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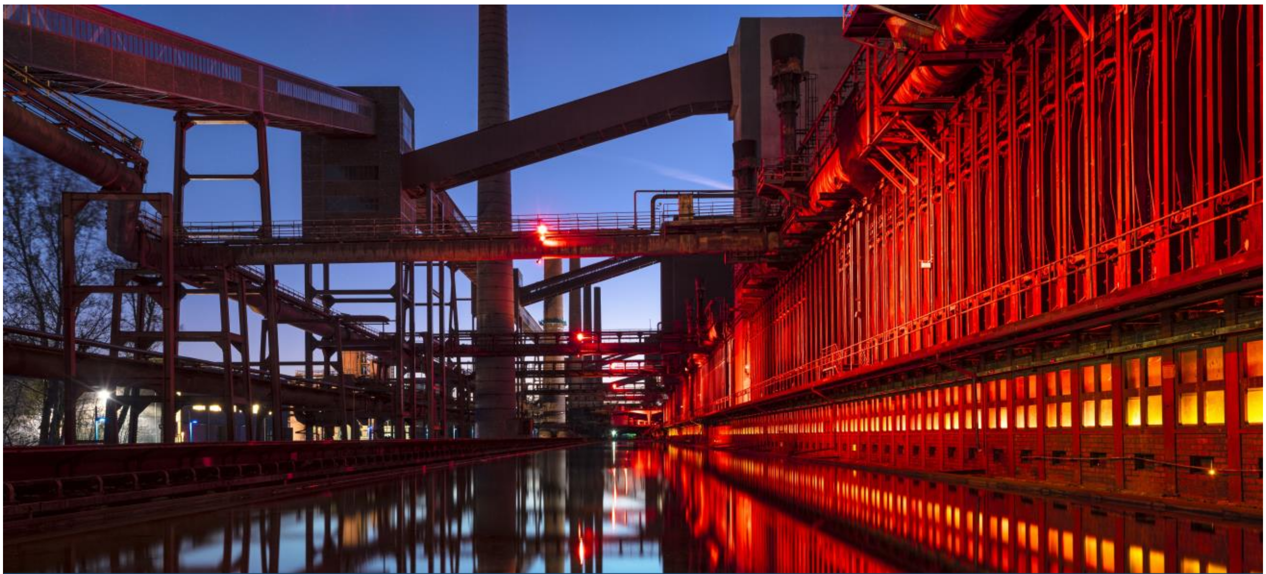

Benchmark-based allocation in emissions trading systems: Experiences to date and insights on design

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Ernst Kuneman, Christopher Kardish, Maia Hall, and William Acworth (Secretariat of the International Carbon Action Partnership)

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Disclaimer

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Acronyms

BAT	Best available technique
BF	Blast furnace
CAF	Cap adjustment factor
CARB	California Air Resources Board
CLL	Carbon leakage list
CO_2	Carbon dioxide
CSCF	Cross-sectoral correction factor
EAF	Electric arc furnace
EBB	Energy-based benchmark
EITE	Emissions-intensive, trade-exposed
ETS	Emissions trading system
EU	European Union
EU ETS	European Union Emissions Trading System
GHG	Greenhouse gas
K-ETS	Korea Emissions Trading System
MWh	Mega-watt hour
NZ ETS	New Zealand Emissions Trading System
OBA	Output-based allocation
PBB	Product-based benchmark
WCI	Western Climate Initiative

1 Introduction

Free allowance allocation is a feature of almost every operating emissions trading system (ETS). This is primarily to reduce the risk of carbon leakage, whereby production shifts to countries without comparable carbon costs or domestic producers lose market share to more emissions-intensive competitors, or to lower transition costs. Free allocation is typically provided either based on historical emissions (grandparenting) or on efficiency benchmarks (benchmarking). Grandparenting is easier to implement, because only past emissions need to be collected and verified, and it may be more palatable to covered entities at the beginning of an ETS and thus more politically advantageous. Benchmarking approaches, expressed in terms of emissions per unit of output, are more data- and resource-intensive as they require detailed production and emissions data at the firm level to develop sectoral benchmarks. Benchmarking offers the advantage of removing the link between an individual firm's historical emissions and the allowances they receive, better preserving abatement incentives for regulated entities and rewarding early-movers and best performers. Historically, markets have tended to progress in the direction of benchmarking alongside greater use of auctioning, with free allocation largely reserved for emissions-intensive trade-exposed (EITE) industries that are considered most at risk of carbon leakage.

