6, DK–1050 Copenhagen K, Denmark, copyright 1993-2003.
[9] Ibid.
[10] Ibid.

overview of how the environmental impacts related to TBT unfolded, were noticed and ultimately linked to the presence of TBT. Section 2.2 furthers this discussion by linking this narrative to the research timeline, illustrating how NERC scientists and others contributed to the increased understanding of the negative effects of TBT and its causes.

There were a number of independent observation made in the United States (US) and United Kingdom (UK) in the early 1970s of marine invertebrate species exhibiting signs of imposex – the growth of male reproductive organs in females. In the UK, Blabber (1970) recorded imposex in female dogwhelks in Plymouth Sound, UK, noting that the incidence of this was higher closer to the harbours than further away. American scientist Smith (1971) made similar observations in the American mud-snail in the US east coast.[11]

Further observations were made in France and the UK when the commercial production of Pacific Oysters was affected. In France, the 1975 culture of pacific oysters in Arcachon Bay failed to reproduce and shells of mature oysters became deformed.[12] The observations in France provided the first link between imposex in oysters and TBT. In the UK, the Pacific oyster culture failed to be introduced in the 1980s, due to reproductive failures and shell deformities very similar to those observed in Arcachon Bay, France.[13]

Despite these initial observations, the analytical capabilities to identify the cause of imposex were not available until the 1986 when imposex could be linked to the presence of boats and vessels in near shore marine water. [14] A growing understanding of TBT impacts on the environment revealed that the compound persists in water and in sediment, killing sealife other than that attached to the ship hulls and possibly entering the food chain. TBT was found to cause shell malformation and thickening, prevention of larvae release (therefore reducing the ability to reproduce) and inhibition of invertebrate development. Some fish and bird populations were also indirectly affected by TBT via the food chain, not only through exposure and bioaccumulation, where the developmental rate of fish embryo is slowed and their immune system is weakened, but also through restrictions in food supply due to decreases in diversity and abundance of benthic invertebrates.[15]

Although limited, there is also evidence of some immunosuppression in mammals, such as sea otters and dolphins, and some public concern over the harmful effects of TBT on human health, through contamination of drinking water and seafood.[16] Some residues of TBT were widely detectable in US seafood from fish farmed in TBT-treated cages, but the estimated human intakes were significantly below levels thought to be of concern for human health.

2.2 Regulatory measures and the contribution of NERC science

Although the full effects of TBT and booster biocide contamination on the marine environment were eventually recognised, legislation banning its use was delayed in coming forward and was implemented incrementally as new information came to light. Initially, the hazardous effects of TBT were underestimated due to both technical and socio-economic reasons. As Price and Readman highlight (2013)[17]:

- It was incorrectly assumed that an environmental quality target of 20 ng/l would be sufficiently protective;
- Persistence, accumulation and wider dissemination in the environment were underestimated;

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[11] Ibid.
[12] Ibid.
[13] Waldock and Thain (1983) cited in Santillo, D. et al (2001).
[14] Alzieu et al (1986), cited in Santillo, D. et al (2001).
[15] Giacomello, A., Guha, P., Howe, P., Jones, K., Matthiessen, P., Shore, R., Sullivan, C., Sweetman, A., Walker, L. (2006), The benefits of chemical regulation – Four case studies. A report for Defra, CEH, BGS and Lancaster University.
[16] Antizar-Ladislao, B. (2008), Environmental levels, toxicity and human exposure to tributyltin-contaminated marine environment: A review. Environment International volume 34, issue 2, pages 292-308.
[17] Price A. and Readman, J. (2013) Booster biocide antifoulants: is history repeating itself? In: Late lessons from early warnings: science, precaution, innovation. Part B - Emerging lessons from ecosystems. The European Environment Agency. Part 12. 297-310.

- Organotin bioaccumulation, or the accumulation of a substance in a living organism, was underestimated;
- The geographic scale of the problem was not fully appreciated due to inadequate data collection; and
- Imposex observed in 1970 in predatory gastropods in Arcachon Bay was considered acceptable – it had no immediately obvious economic cost – whereas the failure of the oyster stocks soon after was not acceptable.

The timeline set out in Figure 1 provides a detailed summary of (1) UK and European Union (EU) legislation, and (2) the research which provided the evidence to inform policy decisions. The legislation timeline highlights how after the initial ban on TBT use for small vessels was implemented (1985 and 1989), booster biocides which initially were used as a TBT substitute (e.g. Irgarol 1051), were later found to be causing similar environmental effects to TBT and were later banned (2001).

Simultaneously, details are provided on the progression of TBT research and highlight the material contribution that NERC science has made over the decades. This includes initially establishing a correlation between shipping activities and imposex in dogwhelks, to evidencing the effectiveness of legislation banning the use of booster biocides two decades later.

Figure 1 Policy and Research Timeline relating to TBT and Booster Biocides

Policy Timeline

1985: UK Environmental Quality Target (EQT) introduced requiring achievement of 20 ng/ L for UK water. Controls on sale of TBT paints for use on small vessels <25m

1987: Total ban on use of TBT paint and all other triorganotins on vessels <25m and fish cages in the UK

1987: Paris Commission proposes PARCOM Recommendation 87/1 which called for ban on retail sales for application to pleasure boats and fish cages.

1988: Paris Commission proposes PARCOM Recommendation 88/1 which called for restrictions relating to docking activities and hull maintenance operations.

1989: EQT (1985) replaced by an **Environmental Quality Standard (EQS)** of 2 ng/L for TBT

1989: EC Directive 89/677/EEC bans TBT on small vessels less than 25m

2001: UK removes Irgarol 1051 from the market and other harmful metallic booster biocides for small vessels, including diuron

2001: IMO International Convention on the Control of Harmful Anti-fouling Systems. Adopted in 2001 and banned the use of TBT on all vessels after 2003 and required its absence as an active coating on all vessels after 2008 globally.

Research Timeline

There were a limited number of publications that hinted at the environmental toxicity before TBT became widely used as an antifoulant, however research at this time showed an incomplete picture and/or misleading data which masked the true extent of the problem. Moreover, research at this time focused on observations such as imposex or malformations and did not initially establish the link to TBT. The most promising study was published in 1974 (Ritchie et al), which highlighted the impact of TBT on egg-laying of freshwater molluscs, however the findings were not reported through the relevant channels so it did not come to the attention of those involved in marine environmental protection.

Shortly after when the oyster population in Arcachon Bay, France, failed to reproduce (1975), French researchers focused on the role that TBT antifoulants could have had, first proving the link between TBT and the oyster population (Alzie et al, 1980, 1982; Alzie et al. 1986). This research then informed UK tests at the Fisheries Laboratory (later CEFAS), which were able to link oyster shell malformations to the presence of TBT (Waldock and Thain, 1983), when it was previously believed to have come from excessive amounts of sediments (Key at al,. 1976).

In 1986 Bryan et al. published a study which highlighted how the rapid rise of imposex in the Plymouth Sound coincided with increases in TBT applications and that dogwhelk populations closest to shipping activities exhibited the highest levels of this condition. This study is recognised as marking wider acceptance of the central role of TBT in the decline of oysters and dogwhelks after it was published in the *Marine Pollution Bulletin* (1986). This research was produced while Bryan was a researcher at the Marine Biological Association (MBA), an organisation supported by NERC. When NERC's Plymouth Marine Laboratory (PML) was established in 1988, a number of MBA staff moved across to join PML.

In 1987 Langston et al, a researcher from NERC funded Plymouth Marine Laboratory, reported seasonal concentrations of TBT in marina waters within Poole Harbour, UK. He also published evidence on the persistence and impacts of TBT in sediments. In the same year, Cleary (a PML funded researcher) and Stebbing highlighted the accumulation of organotins in the sea surface, which posed a threat to buoyant fish larvae.

In 1988 Gibbs et al, a researcher from NERC funded Plymouth Marine Laboratory found that sterilisation of female dogwhelks occurs at TBT concentrations as low as 3-5 ng/l TBT and that almost all females were affected once concentrations reach 10 ng/l TBT. This is well below the EQT target set in 1985 and supported the case for replacing the EQT (1985) with the EQS (1989).

In the years following, both Gibbs and Langston continued to build the research base of evidence that linked the presence of TBT to imposex in dogwhelks and sediment contamination in the UK (1991, 1993, 1994, 1996).

After the TBT ban was implemented, booster biocides such as Irgarol 1051 became more widely used and also had damaging environmental impacts. Readman et al. (1993), who is currently a Fellow at PML although not at the time of this study, was the first to report the contamination of Irgarol 1051 in Cote d'Azur, France. This study led to a host of others including research conducted at PML and NERC's National Oceanography Centre (NOC), which subsequently developed the evidence base on the damaging effects of booster biocides. In 1999 the Assessment of Antifouling Agents in Coastal Environments (ACE) project began, which was led by PML and developed a picture of booster biocide concentrations across Europe.

In 1997 Scarlett et al., a PML researcher, published findings on the occurrence of Irgarol 1051 in certain UK estuaries. This research highlighted the similar effects it had to TBT and contributed to the evidence based leading to the 2001 ban of this harmful booster biocide.

More recently, NERC funded researchers have focused on evidencing the effectiveness of legislation banning booster biocides. In 2006 Cresswell et al. (PML funded) conducted a study on water samples from UK areas with previously high levels of Irgarol 1051 contamination. The study provided empirical evidence that regulation had successfully lowered concentrations below the EQS standard of 24 ng/L.

Source: Deloitte elaboration with reference to Santillo, D. et al (2001), Giacomello et al. (2006) and Price A. and Readman, J. (2013).

The review of the legislation and research timeline evidences the extensive contribution that NERC science has undertaken in the area of biofouling. NERC science has often provided the necessary evidence base specific to UK marine environments of TBT contamination (e.g. Bryan et al. 1986 and Langston et al. 1987), and has provided some of the first studies, which have the effect of catalysing further research to come forward (e.g. Readman et al. 2003). As a result, it is concluded that NERC science made a material contribution to the policy decisions banning TBT.