Well-designed benchmarks approximate abatement incentives otherwise attained under auctioning while supporting industries—such as those operating in international markets—that cannot recover carbon costs. While benchmarking offers clear advantages over alternative free allocation methods, it is complex and requires careful consideration of design elements along with an in-depth understanding of the industries that will receive free allowances. Choices on the design of benchmarks and related aspects that determine final allocation will have implications for effective carbon costs of regulated entities, abatement incentives across the products' value chains, and ultimately the development of markets for low-carbon alternatives. As pressure mounts to decarbonize heavy industry and allowance budgets decline on par with steeper climate targets, getting the incentives of benchmarks right is pertinent to transition pathways under an ETS and helps optimize the distribution of fewer freely allocated allowances.

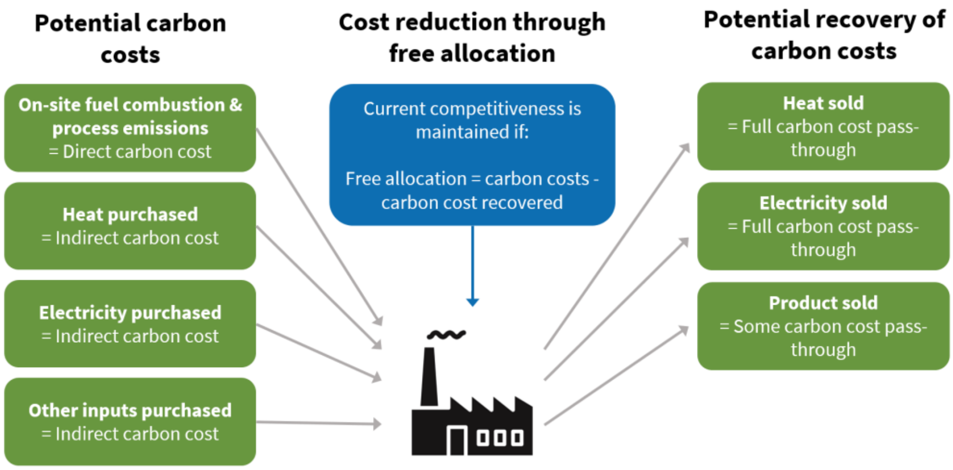
In principle, free allocation lowers the effective (i.e., average) carbon cost for industry, though it forgoes revenues that could be used to facilitate the industrial transition through the auctioning of allowances. Economic theory holds that the abatement incentives for emitters will be equivalent whether they receive allowances for free or have to purchase them at auction, treating freely distributed allowances as an opportunity cost like other resource costs (Åhman et al., 2005; Reguant & Ellerman, 2008). The allowance price therefore serves as the main driver of mitigation, with firms investing when it exceeds their marginal cost of abatement. In line with this view, the cap trajectory and not the method of allocation determines emissions abatement. However, in practice there may be numerous reasons that firms treat free allowances differently, elevating the importance of allocation policy. For instance, some researchers focusing on behavioral economics have found evidence of cognitive biases such as "endowment effects", where potential losses (costs) are perceived as greater than potential gains from improving emissions intensity of production and selling excess allowances on the secondary market (Song & Ahn, 2019; Wang et al., 2020). Future price uncertainty of allowances and other variables around compliance costs may only increase endowment effects particularly in low and high carbon price environments (Venmans, 2016). Allocation choices will impact product prices, the profits of installations, and the welfare of consumers (Åhman et al., 2005; Dallas Burtraw et al., 2001). Importantly, free allocation can leave gross margins of carbon-intensive assets intact as firms may not fully internalize the opportunity costs of freely allocated allowances as compared to the marginal cost of allowances bought at auction or in the secondary market. This could possibly delay the pace of industrial decarbonization.