Since 1981, NERC has invested over £3.9m (2016 prices) in support of TBT related research[18]. Additionally, since 2001, NERC funded over £360,000 in grants to universities for research related

[18] NERC's input has been estimated through a review of TBT related publications authored by a sample of NERC-funded scientists identified through an EEA report on TBT by Santillo, D. et al. Consequently, the value is an indication of NERC's minimum contribution to the overall impact.
The review captured the time period over which each scientist was publishing material related to TBT. The number of TBT related publications they authored was compared to their overall publishing activity during the period to estimate the proportion of their time spent on the subject.

to bio-fouling. NERC also invested in fundamental research with future potential applications to anti-fouling research. For example, it invested £950,000 in grants related to barnacles' ecology (barnacles being one of the main sources of bio-fouling). NERC also invests in research on other biofouling organisms, such as bacteria and seaweeds.

The remainder of this study thus focuses on the benefits and costs stemming from the TBT regulation, showing that NERC science was a fundamental contributor to these policy decisions, without which no regulation would have been passed in the UK, or not until a much later date.

The logic for this can be traced to the fact that – without NERC's substantial contribution – legislation would not have been passed or – at least – would not have been passed in the same timeframe. The ordinary, risk-based legislative process usually requires that a fully formed evidence base needs to be available before costly legislation is passed. Normally, all the pieces of the puzzle need to be brought together before the legislator feels confident that there is a need to act, with measurable risks and credible causality mechanisms fully demonstrated. In these circumstances, no action will take place until all these elements are provided[19]. This means that each scientific contribution is essential to action being taken and, under these circumstances, the meaningfulness of attribution criteria is debatable:[20] all contributions can validly claim they are essential.

However, such an approach clearly makes attribution impossible and this conflicts with the requirements of the Green Book, which require attribution of appropriate portions of an impact to its input components. Therefore, the analysis proposes a tentative apportioning of NERC's contribution to the benefits and costs of the regulation. This apportioning is based on the proportion of scientific papers that have been acknowledged as having made a contribution to TBT legislation in two third party evaluations of the history of TBT.[21] These suggest that 19-26% of the scientific basis for the TBT ban originated from NERC. These proportions are used as a rough order of magnitude estimate for apportioning– this is not precise and is for illustrative purposes only[22].

2.2.1 Technologies available post-TBT ban

Following the 1987 ban of TBT use on small vessels, copper became the predominate antifoulant used, due to its toxicity to many marine organisms.[23] Nevertheless, there are some organisms that are resistant to copper, including several types of algae. This led to booster biocides being added to the copper antifouling paint to improve its effectiveness. These included the herbicides Irgarol 1051 and diuron. As has been noted in further detail in section 2.2, Irgarol and diuron were banned from use in the UK in 2001 after their toxicity and persistence in the environment were traced to its usage.

The use of booster biocides in the first alternatives available to small vessels after the 1987 ban of TBT is an important point to note. While these alternatives were available to small vessels, their use was subsequently banned in 2001, before the second TBT ban, which affected large vessels. As a result, the same alternative antifoulants available to small vessels were not available to large vessels over a decade later. This point relates to the innovation factor – had booster biocides not been banned, there would have been no impetus for the private sector to innovate again after the

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To estimate the value of this time, we have referred to Plymouth Marine Laboratory's 2014 annual report and calculated the total staff costs by the total number of employees to produce an average annual cost for a marine scientist. This average was multiplied by the estimated time spent on TBT by each author to arrive at a value for NERC's investment.

[19] The exception to this are rare circumstances in which environmental legislation is passed on the basis of a precautionary approach,

[20] This can be seen as a situation where the production function behind new legislation exhibits something similar to lexicographic preferences, where the amount of legislative output is a function of the lowest level of any of the necessary inputs, rather than a combination of all inputs.

[21] Santillo et al. (2001) and Price and Readman (2013)

[22] It should be noted that such an apportionment is not normally possible: besides the logical challenges provided by the previous discussion on the legislator's requirements for a complete picture, there is the very practical problem that only a minority of environmental pieces of legislation will have had enough prominence to become the subject of detailed analyses on how they came about.

[23] Dafford, K., Lewis, J., Johnston, E. (2011), Antifouling Strategies: History and regulation, ecological impacts and mitigation. Marine Pollution Bulletin, Vol. 62, pages 453-465.

second TBT ban, since the large vessels could use the same alternative available to the small vessels. Since this is not the case, an innovation factor is applied after both bans, representing the fact that private sector investment was required after the second ban to develop a suitable alternative.

After the second TBT ban in 2003, the use of non-toxic and 'natural' alternatives became more widespread on the market. Additional research and development was devoted to the development of foul-release coatings which used silicone elastomers, waxes or silicone oils.[24]

2.2.2 Consequences of the ban

The successive bans on TBT led to material impacts to the private sector and the environment. There were no material impacts identified as accruing to the public sector as part of this assessment.[25]

Private Sector Implications

Both costs and benefits accrue to the private sector as impacts of the successive bans of TBT. The most prominent impacts and those included in this study are highlighted below. Specific details on the assumptions underpinning these costs and benefits and provided in the detailed calculations outlined in Chapter 4.

- **Shellfish production**: research indicated that it was not only dogwhelks and oysters that were damaged by TBT contamination, but other organisms were also suffering, such as clams, fish larvae, and intertidal organisms.[26] As the presence of TBT and booster biocides declined after the bans, research indicated that the existence and production of these species increased, which can be measured as additional value added (GVA) to the UK economy.[27]
- **Compliance costs to vessels**: compliance costs are incurred by vessel owners in order to adhere with the regulations banning TBT by using an alternative antifoulant. Research has indicated that alternative antifoulants immediately available after the successive TBT bans were not as effective as TBT and therefore caused vessel owners to incur additional costs to prevent biofouling.[28]
- **Innovation**: although an initial cost of compliance is incurred by the private sector, this is known to be temporary given alternative antifoulants that entered the market after the ban, which were equally or even more effective than TBT based antifoulants. In this sense, the TBT bans spurred innovation in the private sector market for antifoulants: due to the ban, the private sector invested into developing alternatives which were as effective as TBT, but did not contain the harmful chemical. This finding is supported in the work by Giacomello et al (2006) et al, which notes that the first TBT ban "led to the development of new non-biocidal approaches to fouling control […] so there were associated economic benefits with this."[29] This effect is accounted for in the quantification of benefits and costs as an 'innovation factor', or a period over which the compliance cost declines and eventually reduces to zero or becomes a cost *saving*. Section 4 provides more information, including evidence to substantiate the presence of an innovation factor, and methodological details on how it was accounted for in the quantification.

Environmental Implications

The environmental impacts can be understood within the context of ecosystem services – services provided by the natural environment that benefit people. These services can range from food and fuel, to others which are more abstract such as recreation and the appreciation of nature. In economic terms, these can be understood as providing either a use or non-use value. Use values

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[24] Ibid.
[25] Note that UK regulations on TBT do not apply to warships, naval auxiliary or other ships owned or operated by the State and used on government non-commercial services. Source: Merchant Shipping (Anti-Fouling Systems) Regulations 2009, Marine Guidance Note, Maritime and Costal Guard Agency).
[26] Eventually, the total tally of organisms that developed imposex as a result to exposure to TBT reached 150 species. Source: Vos *et al.,*2001, cited in Price A. and Readman, J. (2013).
[27] Evans et al. (2000) cited in Price A. and Readman, J. (2013).
[28] Chambers, L. D. et al. (2006), Modern Approaches to marine antifouling coatings. Surface & Coating Technology v. 201 p. 3642-3652.
[29] Giacomello et al. (2006), page 87.

can be either direct use (e.g. recreation or food) or indirect (e.g. water regulation or nutrient cycling). Non-use values refer to existence[30] or bequest[31] values for future generations.[32]

This assessment has identified two specific use-values which are supported as a result of the TBT bans:

- **Value of marine environment for divers and anglers**: the marine environment of the sites that divers and anglers visit provide a direct use-value to these users. This use-value is understood to be the annual recreational value that users receive, which is estimated using a travel cost choice experiment method. This study then estimates the additional value that was achieved as a result of the TBT ban.
- **Nutrient cycling**: nutrient cycling is an important environmental process which is completed by invertebrates, including those affected by TBT. Nutrient cycling is nature's way of recycling organic and inorganic matter into forms that the ecosystem can use for food and other nutrients. Further detail on the role of invertebrates in nutrient cycling is provided in Section 4. Nutrient cycling provides an indirect use-value to society and the value associated with the reduction of TBT and increase in nutrient cycling can be measured. Note that nutrient cycling occurs in estuaries, which are typically different to the marine environment protected sites used to assess the recreational use value.

This chapter has provided the context within which the benefit cost analysis has been undertaken. In addition, the materiality of the contribution that NERC science has made in the area of biofouling has been evidenced and this is an important underlying assumption that is carried through the remainder of this report.

[30] Existence values reflect the benefit people receive from knowing that a particular thing, such as an environmental resource or organism, exists.
[31] Bequest values reflect the value of satisfaction from preserving a natural environment for future generations to use and enjoy.
[32] Defra (2007), An Introductory guide to valuing ecosystem services.

3 Impact assessment methodology

The methodology used to conduct this impact analysis follows the principles of the HM Treasury's Green and Magenta Books. It takes into account factors such as gross to net benefits, additionality and discounting where appropriate. Further details on the assumptions underpinning this methodology are provided in the sections below and in the detailed benefit cost assessment in Chapter 4.

The structure of the impact analysis is informed by an impact framework, which links NERC research to outputs and outcomes, and identifies the costs and benefits associated with these impacts.

3.1 Impact assessment framework

The impact assessment framework sets out the range of impacts (costs and benefits) that stem from the ban of TBT and booster biocides. This framework is presented in Figure 2 and can be understood as follows.