Benchmarks retain abatement incentives of an ETS, but their design entails trade-offs that become more pronounced as jurisdictions move to decarbonize hard-to-abate sectors. This paper sheds light on how benchmark-based free allocation is designed across major ETSs, particularly on the implications of benchmark design choices on abatement incentives. Specifically, we draw from the experiences and design features of the European Union ETS (EU ETS), the California Cap-and-Trade Program, the Québec Cap-and-Trade System, the New Zealand Emissions Trading Scheme (NZ ETS), and the Korea Emissions Trading System (K-ETS). As ICAP's work program focuses on the linkages between allocation policy, carbon leakage, and deep decarbonization, this paper centers specifically on industrial allocation. The paper complements experiences to date from these ETSs with insights from the literature on benchmarking to highlight considerations in benchmark design with significant implications for abatement and low-carbon investment incentives.

2 Benchmark design

Industrial sectors under an ETS face direct carbon costs from their on-site emissions from fuel combustion and industrial processes, as well as indirect carbon costs from heat, electricity, and other inputs that they purchase off-site (left side of Figure 1). Free allocation primarily aims to reduce the costs imposed by the ETS that cannot be recovered. This may apply to sectors exposed to international competition with little possibility to pass on carbon costs in product prices (right side of Figure 1). These sectors are at risk of carbon leakage or loss of competitiveness resulting in carbon-intensive production shifting to markets with less stringent environmental policies. Other reasons for free allocation include providing compensation for the devaluation of existing assets, addressing distributional concerns among adversely affected stakeholders, and broadly easing the transition to an ETS.

Figure 1: Sources of carbon costs for industrial facilities and the role of free allocation



Source: based on (California Air Resources Board [CARB], 2017). Contractual arrangements or market structure may limit cost pass-through for heat and electricity sold in some jurisdictions.

Free allocation to industrial installations is calculated by multiplying the relevant benchmark with production data for the underlying activity. Benchmarks are measured in t/$CO2_e$ relative to a unit of production (metric tons), or predefined input sources used in the production process. They are a key determinant of the overall free allocation that an installation receives. Discount factors may furthermore apply to bring free allocation in line with the overall cap trajectory or differentiate allocation based on a producer's leakage risk. Taken as a whole, the free allocation formula takes the following form:

$$Free\ allocation = Benchmark * \ Production\ data * Discount\ factors$$

Key elements of this equation are discussed in the sections that follow. While all three components impact producers' average cost of carbon, and hence abatement decisions, benchmarks are particularly relevant. For one, choices on benchmark design may impact the range of low-carbon investment opportunities promoted under a given carbon price. As jurisdictions shift to more targeted free allocation approaches to align with long-term mitigation objectives, the weight of benchmarks in determining overall free allocation levels may also increase, rendering them increasingly important for abatement decisions at the firm level.

Setting the right benchmark is a complex task that requires weighing multiple considerations, including determining the scope for abatement, ensuring a credible investment signal, addressing leakage and economic competitiveness concerns, mitigating internal market distortions, navigating political sensitivities surrounding expected increases in production costs, as well as the technical feasibility and administrative capacity to adopt data-intensive design provisions.

The design of benchmarks inherently reflects choices on the tradeoffs between such objectives (which may differ per sector), as well as the pace of emissions reductions already achieved, and the jurisdiction's broader climate goals. In principle, the more broadly benchmarks are defined, the greater the range of abatement options they help unlock. Higher benchmark differentiation, i.e., the use of multiple sub-benchmarks for a single product, increases administrative complexity and reduce abatement incentives but may address industry concerns. The balance struck in benchmarks hence requires reevaluation as the ETS matures, as climate frameworks are revised, or as new low-carbon technologies reach market maturity.

2.1. Choosing the type of benchmark and comparable activities

Greenhouse gas (GHG) benchmarks offer metrics that facilitate comparing the emissions performance of similar industrial activities. GHG benchmarks used in ETSs can be grouped broadly in two categories.[1] **Product-based benchmarks (PBBs)** are a function of the amount of GHG emissions released per unit of industrial product. **Energy-based benchmarks (EBBs)** reflect how much GHGs are emitted from combustion energy that is used at a facility. Unlike PBBs, which are expressed in terms of output, EBBs are expressed as inputs to the production process and are mostly used as a fallback option targeting one (albeit significant) segment of the installation's emissions profile. Variations within each exist and are elaborated upon in more detail below.