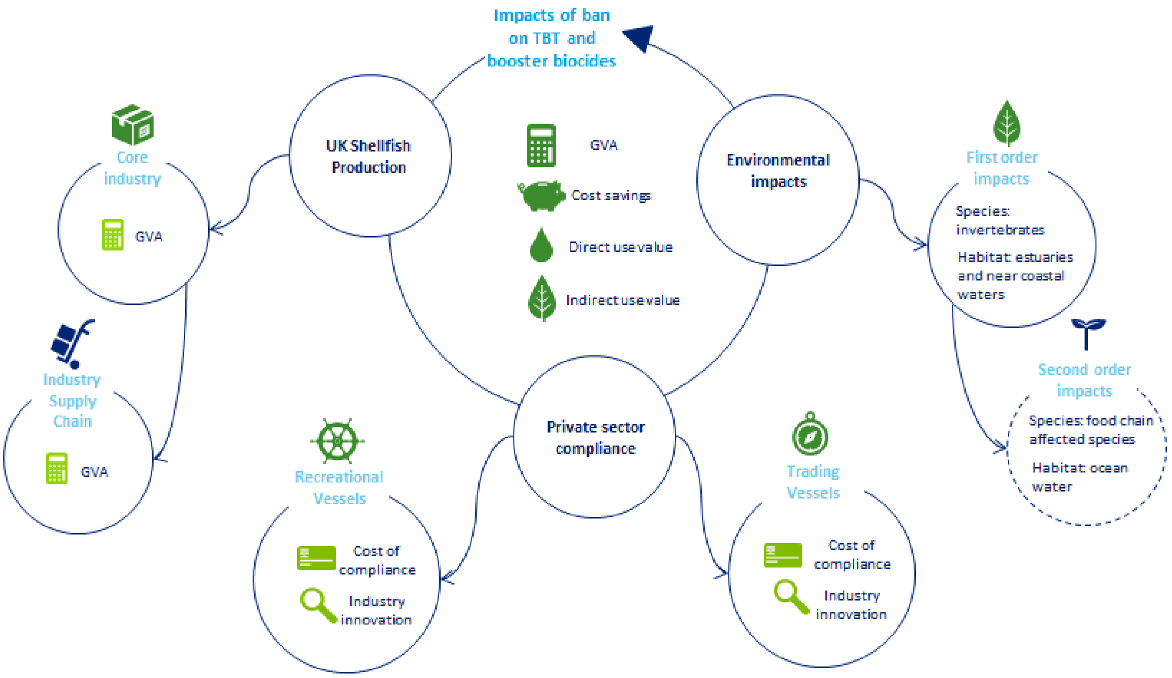
From the private sector perspective, impacts relate to shellfish fisheries and farming (hence, shellfish production for short) and costs generated by environmental compliance. Shellfish production is measured in terms of direct gross value add (GVA)[33] generated by the industry itself, as well as that supported through its supply chain. Compliance with the new regulations initially incurs a cost to industry, however it also leads to innovation and cost savings in the longer term. As mentioned in Section 2, innovation in the private sector antifoulant market results from the need to develop alternative antifoulants to replace those containing TBT. Evidence presented in Section 4 highlights that the alternatives which were eventually created were as effective, and in some cases more effective, than TBT. In terms of impact, this results in the cost of compliance declining once the new antifoulants have been adopted to zero or even lower, resulting in a positive cost saving benefit. These impacts are assessed for vessels smaller than 25m in length, which were affected by the first UK TBT ban (1987) as well as vessels larger than 25m in length, which were affected by the second UK TBT ban (2003/2008).

From a general public perspective, environmental impacts are considered in terms of both their direct and indirect use values. These environmental impacts can be understood as first order impacts, which include species and habitats most affected by TBT. Second order impacts include species and habitats which are affected by TBT, but either to a lesser degree or indirectly.[34] Note that, while second order environmental impacts are recognised, they are not quantified as part of this study.

[33] Gross value add is a measure of output which adjusts for intermediate inputs to give a true measure of economic value. It is either defined as gross output (GO) less intermediate inputs, or the sum of pre-tax operating surplus (profit) and total wage payments (including taxes).

[34] As explained in Section 2.1.2, organisms directly affected by TBT and those first identified as being negatively impacted upon by the pollutant were invertebrates such as oysters, and these are considered first order impacts. However, other organisms were likely affected through the food chain and these are considered second order. Spatially, first order impacts refer to those coastal areas most affected by TBT. Although TBT was present in ocean water, studies found concentrations highest in coastal waters near harbours and these are considered first order impacts from a spatial dimension.

Figure 2 Impact Assessment Framework



Source: Deloitte

3.2 Quantitative and qualitative methodology assumptions

The detailed methodological assumptions applied to the quantification of impacts are specific to each impact area and provided in the relevant sections of Chapter 4. Below is a summary of the general assumptions which have been applied across all impact analysis and quantification.

- All prices are stated in 2016 values, re-based using consumer price index (CPI) data from the ONS.
- All impacts are appraised over the period 1975 – 2014 and final impact figures are presented as total costs and benefits summed over the appraisal period.
- Overall impacts are presented as 'net', taking into account both costs and benefits.
- Impacts are presented at the UK level and international impacts have not been considered within the scope of this study.
- A conservative approach for inferring causality has been taken. This study has not endeavoured to prove a causal link and therefore adopts causality assumptions that have been used in secondary research publications.
- Impacts have been calculated as both total impacts stemming from the ban, as well as apportioned impacts, which consider what proportion of total impacts can be attributed specifically to NERC science.

This research focuses on analysing the most prominent and quantifiable impacts relating to the ban of TBT and booster biocides. It does not represent an exhaustive exploration of all impacts stemming from the ban, nor does it attempt to disentangle effects from TBT compared to booster biocides, both of which resulted in similar environmental impacts and the removal of certain antifoulants from the market post-ban.

3.3 Structure of impact analysis

Chapter 4 presents the step-by-step analysis that was undertaken to quantify the private sector and environmental impacts set out in the impact framework above. Where necessary, further detail on the methodological assumptions made in assessing individual costs and benefits are provided in Annex 1.

Each impact assessment in the following section took a very similar approach and is structured as follows:

- **Methodology and rationale**: an overview of the approach taken to measure specific costs and benefits and the rationale underpinning the approach.
- **Quantification of costs and benefits**: a step-by-step description of the quantification of costs and benefits and an interpretation of the findings.

4 Benefit cost assessment

4.1 Evaluation of scientific research: the need to modify standard approaches

The methodology below illustrates an example of a benefit cost analysis applied to research. Applying benefit cost analysis to research poses two specific challenges: (1) the outcomes of research are intrinsically uncertain, and (2) science and scientific research is cumulative, making it difficult to isolate and link a specific output to one specific input. These two points are explored in further detail below.

The fact that the outcome of research is intrinsically uncertain means that it is nearly impossible to forecast what the benefits will be from commissioning a piece of research. Compared to other investment areas, such as infrastructure where benefits can be forecasted, scientific research is exploratory. In this sense, scientists are judged against the quality of their research methodology, not against a pre-agreed set of deliverables. This uncertainty and its associated information failures are the very reason why research is funded by government and largely self- governed. It is mainly other scientists who are in charge of assessing the quality of a proposed piece of research. The impossibility of knowing what exploring the unknown will deliver implies that an end to end evaluation is impossible for science, at least at the level of individual grants.

Secondly, science is cumulative, meaning that the outcome of individual pieces of research is extremely difficult to isolate and that single grants in isolation cannot achieve impact. Scientific research builds and expands upon previous knowledge and it is highly collaborative and interdependent. As such, attributing the benefits of science to an individual grant is extremely difficult and any such attribution can only be regarded as an approximation. Citations are often used as a way to try to capture some of this cumulative process.[35] This is the approach that has been taken here, where the papers cited in two reviews of the history of TBT and antifoulants have been used to attribute the impact of NERC. The share of NERC papers out of all the academic papers cited has been used to calculate an attribution percentage. Citations are arbitrary to some extent, as it would have been possible to expand the analysis to the papers cited in the papers cited in the reviews and so on, potentially indefinitely. However, the decision to stop with the papers cited in the review, excluding those cited by subsequent papers, can be justified by the fact that these are the ones most likely to be most relevant.

Notwithstanding these challenges, which require some adjustments to standard evaluation techniques, the analysis attempts to capture the impact of science on policy making. Science can have profound and unpredictable positive effects on society. The policy around the TBT ban is a fine illustration of this.

4.2 Overview of the costs and benefits

The benefit cost assessment of the UK ban of TBT is presented in the following sections. Each section is dedicated to a specific impact accruing to either the private sector or the environment.

The successive bans on TBT in the UK occurred over a time period of more than two decades, with impacts occurring at different points in time depending on which ban they are associated with and to whom they are accruing. As such, the following analyses refer to three distinct periods (Figure 3), as well as presenting impacts in aggregate over the entire appraisal period (1975-2014).

[35] Salter A.J. and Martin B.R (2001) "The economic benefits of publicly funded basic research: a critical review" Research Policy 509-533.

Figure 3 Policy and Research Timeline relating to TBT and Booster Biocides.

	Time period	Relevant impacts
Period 1: Pre-TBT ban	1975-1986	This period is used to establish a baseline of economic and environmental performance before the ban.
Period 2: Post-ban of TBT use on small (<25m) vessels	1987-2007	• Private sector impacts: UK shellfish industry • Private sector impacts: Cost of compliance for small vessels (<25m) • Environmental impacts: direct use value • Environmental impacts: indirect use value
Period 3: Post ban of TBT use on all vessels	2008-2014	• Private sector impacts: UK shellfish industry • Private sector impacts: Cost of compliance for large vessels (>25m) • Environmental impacts: direct use value • Environmental impacts: indirect use value

Source: Deloitte

The first period is used to establish a baseline for the subsequent period in order to then compare how outputs and outcomes change once the first ban on TBT is implemented. No benefits or costs due to the TBT ban are incurred during Period 1. Where relevant, Period 2 serves as a baseline for assessing impacts achieved in Period 3 after the second ban.

The benefit cost assessment of the UK ban of TBT is presented in the sections which follow. Each section is dedicated to a specific impact accruing to either the private sector or the environment.

This is consistent with previous work on the topic in a report to Defra (Giacomello et al, 2006), and we have attempted to follow their evaluation model as closely as possible, updating it as appropriate.

4.3 Private sector impacts: UK shellfish industry

4.3.1 Methodology and rationale

As highlighted in Chapter 2, evidence suggests that the UK ban on TBT booster biocides allowed for the recovery of the UK shellfish population. The method used to calculate this impact is adopted from the approach used in the study *The Benefits of Chemicals Regulation, a report to Defra* to assess the impacts of the first TBT ban in 1987.[36]

This approach quantifies the gross value add (GVA) that the production of the affected species contributes to the UK economy both directly and indirect through the supply chain, over and above what this contribution would have been in absence of any ban on TBT. This approach has been adapted and extended to account for the second TBT ban and therefore considers the possible uplift in production between Period 1 and 2 and also between Period 2 and 3.

4.3.2 Quantification of Benefits and Costs

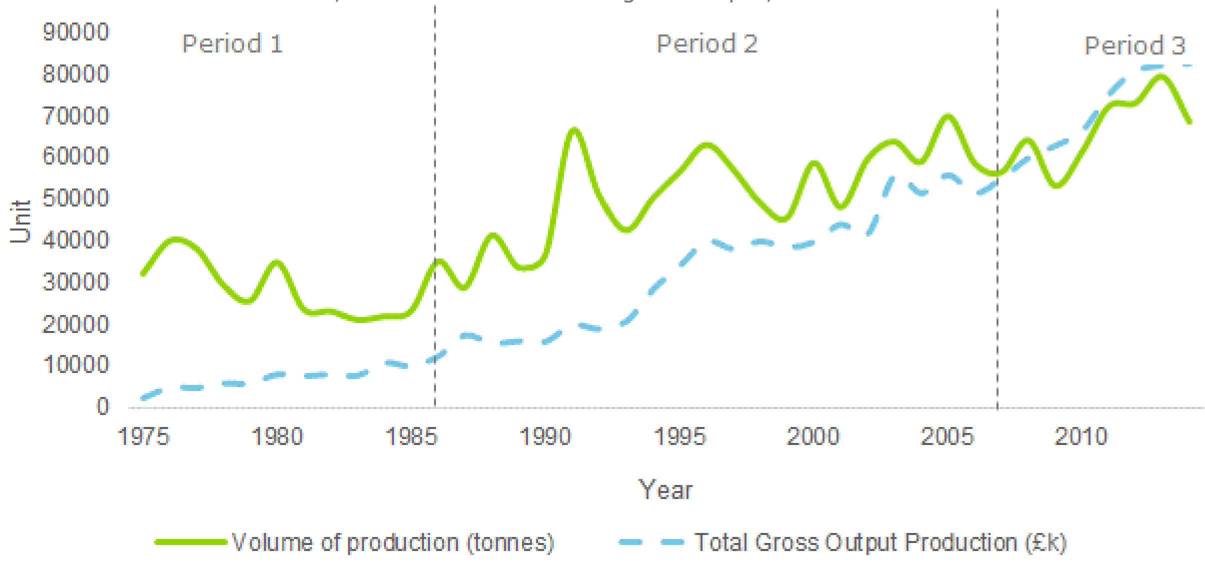
In order to calculate any uplift in shellfish production as a result of the TBT ban, it is necessary to define which species would be considered in this analysis. Research has shown, with medium and high confidence levels, which commercial shellfish are negatively affected by TBT.

[36] Giacomello, A., Guha, P., Howe, P., Jones, K., Matthiessen, P., Shore, R., Sullivan, C., Sweetman, A., Walker, L. (2006), The benefits of chemical regulation – Four case studies. A report for Defra, CEH, BGS and Lancaster University.

These include[37] [38]:
- Native oysters
- Pacific oysters
- Whelks
- Cockles
- Scallops
- Mussels

Time series of production volume and value for the species listed above was constructed using the *Annual Reports from UK Sea Fisheries Statistics Archive* on UK shellfisheries production.[39] Two time series were constructed: (1) volume of production per annum, in tonnes, by species and (2) total production value per annum, by species (Figure 4).

Figure 4 UK Shellfish Production, tonnes and £K nominal gross output, 1975-2014



Source: UK Sea Fisheries Statistics Archive. Note that the two dotted lines represent the TBT bans.

All shellfish species have been aggregated into a single measure in Figure 4 for communication and informative purposes, in order to show the trajectory of overall production volume over the appraisal period. The subsequent calculations which apply values to volume is done by species, as explained below.

Since the value of production for each species extracted from the Annual Reports is presented in terms of gross output (GO)[40], a GVA ratio of 55% was used to translate GO production values to GVA.[41] This ratio was applied to the annual GO of shellfish production, by species, translating each year's GO into GVA.

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[37] Giacomello, A., Guha, P., Howe, P., Jones, K., Matthiessen, P., Shore, R., Sullivan, C., Sweetman, A., Walker, L. (2006), The benefits of chemical regulation – Four case studies. A report for Defra, CEH, BGS and Lancaster University.
[38] Note that Periwinkles were also included in this list, however they were not included in this analysis because their production values were not consistently reported in the UK Sea Fisheries Statistics Archive over the required time series.
[39] Data was abstracted from each individual annual report. Annual reports were accessed at: http://webarchive.nationalarchives.gov.uk/20140507202222/http://www.marinemanagement.org.uk/fisheries/statistics/annual_archive.htm#2009
[40] Gross output represents the total amount of turnover or expenditure in an economy, or sector. This is not a representative measure of economic output however, as it double counts. Specifically it includes economic output plus intermediate inputs to production, which naturally represents other sector's output
[41] A GVA ratio estimates the proportion of a sector's GO which is GVA. The GVA ratio was sourced from the Deloitte's UK Input-Output model.

A Type 1 multiplier of 1.3 was then used to calculate the indirect supply chain GVA supported by the shellfish industry production.[42] This multiplier was applied to each annual GVA value, yielding a combined value of direct and indirect GVA per annum. 77% of these impacts are direct, with the remaining 33% stemming from the supply chain impacts (indirect). The annual direct and indirect GVA figures were then rebased to 2016 prices using consumer price index (CPI) data from the ONS.

At this stage, a timeline has been constructed which represents the direct and indirect economic contribution that the shellfish industries have made to the UK over the appraisal period. These figures represent gross impacts, in that they do not take into account the uplift that was achieved specifically as a result of the TBT bans.

In order to isolate and estimate the uplift in GVA that was achieved as a result of the bans, the following steps were taken. First, the average annual direct and indirect GVA contribution was calculated for Period 1, before the first ban. It was assumed that if no ban were implemented, this level of output would remain constant. It should be noted however, that this is a simplified and conservative assumption, as some research suggests that production values would likely have begun to decline as a result of TBT contamination.[43]

Second, the difference between the annual GVA contribution for each year in Period 2 and the average annual GVA contribution in Period 1 was calculated, yielding the estimated uplift in production that occurred over and above the output levels of Period 1. This exercise was repeated in order to calculate the additional uplift achieved in period 3 – the average annual GVA contribution for Period 2 was used as a baseline in order to estimate the additional uplift achieved in Period 3 on a per annum basis.

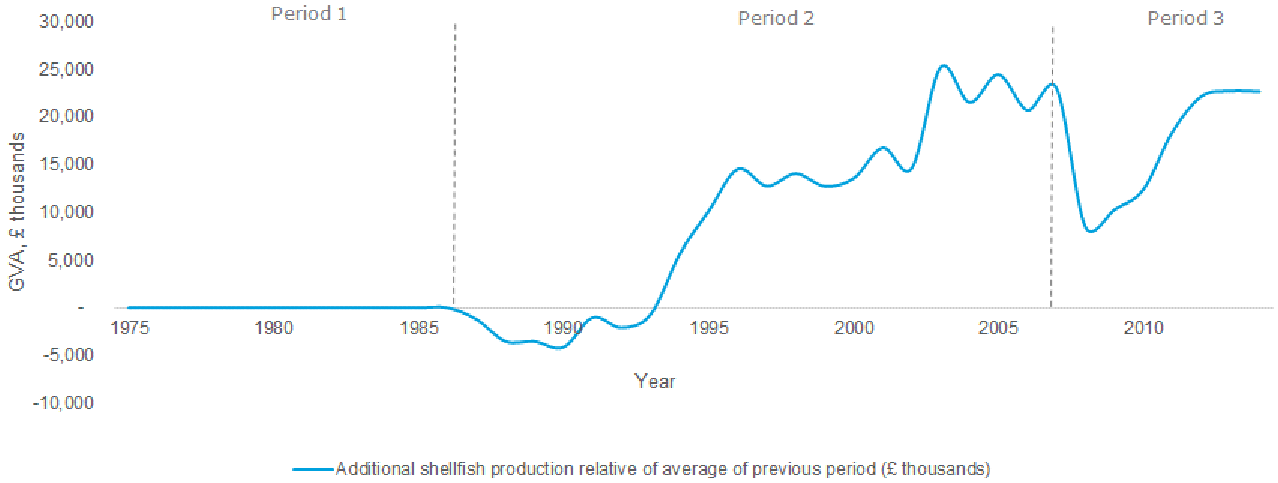
At this stage, the calculations yield a time series of the 'additional' GVA contribution achieved in Period 2 and Period 3. However, there are factors other than the TBT ban which may have led to this production uplift, such as the global financial crisis in 2008. To account for this possibility, an assumption of 50% causality was applied to the annual production uplift. This assumption is the same used in the Defra report from which this methodology was adopted and is based on two scientific studies which showed a 50% increase in native oysters in the Crouch Estuary, UK from 1986 to 1992 and a 94% reduction in TBT over the same period of time and location.[44]

Over the period 1975 – 2014, the TBT ban is estimated to have supported an additional £330 million GVA (direct and indirect) in the UK economy through the shellfish industry.