PBBs reward early action and provide the strongest incentive for mitigation by setting a uniform efficiency benchmark for a variety of production methods and technologies used to produce the same product. For this reason, they are generally the preferred approach. The European Union (EU), California, New Zealand, and Korea have established 52, 88, 44, and 18 PBBs respectively (see Annex Table 3). These are set at the activity or product level and are reported in the legislation or regulation. In small sectors with very few installations concerned, jurisdictions have also set PBBs for individual facilities.[2]

In developing PBBs, two key principles have emerged. First, California, the EU, and New Zealand have followed the **one-product one-benchmark approach**, which implies PBBs should not be differentiated based on technology, fuel mix, size and age of facility, climatic circumstances, or raw materials. This ensures that the full range of abatement opportunities is incentivized, including switching between fuels, technologies, and input materials. In practice, however, jurisdictions at times differentiate the same or similar products by setting multiple benchmarks to account for divergences in the production process. The other approach involves **setting the benchmark based on data that is representative of normal operation years and excludes outliers** (see 3.1). Typically, an average time span of two to four years closest to the introduction of the benchmark is used as the reference period. In most cases, policymakers have qualitatively assessed whether these years were representative. Where this was in doubt, jurisdictions have allowed for adjustments or for an alternative reference period to be used.

2.1.1. Defining product-based benchmarks

In developing PBBs, industrial activities must first be grouped and identified for coverage by a shared product definition and emissions standard. The product(s) to be covered by a single benchmark can either be homogeneous in nature or close substitutes that differ slightly in core characteristics but share the same applicability (PMR, 2017). PBBs set a common emissions intensity standard for the defined output ($tCO_{2(e)}$/t product) and may cover a range of entities with differing production methods

[1] The third less frequently applied type is process benchmarks, which cover emissions from chemical processes.
[2] In the case of Quebec, PBBs established for individual facilities are not reported in the legislation for confidentiality reasons. In California, CARB receives approval from a facility prior to publishing a benchmark based solely on its data.

and inputs. They increase a producer's marginal carbon cost in proportion to its emissions intensity above the benchmark value multiplied by the allowance price.[3] In this vein, a producer with an emissions intensity of 0.75tCO_2/t covered by a product benchmark of 0.5 tCO_2/t would incur a marginal carbon cost of 0.25 times the allowance price for each additional tonne of product produced. Crucially, PBBs do not discriminate between *inputs* to the production process (i.e., fuel, materials, technologies, and plant characteristics) of the defined output. This ensures that they unlock the full range of abatement options within the installation's boundary given a credible allowance price. In sectors dominated by one production technology, applying a uniform product benchmark is relatively straightforward. However, where inputs and technology deployment in a sector are more diverse, jurisdictions have at times deviated from this approach. For example, a single product benchmark is used for cement production in California, reflecting the sector's dominant production capacity in Portland (i.e., grey) cement.[4] In contrast, differentiated benchmarks for cement production (grey and white clinker) have applied in the EU. The same goes for steel and glass where different production techniques are used. The extent to which benchmarks can be differentiated depending on product type and jurisdiction is also shown in Table 1.

Differentiated product benchmarks tie free allocation to one or more components of the production process and are hence narrower in scope. They often apply to trade-exposed sectors considered to be at risk of leakage, granting them higher shares of free allocation. This limits the need for internalizing carbon costs in product prices and thereby mitigates the risk of reduced competitiveness. Jurisdictions also use differentiated benchmarks to address concerns regarding the fairness of the instrument (e.g., aimed at participants with high investment costs in a particular technology), or to align with broader political considerations. While each of these criteria is often deemed important to ensure a smooth transition pathway under the ETS, differentiating benchmarks in line with them affects the range of abatement incentives and the system's neutrality in promoting low-carbon technologies. It can result in subsidies for carbon-intensive producers receiving the more generous benchmark (Sandbag, 2016).

Table 1: Benchmarks across ETSs

Benchmark (t CO2e/t product)		**EU ETS (Phase 4)**	**K-ETS (Phase 3)**	**California (/short ton)**	**Quebec**	**NZ ETS**
Product	Iron/Steel	**EAF**: 0.215[5]	**EAF**: 0.31824	**EAF**: 0.170		Molten iron: 3.2613
		Blast furnace: 1.288[6]	Hot metal: 0.4287			Cast carbon steel slab: 0.1190
	Coke	0.217	0.8703	Calcined coke: 0.632		Vanadian-bearing materials: 0.280

[3] This assumes output-based allocation whereby a producer receives free allowances at the benchmark level for each additional unit of output. The marginal cost of carbon may be higher if free allocation is based on historical activity levels.
[4] The benchmark for cement manufacturing in the California is 0.741 allowances per short ton of product.
[5] Updated from 0.283 allowances per tonne of product in Phase 3.
[6] Excluding on-site production of carbon-intensive inputs. Updated from 1.328 allowances per tonne of product in Phase 3.