Figure 5 illustrates the additional uplift in direct and indirect GVA supported as a result of the two bans of TBT. After the first ban in 1987, the additional GVA dips into negative territory, due to the fact that the additional uplift attributable to the TBT ban (50% of the total additional uplift) is less than the average GVA contribution in Period 1. A similar trend is seen after the second ban in 2008, when the additional uplift shortly after the ban drops relative to the previous period, but quickly increases in the years that follow. This observation accords with the fact that the damage caused by TBT is likely to persist for five to ten years after a ban is implemented.[45] As a result, it makes sense to expect a lag between implementing the ban and production benefits materialising. It also is a mere representation of the way in which the calculations have been completed, since the preceding period's average production is greater than the additional production uplift, attributable to NERC, in the early years post-ban.

[42] A type 1 multiplier is a ratio that is applied to direct impacts in order to estimate what indirect impacts (i.e. supply chain impacts) will arise as a result of the direct activity. This ratio is equal to (direct impacts – indirect impacts)/ direct impacts. Therefore, using algebra to rearrange the equation, it is possible to see how multiplying the Type 1 multiplier by the direct impacts will yield the sum of direct and indirect impacts. The type 1 multiplier was calculated using Deloitte's UK Input-Output Model.
[43] Giacomello, A., Guha, P., Howe, P., Jones, K., Matthiessen, P., Shore, R., Sullivan, C., Sweetman, A., Walker, L. (2006), The benefits of chemical regulation – Four case studies. A report for Defra, CEH, BGS and Lancaster University.
[44] Studies referred to are Rees et al. 2001 and Waite et al 1991 cited in Giacomello, A. et al (2006)
[45] Giacomello, A. et al (2006)

Figure 5 Additional Shellfish Production, GVA, £ thousands, 1975-2014



Source: Deloitte calculations. Note that dotted lines represent the TBT bans.

4.4 Private sector impacts: Cost of compliance for small vessels <25m

4.4.1 Methodology and rationale

The private sector impact of the first TBT ban in 1987 relates specifically to the cost that owners of small vessels (<25m) had to incur in order to comply with the ban. The methodology used to assess the cost of compliance to small vessel owners is adapted from *The Benefits of Chemicals Regulation, a report to Defra*.

This method makes key assumptions on

1. Who are those small vessel owners likely to incur additional costs to prevent antifouling after the ban and
2. How much more are these costs over and above what they were before the ban.

The assumptions made on which small vessel owners are likely to incur additional costs after the ban are based on several considerations:

- The antifoulants available immediately after the first TBT ban were not as effective as the TBT based antifoulant and as a result had to be reapplied more often;[46] [47]
- The cost of the alternative antifoulants were not materially different to the cost of TBT;[48]
- Some recreational vessel owners would choose to accept a higher degree of biofouling rather than incurring additional cost to apply the antifoulant coating more frequently.

Quantification of costs and benefits

The first step to quantifying the cost of compliance to small vessel owners is to estimate the number of vessels <25m present in the UK during the time of the first TBT ban. The Royal

[46] Giacomello, A. et al (2006) and Chambers (2006)
[47] Note that Chambers (2006) suggests that the cost of using alternative antifoulants to TBT is double in relation to TBT-based paints due to their shorter lifespan, and Giacomello et al (2006) notes that the alternatives available immediately after the first TBT were not much different in price compared to TBT. As such, the doubling of cost is associated with its effectiveness, which can logically be assumed to be half as efficient as TBT. The average TBT antifoulant lifespan was 4 years, which suggests that the lifespan of alternatives was about 2 years. However, racing yachts accept an absolute minimum amount of fouling and therefore would not wait the entire four (or two) years before reapplying the antifoulant coating.
[48] Giacomello, A. et al (2006)

Yachting Association (RYA) publishes the Annual Watersports Participation Survey, which provides the total number of recreational vessels in the UK.[49]

To make this figure fit for purpose, it was assumed that these recreational vessels are <25m. Data from these annual surveys was sourced for years 2009-2014. To build the time series of this figure over the years in Period 2 when the impacts of the first TBT ban were relevant, the number of recreational boats was pegged to UK population data sourced from ONS. This data yields the time series in Figure 6.

Figure 6 Total number of UK recreational vessels, 1987-1994

	1987	1988	1989	1990	1991	1992	1993	1994
Number of UK recreational vessels (‘000)	1,004	1,006	1,009	1,012	1,015	1,018	1,020	1,023

Source: Deloitte calculations based on Annual Watersports Participation Survey (RYA) and Population Estimates (ONS)

It was assumed that of the total recreational vessels in the UK, those most likely to accept higher costs to avoid biofouling after the TBT ban were racing yachts.[50] This is due to the fact that they require their vessels to travel at maximum speed, which can easily be affected by the smallest amount of biofouling. Ratios from Giacomello, A. et al (2006) were used to estimate the total number of UK yachts raced, yielding the following time series (Figure 7).[51]

Figure 7 Total number of UK yachts raced, 1987-1994

	1987	1988	1989	1990	1991	1992	1993	1994
Number of UK yachts raced (‘000)	86.9	87.0	87.3	87.5	87.8	88.1	88.3	88.5

Source: Deloitte calculations based on estimates from Giacomello, A. et al (2006)

Next, the additional cost that racing vessels were likely to incur after the ban was estimated. Research conducted by Giacomello, A. et al (2006) suggests that owners of racing yachts were likely to incur an additional £537 (2016 prices) per annum in antifouling after the TBT ban. This is in line with Chambers (2006) who noted that the costs of alternatives are double in relation to TBT based paints due to their shorter lifespan.[52]

It is then necessary to determine the length of time over which the private sector had to incur the cost of compliance. This time period is dictated by the innovation factor. The innovation factor is used in the calculations to consider (1) when an alternative antifoulant that was at least as effective as TBT was released, and (2) the time it took for this product to be adopted by the market.

Research indicates that three years after the ban, alternative antifoulants such as Copper Acrylate technology were brought to market which had a similar lifespan and cost to TBT. [53] As such, the full cost of compliance (£537 per annum per racing yacht) is incurred for the first three years after

[49] This survey was accessed from: http://www.rya.org.uk/about-us/what-we-do/Pages/participation.aspx
[50] This assumption is based on consultations carried out to inform Giacomello, A. et al (2006).
[51] Giacomello, A. et al (2006) calculated that in the year of their analysis, 36% of recreational vessels are sailing yachts and that 24% of sailing yachts are raced.
[52] Chambers, L. D. et al. (2006), Modern Approaches to marine antifouling coatings. Surface & Coating Technology v. 201 p. 3642-3652.
[53] Anderson, C., (2003) TBT Free Antifouling and Foul Release Systems, Department of Marine Sciences, University of Newcastle upon Tyne.

the ban (1987 – 1989, inclusive), after which it is assumed racing yacht owners begin to adopt the alternative antifoulant. It is assumed that one third of racing yachts adopt the more effective alternative antifoulant each year, for three years, beginning in 1990 and ending in 1992 once all racing yachts would have adopted it.[54] This calculation is set out in Figure 8. Beginning in 1993, it is assumed that racing yachts are no longer incurring the cost of compliance.

Figure 8 Total cost of compliance for <25m vessels, £ thousands, 2016 prices, 1987-1994

	1987	1988	1989	1990	1991	1992	1993
Cost of compliance (£ '000)	46,618	46,710	46,842	35,230	23,569	11,815	0

Source: Deloitte calculations

As a result, it is estimated that the total cost of compliance resulting from the first TBT ban totalled £210.8 million, incurred over the period 1987-1992.

It is important to highlight that the alternative antifoulants that became widely used after the first TBT ban, such as those containing booster biocides such as Irgarol 1051 and diuron, were later found to have similarly damaging effects to TBT-based antifoulants.[55] The use of these booster biocides was subsequently banned in 2001, as noted in Figure 1, Section 2.[56]

4.5 Private sector impacts: Cost of compliance for large vessels >25m

4.5.1 Methodology and rationale

The method used to estimate the cost of compliance for large vessels (>25m) after the second TBT ban follows that used to estimate the impact on small vessels after the first ban. Conceptually, the approach identifies the number of vessels likely to have been impacted and estimates the additional cost of antifouling that these vessels would have to incur as a result of the ban. Practically, this approach is required to make allowances for the phased introduction of the second ban.

Contextually, it is important to recognise that this second ban on TBT was implemented in 2003, after the UK had banned the use of booster biocides such as Irgarol 1051. Up until 2003, large shipping vessels were still using TBT based antifoulants, so were not dependent upon the alternative antifoulants that small vessels were using after the 1987 TBT ban. As such, the 2003 TBT ban for large vessels meant that they were unable to use the same alternative antifoulants that became available after the first TBT ban, thereby requiring the private sector to invest in the development of new alternatives that were free of both TBT as well as booster biocides.[57]

The IMO Convention stated that after 2003 vessels could not apply or reapply antifoulants containing organotin compounds (including TBT) which act as biocides; however, they could continue in use if they had applied such antifoulants before 2003. Effective from January 2008 the ban stated that all vessels must not bear organotin containing antifoulants, meaning that after this point, no vessels in UK waters should be releasing these pollutants into the ecosystem.[58]

[54] Note that the three year adoption period is an estimate used in absence of any more specific data on what the actual adoption rate was. Although the lifespan of this antifoulant is equivalent to TBT and therefore 4 years, this is reduced to 3 years since it is acknowledged that racing yachts tolerate a lesser degree of fouling and are therefore likely to apply the antifoulant more often.

[55] Cresswell, T., Richards, J., Glegg, G., and Readman, J. (2006), The impact of legislation on the usage and environmental concentrations of Irgarol 1051 in UK coastal waters. Marine Pollution Bulletin, Vol. 51, pages 1169-1175.