						Flat products of hot-rolled carbon steel: 0.163
						Cast carbon steel billet: 0.1493
						Long products of hot-rolled carbon steel: 0.147
	Sintered ore	0.157	0.2789[7]	-		
	Cement	**Grey**: 0.693[8]	**Grey**: 0.8000[9]	0.742[10]	0.7767[11]	**Grey**: 0.0234
		White: 0.957				**White**: 0.9615
	Glass	**Float**: 0.399	-	**Flat**: 0.495		0.5946
		Colorless: 0.298				
		Colored: 0.237		Container: 0.270		
		Continuous filament glass fiber: 0.309				

Source: based on (CARB, 2019); (Kwon & Ritchie, 2021b); (Climate Change Response Act, 2002; Government of Quebec, 2021; European Commission, 2021).

In principle, the more granular benchmarks become, the narrower the range of abatement options they promote. Benchmarks are therefore ideally not differentiated within an industrial activity. Since PBBs apply to the same or very similar products, tying them to specific fuels, processes, or technologies distorts incentives to adopt the most cost-efficient means of achieving emissions reductions. Applying a technology-specific benchmark to the allocation of free allowances, for instance, would amount to a subsidy for that particular technology relative to a more stringent uniform benchmark or full auctioning. This would effectively distort and undermine price signals to invest in more cost-effective, lower-carbon options (Nelis et al., 2009). Being defined on technology and inputs, differentiated PBBs deliver abatement incentives within the production process of the defined product subcategory. However, they reduce the economic rationale for switching to low-carbon production technologies that might entail coverage by a lower benchmark or none at all.

Differentiated PBBs risk excluding clean technologies (e.g., hydrogen-based primary steel) and low-carbon substitute materials (e.g., non-clinker-based cement) from coverage in the system. This foregoes the competitive advantage of innovative technologies' high carbon efficiency being reflected

[7] Excluding sintered ore that reenters the benchmark boundary.
[8] Updated from 0.766 allowances per tonne of product for grey cement, and 0.987 for white cement in Phase 3. These benchmark values would be updated once more for the second half of Phase 4 of the ETS (2021-30) but are currently undergoing a broader revision in view of updated climate targets and proposed reforms to the EU ETS.
[9] This value covers total GHG emissions, whereas earlier phases excluded process emissions.
[10] Applies to GHG emissions for the production of clinker and the mineral additives added to the clinker produced.
[11] Applies to GHG emissions for the production of clinker and the mineral additives added to the clinker produced. This value is for 2022; it decreases to 0.7721 in 2023.

in investment planning, thus delaying their market readiness. A uniform product benchmark can help mitigate market entry barriers by rewarding low-carbon producers, regardless of technology, with (surplus) allowances to the same extent that it subsidizes carbon-intensive competitors. A level playing field is thus ensured.[12] A timely shift to uniform product benchmarks then becomes a crucial element in ensuring the ETS is effective at increasing the rate of market uptake of clean technologies with high emissions mitigation potential. In line with these considerations, the European Commission has proposed to revise and broaden benchmark definitions removing explicit references to inputs or components of the production process.[13]

Despite the advantages of uniform PBBs, differentiated benchmarks may be more appropriate in certain cases. Their use should be carefully assessed against the mitigation objectives the ETS is intended to support and follow clearly defined principles aimed at leveling the playing field and safeguarding long-term investment signals. First, benchmarks must be defined such that they cover the scope of the entity's production process and be based on clear, uniform emissions boundaries (PMR, 2017). In sectors where emission boundaries are less clear, and variations in production activities and associated scope 1 (direct) and scope 2 (indirect) emissions are present among entities producing the same product, multiple benchmarks might be required. This usually reflects differences in production scope within a particular sector and does not inevitably necessitate benchmark differentiation along technology, fuel, or input material used. Taking the example of primary steel production, scope 1 emissions are greater for integrated plants producing coke on-site —a key input material— compared to standalone steel producers purchasing it. Therefore, jurisdictions tend to use separate benchmarks for intermediate products (e.g., coke) targeting direct emissions along the value chain. This approach does not account for potential indirect emissions savings downstream, which is discussed further in 2.2 (Zipperer et al., 2017).