[56] Dafforn, K., Lewis, J., Johnston, E. (2011), Antifouling strategies: History, regulation, ecological impacts and mitigation. Marine Pollution Bulletin, Vol. 62, pages 453-465.

[57] Ibid.

[58] Merchant Shipping (Anti-Fouling Systems) Regulation 2009, Maritime and Coastguard Agency, Marine Guidance Note.

Additional assumptions similar to the approach for small vessels also underpin this approach:

- The antifoulants available immediately after the second TBT ban were not as effective as the TBT based antifoulant and as a result had to be reapplied more often[59]
- The cost of the alternative antifoulants were not materially different to the cost of TBT[60]

4.5.2 Quantification of costs and benefits

The UK Department for Transport's Maritime and Shipping Statistics were used to estimate the number of large UK owned and operated vessels at the time of the IMO Convention.[61] This dataset provided the total number of trading and passenger vessels broken down by time as well as gross tonnage.[62] Using this information, it was possible to develop a time series of both number of vessels and gross tonnage of these vessels over the period 2000-2014.

A study Abbot, A. et al. (2000) was used to estimate the cost of antifouling, per tonne, before the ban was implemented.[63] The study conducted several in-depth case studies to understand the total costs incurred in order to treat a vessel for biofouling, including the operating expenses while a vessel is dry-docked to apply the antifoulant and the total expenses on the antifoulant itself.[64] These prices reflect the use of an organotin containing antifoulants and therefore establish the cost baseline before the ban. Rebasing the prices and converting them to a price per tonnage of ship, the following assumptions were established (Figure 9):

Figure 9 Antifouling cost assumptions for large vessels, per application of antifoulant

	Operating costs for one antifoulant application (£ GBP, 2016)	Cost of antifoulant for one application (£ GBP, 2016)	Total cost per antifoulant application
Costs for a 15,000 tonnage vessel	£19,900	£144,700	£164,600
Cost per 1 tonnage of vessel	£1.33	£9.65	£10.98

Source: Deloitte calculations based on information from Abbot et al. (2000)

The average cost per antifoulant application per tonnage of vessel (£10.98) can then be multiplied by the gross tonnage per annum figure of the UK trading and passenger vessel fleet.[65] However, the time series that this yields reflects the cost of antifouling if each ship were to apply a new coating of antifoulant every year. It suggests an annual cost of £110.6 million assuming that every vessel re-applies an antifoulant coating every year.

Given that the average lifespan of an organotin containing antifoulant was about four years[66], each re-application cost can be annualised over a four year period before the ban. It was assumed that 25% of large vessels applied a new coating of antifoulant each year, so that after a four year period the entire fleet has incurred the cost of one antifouling treatment. This yields an average

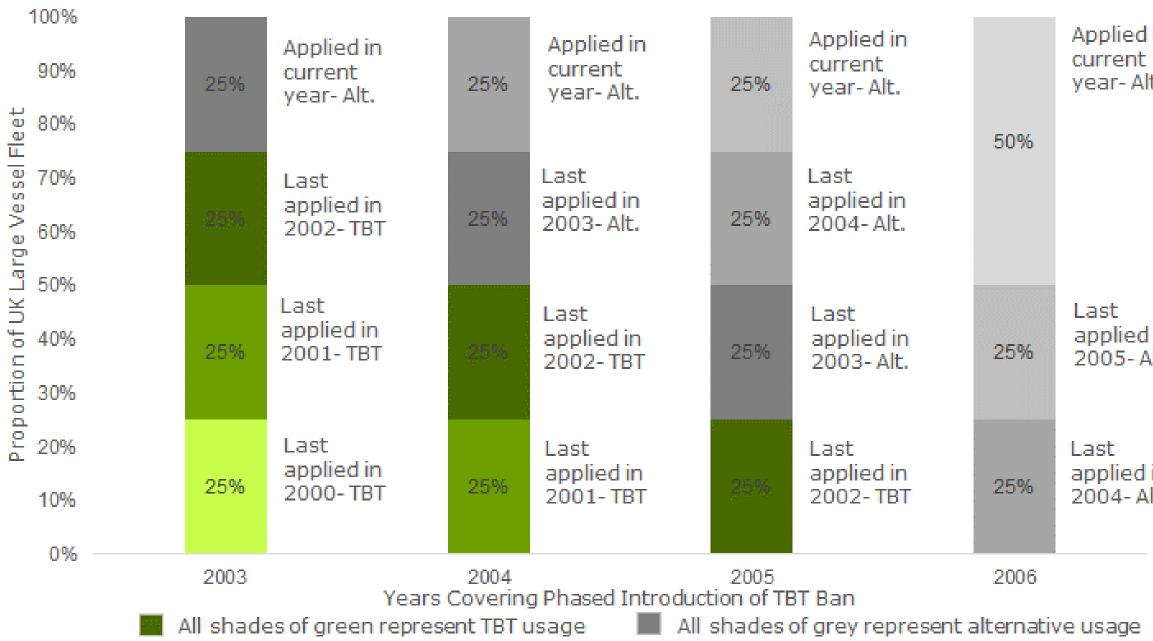
[59] Giacomello, A. et al (2006)
[60] Ibid.
[61] Department for Transport, Maritime and shipping statistics. Accessed from: https://www.gov.uk/government/collections/maritime-and-shipping-statistics
[62] It is acknowledged that there may be a small number of vessels >25m which are not captured by this dataset, however this number is likely to be negligible within the context of this quantification.
[63] Abbott, A., Abel, P., Arnold, D., Milne, A. (2000), Cost-benefit analysis of the use of TBT: the case for a treatment approach. The Science of the Total Environment volume 258, pages 5-19.
[64] Ibid.
[65] Note that gross tonnage per annum is a figure reported by the UK Department for Transport, which provides an aggregate figure for the gross tonnage of the UK's large vessel fleet in a particular year.
[66] Ibid. Note that the average lifespan of the TBT based antifoulant is suggested to be 3-5 years therefore an average of 4 years was adopted for this analysis.

cost of £27.6 million per annum for the total UK fleet of large vessels to apply the TBT-based antifoulant.

Immediately after the first phase of the ban came into effect in 2003, it was assumed that each tranche of vessels that had to re-apply antifoulant each year would use an alternative, which had a shorter lifespan. Research suggests that the average lifespan of alternative antifoulants available immediately after the ban was three years.[67] A shorter lifespan has the effect of increasing the annualised cost of antifouling, since it will be spread over a three year rather than four year period. However, it is important to recognise that vessels that applied their final TBT coating immediately before the ban took effect in 2003 would not need to switch to using the alternative antifoulant until 2007, since the TBT lifespan is four years. As a result, there is a four year period when varying proportions of the UK large vessel fleet is incurring different annualised antifouling costs.

The phase introduction of the TBT ban is visually represented in Figure 10. As shown, in 2003, the first year of ban, it is assumed that the first tranche of ships will adopt the alternative antifoulant. The size of this tranche is estimated to be 25% of the large vessel fleet given the four year lifespan of TBT and the assumption that the number of vessels that re-apply antifoulant each year is spread evenly over that period. An additional 25% of vessels adopt the alternative antifoulant each year until 2006 when all vessels are assumed to then be using the alternative. In this final year, half of the vessel fleet is assumed to re-apply the antifoulant coating – the final tranche which was still using TBT, and the first tranche to apply the alternative which has reached the end of its 3 year lifespan.

Figure 10 Phased introduction of TBT ban and adoption of alternative antifoulant, 2003 – 2006



Source: Deloitte calculations

It should be noted that in 2006, although half the vessel fleet is assumed to re-apply the antifoulant coating compared to a quarter, the costs are annualised and smoothed over time. In other words, it does not mean that a greater proportion of compliance costs will be borne in 2006 compared to the two years before and after when only 25% of the fleet is re-applying the antifoulant coating.

[67] Bader-Eldin (2011), Hull Roughness and antifouling paint, Arab Academy for Science and Technology and Maritime Transport.

The phased introduction visualised above can then be used to allocate the annual cost of compliance to the different tranches of vessels. Those vessels still using TBT will incur a cost annualised over four years, while vessels which have adopted the alternative antifoulant will incur costs annualised over three years. The annualised cost is calculated as:

- **For TBT usage**: (Annual gross tonnage of UK fleet * £10.95) / 4 * % of total vessels using TBT in given year
- **For alternative usage**: (Annual gross tonnage of UK fleet * £10.95) / 3 * % of total vessels using alternative in given year

These calculations are illustrated in Figure 11, yielding a time series of the gross annual cost of antifouling for large vessels before during and after the ban.

Figure 11 Gross annual cost of antifouling for large vessels, £ millions, 2002 – 2014

	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014
Cost of anti-fouling	26.7	37.5	41.3	46.9	56.1	63.4	68.8	71.6	73.7	74.7	74.4	70.3	59.6

Source: Deloitte calculations

It should be noted that the time series in Figure 11 represents the gross annual cost and therefore has yet to be considered against the counterfactual and yet to take into account the innovation factor. Also, the figures presented in the timeline are also a factor of the annual gross tonnage of the UK large vessel fleet and will therefore also fluctuate according to this variable.

Similar to the methodology used to estimate the cost of compliance to small vessels, an innovation factor was applied to the cost of antifouling. Research shows that by 2010, the brand Ecospeed released a Surface Treated Composite antifoulant, which combines a hard, durable non-toxic hull coating with in-water hull cleaning technology.[68] This antifoulant coating only needs to be applied once during for the lifetime of the vessel and can then be wiped clean in the water, thereby avoiding the costs associated with dry-docking and reapplication.

The innovation factor is applied to the time series in Figure 12 based on the assumption that the product is gradually adopted by the market, by a quarter of the large vessel fleet per year, beginning in 2010, meaning that from 2010 onwards, a declining proportion of vessels are still incurring the annualised cost of antifouling.[69] By 2013 it is assumed that all vessels would have adopted this more effective antifoulant and are therefore no longer incurring the annualised cost.

[68] Hydrex Underwater Technology (2010), White Paper: Ship Hull Performance in the Post-TBT Era.
[69] Note that this assumption has been used in absence of more precise data on the market uptake of this antifoulant.

Figure 12 Gross annual cost of antifouling for large vessels including innovation factor, £ millions, 2002 – 2014

	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014
Cost of anti-fouling with inno-vation	26.7	37.5	41.3	46.9	56.1	63.4	68.8	71.6	55.3	37.4	18.6	0	0

Source: Deloitte calculations

The time series is then constructed by subtracting the gross annual cost of antifouling, including innovation, after the ban from the counterfactual. The counterfactual assumes that TBT continues to be used by the large vessel fleet and that firms therefore incur a cost of antifouling annualised over a period of four years rather than three.

The annualised cost for the counterfactual is calculated as: (Annual gross tonnage of UK fleet * £10.95) / 4.

Subtracting the time series in Figure 12 from the counterfactual on an annual basis yields the net additional cost of antifouling, or the compliance cost incurred as a result of the TBT ban (Figure 13).

Figure 13 Cost of compliance for large vessels, £ millions, 2003 – 2014

	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014
Net additional cost of anti-fouling	2.9	5.9	9.4	14.0	15.9	17.2	17.9	0	-18.7	-37.2	-52.7	-44.7

Source: Deloitte calculations. Note that negative costs can be interpreted as benefits due to innovation.

It illustrates that when the innovation factor is taken into account, benefits are generated in the long run. In the time series above, these benefits are shown as the negative costs which occur in years 2011-2014. Over the period 2003-2014, the total cost of compliance in relation to large vessel totals amounts to a benefit of £70.1 million in net savings, due to the innovation factor which generates benefits that outweigh costs.

4.6 Environmental impacts: Value to Divers and Anglers

4.6.1 Methodology and rationale

The environmental impacts of the TBT ban can be assessed according to the recreational use value that divers and anglers attribute to the species and habitats that are most affected by TBT contamination. This method is based on a study completed by Kenter et al. (2013) which measured the annual recreational values that divers and anglers attribute to the presence of certain species in special marine protected areas in England, Wales and Scotland.[70] The rationale behind this method assumes that the recreational value of certain species' habitats would be

[70] Kenter, J.O., Bryce, R., Davies, A., Jobstvogt, N., Watson, V., Ranger, S., Solandt, J.L., Duncan, C., Christie, M., Crump, H., Irvine, K.N., Pinard, M., Reed, M.S. (2013). The value of potential marine protected areas in the UK to divers and sea anglers. UNEP-WCMC, Cambridge, UK.

affected if that species were either damaged or no longer in existence as a result of TBT contamination.

4.6.2 Quantification of costs and benefits

To calculate the use value of the environment affected by TBT, the number of users (divers and anglers) and visits per annum is first established. The most precise data available is that collected in Kenter et al. (2013) through an online survey.[71] This data suggests that in 2013, there were a total of 3.4m diver visits and 60m angler visits to marine protected areas throughout the UK. In the absence of data on how this figure has changed over time, this analysis assumes that the number of diver and angler visits remains constant over the appraisal period.

The use value that divers and anglers attribute to those ecosystem services that are likely to have been affected by TBT is then calculated. Kenter et al. (2013) provides the annual recreational value, per visit by a diver and an angler for specific habitats (Figure 14).

Figure 14 Annual recreational values of habitats for divers and anglers, per visit, £, 2013

	Divers	Anglers
Vulnerable species (including mussels and native oysters)	0.44	0.3
Mostly sandy or gravelly seafloor with oyster, mussel or flame shell beds	0.0	0.0
Mostly muddy seafloor with oyster, mussel or flame shell beds	0.0	0.0
Mostly rocky seafloor with oyster, mussel or flame shell beds	7.61	0.0
Total annual recreational value, 2013 prices	8.05	0.3
Total annual recreation values, 2016 prices	8.38	0.3

Source: Kenter et al. (2013)

Annual recreational values were provided for a number of species' habitats, however only those likely to fall within the first order impacts (as per the impact framework) were included in these calculations. As such, this is a conservative calculation as it is likely that the use value of some second order species was affected by TBT contamination.

After adjusting the annual values to 2016 prices, the total annual recreational value for divers and anglers is multiplied by the total number of diver and angler visits (3.4m and 60m, respectively). This yields a combined annual use value of these habitats, for both divers and anglers, of £47.5m. This value is assumed as the maximum value of these habitats, assuming no negative impacts from TBT damage.

It is then necessary to estimate how the use-value was affected by the ban and what its uplift was compared to if there had been no regulation. This is done by linking the use values to the shellfish production. Using the average annual shellfish production volumes by period, the proportion of total possible production by phase was calculated, assuming that average production in Period 3 represents the maximum production possible (Figure 15).

[71] Note that although this is the best available data, Kenter et al. note the limitations around visitor estimates, which can be accessed in section 2.2.4 of the full report.

This suggests, for example, that before the TBT ban, the UK shellfish industry was producing 43% of its maximum possible capacity.[72] The proportion of total possible production by phase figures are then applied to the absolute combined recreational value (£47.5m), yielding three separate recreational values for each period. These figures suggest what the total recreational value of the specified habitats to both divers and anglers is, assuming that it is directly linked to the quantity in existence and assumed to be produced. While these figures illustrate an uplift in recreational value over the three periods, they do not account for other factors which could be supporting the uplift in value. As such, the same 50% causality assumption used in the shellfish industry impact calculations is applied here.

Figure 15 Shellfish production and recreational values by period

	Period 1	Period 2	Period 3
Average annual production of all shellfish by time period (000', tonnes)	29	52.2	67.4
Proportion of total possible production by phase	43%	77%	100%
Total recreational value based on proportion of total possible shellfish production (£ millions, 2016 prices)	£20.4	£36.7	£47.5
Total recreational value with 50% causality (£ millions, 2016 prices)	£10.2	£18.4	£23.8

Source: UK Sea Fisheries Statistics Archive. Note that periods 1, 2 and 3 correspond to the same time periods used for the calculations relating to shellfish production.

Next, the values per time period must be spread across the time series. The total recreational value, including causality, calculated above is assumed to be the recreational value achieved in the last year of each period. It is assumed that for Period 1, before any TBT ban was implemented, the recreational value remained constant at £10.2 million. It is then assumed that the total recreational value grows linearly across Periods 2 and 3, reaching £18.4 million at the end of Period 2 and £23.8 million at the end of Period 3. Figure 16 and 17 illustrate how the recreational value timeline for each period has been constructed.

Figure 16 Recreational values, Period 2, £ millions, 2016 prices

	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999
Gross recrea-tional values attribu-table to TBT ban	10.6	11.0	11.4	11.8	12.2	12.5	12.9	13.3	13.7	14.1	14.5	14.9	15.3

	2000	2001	2002	2003	2004	2005	2006	2007
Gross recrea-tional values attribu-table to TBT ban	15.6	16.0	16.4	16.8	17.2	17.6	18.1	18.4

Source: Deloitte calculations

[72] Note that the assumption implicit in this calculation is that the industry was not limited by production capacity but instead was limited by volume of shellfish landings, which were constrained in part by TBT.

Figure 17 Recreational values, Period 3, £ thousands, 2016 prices

	2008	2009	2010	2011	2012	2013	2014
Gross recreational values attributable to TBT ban	19.1	20.0	20.7	21.4	22.2	23.0	23.7

Source: Deloitte calculations

Finally, the time series above are subtracted from the counterfactual. The counterfactual assumes that if no TBT ban were implemented, the recreational value for these species would remain at the Period 1 level across the entire appraisal period. Again, this is likely a conservative assumption since it is possible the value could have decreased further as the effects of TBT worsened.

As a result, the additional recreational use value uplift attributable to the successive bans of TBT deliver net benefits beginning after the first ban and continuing over the entirety of the appraisal period.

The total net benefits associated with the use value supported as a result of the TBT ban are £168.1 million over the entire appraisal period (1975-2014).

4.7 Environmental impacts: Nutrient Cycling

4.7.1 Methodology and rationale

The indirect-use value of the ecosystems affected by TBT can be measured by valuing the process of nutrient cycling. Nutrient cycling involves burying contaminants through bioturbating sediments, bringing new nutrients to the surface from deeper layers and processing organic matter into smaller particles and dissolved substances. Invertebrate species, including those affected by TBT, play a large role in the important ecological function of nutrient cycling. Damage to these species would negatively impact upon nutrient cycling, the impact of which can be quantified.

The method used to quantify this impact is adopted from the report Giacomello, A. et al (2006), which attributes an average value of the nutrient cycling service performed by estuarine ecosystems per hectare per annum.[73] This approach then develops a counterfactual of what the reduced value of nutrient cycling would be had a ban on TBT not been implemented and uses this to quantify the additional uplift in value supported by the TBT bans.

4.7.2 Quantification of costs and benefits

Several data points are required to quantify the value of nutrient cycling in UK estuaries. First, the inter-tidal area of UK estuaries, which is 0.5 million hectares.[74] This figure is multiplied by the annual value of nutrient cycling services performed by estuaries ecosystems, which is £21,453 per hectare per year in 2016 prices. This yields £10,727 million, which represents the total annual value of nutrient cycling in the UK.