A second principle centers on benchmarked products needing to be of the same (range of) product quality. Variation in product characteristics and applicability can be a criterion for differentiating relatively similar products into separate benchmarks to reflect differences in value added or market segmentation. This principle is often applicable to heterogeneous sectors with product variations that may render a single product benchmark less feasible. Colored and colorless glass, for example, have divergent product features and serve different consumer groups. When substitution between such products is not practical or deemed desirable, differentiated benchmarks can help reduce costs under the ETS while still promoting efficiency improvements within the product sub-category. However, counter to the logic of the ETS, applying differentiated benchmarks does entail a risk of dampening abatement incentives for the more carbon-intensive producer. The emissions intensities of the production of colored and colorless glass can vary by up to 90% due to the use of different input sources (PMR, 2017). Applying separate benchmarks would hence curb the potential for switching to the lower carbon product (in this case, colored glass) in applications where substitution is possible.

[12] The risk of exclusion from the ETS disappears under full auctioning, which by fully pricing emissions is technology neutral by design, but exposure to international competition can be a limiting factor in ensuring a level playing field for potential breakthrough technologies.

[13] Also under consideration is a provision that would keep installations in the ETS when abatement efforts lead their rated capacity to drop below the system's inclusion threshold. See pages 500–504 in (European Commission, 2021) for an overview of the proposed changes.

Beyond production scope and product quality criteria, jurisdictions have opted for differentiated benchmarks in cases where low-carbon alternatives are limited or not scalable to meet aggregate demand. In this vein, separate benchmarks typically apply to blast furnace (BF) (primary) steel and electric arc furnace (EAF) (secondary) steel in jurisdictions where both production routes are in use. While presenting a viable low-carbon production technique, EAF steel mostly relies on scrap metal as a basic input source and therefore faces limits in the extent to which it can serve overall steel demand. The decision, then, to grant a separate benchmark to BF (primary) steel producers often follows from strategic economic considerations to maintain security of supply in critical construction materials making leakage protection measures a necessity. In line with the product quality principle above, variance in the level of purity between primary and secondary steel has also featured in decisions to use differentiated benchmarks for the sector. However, as new low-carbon production routes emerge that may not have been available at the time benchmarks were first established (Ito et al., 2020), adopting a uniform benchmark should be prioritized.

It follows that uniform product benchmarks are the preferred choice for steering energy-intensive industries on a low-carbon trajectory, but there are limits to their effectiveness. While a PBB does incentivize the full range of abatement options in the production process of the benchmarked product, it will not spur a shift to lower-carbon products that receive different treatment (e.g., a lower benchmark) under the ETS (Flues & van Dender, 2017). Such market distortions are difficult to address when products are differentiated along clear functionality and quality criteria but when their substitutability is not precluded—such as may be the case for construction materials like steel, cement, aluminum, and wood, but also increasingly fiberglass and other plastics. Over the long term, increasing the share of auctioning relative to benchmarked allocation can address these limitations.

2.1.2. Energy-based benchmarks

All systems covered in this note, with the exception of New Zealand, also use EBBs (either heat- or fuel-based benchmarks) as a fallback option. Fuel benchmarks set an emissions intensity standard for combustion energy used as input to the production process (e.g., in tCO_2/TJ). They are often based on a specific reference fuel (such as natural gas) but can also reflect the average emissions intensity of the overall fuel mix within a sector. A heat benchmark goes one step further in targeting heat as an intermediate product. In similar fashion to the fuel benchmark, heat benchmarks are based on the emissions factor relative to the net caloric value of the (reference) fuel but are subsequently multiplied by a given or desired conversion efficiency of the boiler. For this reason, they are often the preferred fallback option for sectors and sub-installations that use heat in the production process. Free allocation under EBBs is calculated as follows:

$$Free\ allocation = Energy\ benchmark * Energy\ consumption\ data * Discount\ factors$$

EBBs differ from fuel-technology benchmarks, which in certain jurisdictions (e.g., Korea and China) apply to electricity generation and, defined as CO_2/MWh, can be grouped as differentiated product benchmarks.[14] EBBs are narrower in scope compared to product benchmarks in that they target only the first segment(s) of the production process, and where possible are replaced by uniform PBBs as data and experience with the ETS evolve.