This figure needs to then be further adjusted in order to take into account both contribution (how much of the nutrient cycling function is carried out by TBT affected invertebrate species) and also causality (how much TBT affected the ability of these species to carry out the nutrient cycling function). It is difficult to say precisely the contribution that oysters make to the nutrient cycling process, however a study from the United States by Newell, R. et al. (2005) found that oysters filtered around 1% of water in the Chesapeake Bay, United States in 1988.[75] An assumption of 1% is therefore used, which therefore assumes that only 1% of the nutrient cycling process is fulfilled

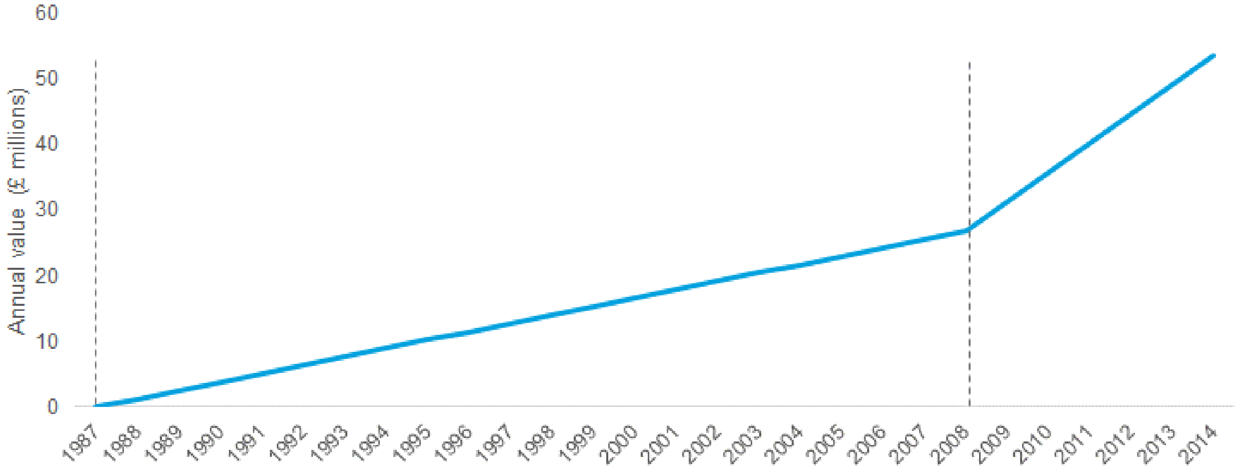
[73] Costanza et al (1997) cited in Giacomello, A. et al (2006).
[74] Giacomello, A. et al (2006)
[75] Newell, R., Fisher, T., Holyoke, R., and Cornwell, J. (2005), Influence of Eastern Oysters on Nitrogen and Phosphorus Regeneration in the Chesapeake Bay, United States. The Comparative Roles of Suspension Feeders in Ecosystems, Vol 47, NATO Science Series: IV – Earth and Environment Sciences.

by invertebrate species likely to have been affected by TBT. In line with the assumptions made by Giacomello, A. et al (2006), this method assumes that the ability of invertebrate species to carry out nutrient cycling is directly proportional to their population and therefore uses 50% causality cased on the earlier reports cited by Rees et al (2001).[76]

Taking contribution and causality assumptions into account, the annual value of nutrient cycling supported as a result of the TBT ban is estimated to be £53.6 million.

Several additional steps and assumptions are made to annualise this figure over the full appraisal period. First, it is assumed that £53.6 million is the maximum annual value of ecosystem services supported at the end of the appraisal period as a result of the TBT bans. To recognise that this value was built up gradually, after both the first and the second ban, the analysis assumes that 50% of the value (£26.8 million) is achieved at the end of period 1 (1987-2007) and the full annual value (£53.6 million) is achieved at the end of period 2 in 2014 (Figure 18).

Figure 18 Annual value of nutrient cycling, £ millions



Source: Deloitte calculations

The annualised value of nutrient cycling is illustrated in the figure above. The total value of nutrient cycling supported as a result of the TBT ban over the appraisal period (1975-2014) is £549.7 million.

4.8 Summary of impacts

4.8.1 Total Impacts

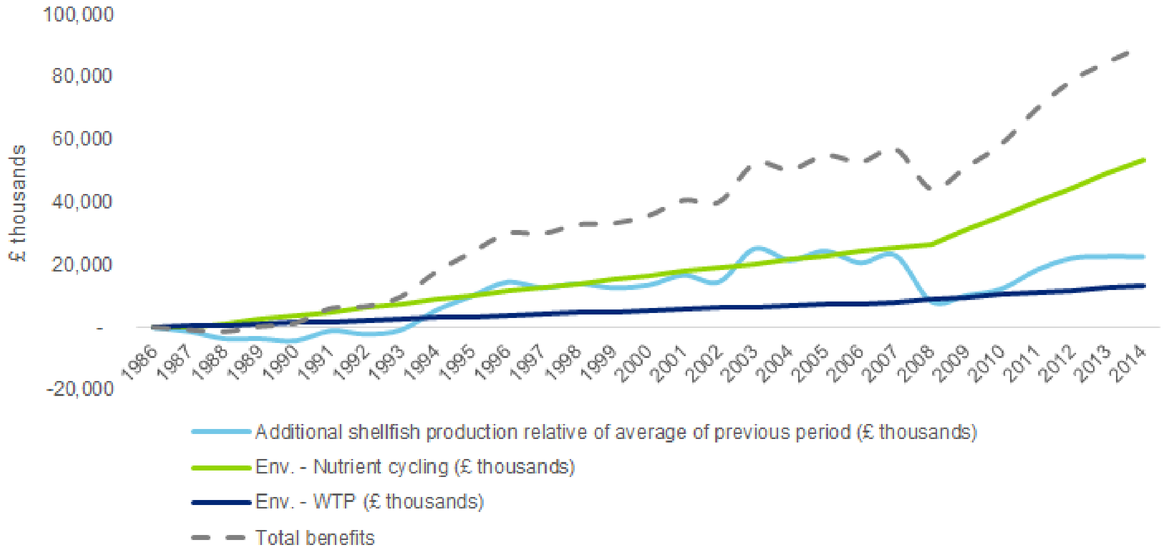
Over the full appraisal period 1975 – 2014, absolute benefits supported as a result of the TBT ban amount to £1.05 billion (Figure 19). The absolute costs incurred as a result of the TBT ban over the full appraisal period amount to £140.7 million.

As a result, the total net benefits supported amount to £907.9 million.[77]

[76] Ibid.

[77] Note that presentation of total net benefit figure does not sum exactly with net benefits and costs due to rounding.

Figure 19 Annual benefits as a result of the TBT ban, £ thousands



Source: Deloitte calculations

Interestingly, the market benefits due to increased shellfish production represent 30% of the net benefits, with the non-market benefits relating to the direct and indirect use values of the environment representing the remaining 70% of the total benefits. These findings support the wider justification for the value of environmental science, evidencing the significant public value that is derived through ecosystem services.

4.8.2 NERC Impacts

The £907.1 million in net impacts supported by the TBT bans presented above are not specific to the contribution that NERC science made to achieve the ban. Instead, the impact figure represents the contributions made by NERC as well as other researchers and policymakers. These impacts can be apportioned to NERC to give a rough order of magnitude estimate of those benefits attributable to NERC science.

A range of 19-26% of apportionment was calculated, representing the number of NERC citations as a proportion of total scientific citations in the studies Santillo et al. (2001) and Price and Readman (2013). As explained in Section 1 and 2, this is an acceptable method and the most suitable option given the challenges highlighted in apportioning impacts from scientific research, which is cumulative.

Apportioning both the benefits and costs based on the aforementioned range suggests that total net impacts attributable to NERC science over the appraisal period range from £172.5 million - £236.0 million.

Over the appraisal period, NERC spend on TBT research is estimated to be £3.9 million. This input spend can be used to calculate two benefit cost ratios (BCRs).

The first BCR assesses the value for money of NERC research specifically, and is calculated as:

- Net Impacts Apportioned to NERC / NERC spend on TBT research.

Using the range of net impacts based on apportionment, this yields a BCR ranging from 44.1 – 60.3.

The second BCR assesses the value for money of the TBT regulation and is calculated as:
- Benefits Apportioned to NERC / (Costs Apportioned to NERC + NERC spend on TBT research).

Again, using the range of benefits and costs attributed to NERC, this yields a BCR ranging from 6.5 – 6.7.

Figure 20 Apportioned net benefits and costs and BCR ratio

	Low Range	High Range
A. Benefits (£ thousands)	£ 199,221	£ 272,619
B. Costs (£ thousands)	£ 26,726	£36,573
C. Net Impacts (£ thousands)	£ 172,495	£ 236,046
D. NERC Spend (£ thousands)	£ 3,916	
BCR 1 (C/D)	44.1	60.3
BCR 2 (A/[B+D])	6.5	6.7

Source: Deloitte calculations

Annex 1: Further Considerations

The details of the impact modelling undertaken as part of this assessment is provided in the main body of the report (Chapter 4). This section provides a brief commentary on further considerations and assumptions that have been made in this analysis.

The table below highlights specific areas where further consideration has been made.

	Further Considerations
Demand side of shellfish production	The analysis undertaken in this impact assessment is based on supply side information relating to shellfish production levels in the UK. The report to Defra by Giacomello, A. et al (2006) notes that the UK shellfish market is a price taker and that a reduced national production would not cause prices to significantly rise.
Changes in TBT production and use over time	This study makes assumptions on the use of TBT antifoulants based on the fact that it was the most effective and widely used antifoulant on the market before it was banned. No data has been identified which provides a market breakdown of the use of TBT compared to other antifoulants available at the time.
Other marine life affected through the food chain	The impact assessment framework recognises that there are both first order and second order species and habitats that were likely affected by TBT. First order species and habitats have been quantified as part of this study and include invertebrates such as mussels and oysters and habitats such as estuaries and near coastal waters, which research shows as having the highest TBT concentrations. Second order impacts refer to species such as birds and dolphin which were likely affected by TBT, as well as wider ecosystems such as ocean water, which undoubtedly were also contaminated with the harmful antifoulant. However given the indirect/ diffused impact on these second order species and habitats, the impact has not been quantified as part of this study.
Uncertainty of causality assumptions	It is recognised that the causality assumption of 50% used to assess the impacts for UK shellfish production and use values could be more precise, however in absence of more robust information, this assumption is adopted from the report for Defra used to assess the impacts of TBT regulation.

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