EBBs present appropriate alternatives for industrial activities where developing PBBs is challenging. This can occur in sectors with too few installations and limited production data for PBBs to be effective. Sub-sectors of the chemical industry, for instance, can be highly concentrated even in large systems as the EU ETS, and would in effect set their own benchmark if PBBs were to apply – limiting incentives for abatement. In such cases, product benchmarks could be based on a reference technology (best available technique, or BAT) not yet deployed in the jurisdiction.[15] However, data constraints may limit the feasibility of this approach. EBBs avoid the issue by tying free allocation to a reference fuel or standard heat conversion process that is not unique to the sub-sector. Furthermore, PBBs can be complex to implement in sectors with heterogeneous products where the administrative cost of developing a multitude of product benchmarks may outweigh their share of emissions (e.g., the food processing industry). In such cases, EBBs provide a more efficient alternative targeting the common segments in otherwise differing industrial activities. Since the carbon content of fuels is known ex ante, energy benchmarks are also appropriate when historical emissions intensity data is not available. They generally are more effective in industrial activities where fuel or heat consumption accounts for most emissions (e.g., paper production). Finally, jurisdictions may opt for EBBs to provide transitionary assistance to producers using emission-intensive fuels, or to promote fuel-specific efficiency increases in certain technologies of choice.

When applied uniformly to a sector or product, EBBs incentivize usage of the full range of energy sources with carbon intensities below that of the reference fuel. In addition to promoting a switch to low-carbon fuels, a heat benchmark encourages improvements in the conversion efficiency of the energy input source (e.g., into a heat carrier like steam). This is because they target the energy conversion process, tying free allocation to the intermediate product instead of the fuel input. The energy consumption data used to determine free allocation is relevant for firm-level abatement decisions under either benchmark. Under a fixed-historical approach, free allocation does not adjust on par with energy consumption levels until a new baseline period is set. Consequently, producers may face an incentive to increase energy efficiency, besides shifting to lower carbon fuels, in order to lower potential allowance shortfalls. This incentive weakens with more frequent updating of energy consumption data and disappears under annual updating, which in some certain sectors might be necessary for adequate leakage protection.

EBBs target inputs (energy) rather than outputs (products) and mainly impact decisions on the fuel mix used. Since they do not target the full scope of production activity within the installation boundary,

[14] In the case of the Chinese national ETS, benchmarks are differentiated between conventional coal-fired generators with capacity above 300 MW (0.877 tCO_2/MWh), conventional coal-fired generators with capacity below 300 MW (0.979 tCO_2/MWh), unconventional coal-fired generators (1.146 tCO_2/MWh), and gas-fired generators (0.392 tCO_2/MWh). In the K-ETS, electricity benchmarks are differentiated to one benchmark per fuel-technology source based on weighted-average emission intensities.

[15] BATs are advanced and proven approaches to prevent and control industrial emissions and their environmental implications. These techniques are developed at a scale that allows them to be implemented under economically and technically viable conditions OECD (2020).

they fall short of incentivizing the complete range of abatement options in the production process. Process emissions[16] can comprise significant shares of an installation's emissions profile (e.g., in cement production) and must be covered separately in an energy benchmark approach.

2.2. Setting the scope of the benchmark

The **scope or "boundary" of the benchmark**—i.e., the elements and activities leading to the production of the output—determine which emissions are included in the benchmark. Decisions on benchmark scope impact the breadth of abatement incentives under the ETS. Industrial production can involve direct emissions from on-site fuel combustion, chemical transformations during the production process, and indirect emissions from purchased carbon-intensive inputs (e.g., electricity, clinker, and coke). Whether direct or indirect, the activities covered should only be those within the control or responsibility of the covered entity, and the boundaries should target the most common and carbon-intensive activities (PMR, 2017).

Jurisdictions have taken different approaches with respect to benchmark scope. Within the EU ETS, free allocation is based solely on the direct emissions component although indirect emissions are included in the scope of the benchmark for products where direct and indirect emissions from electricity are interchangeable, fuel and electricity are interchangeable inputs, or imported heat or waste gases are used. This concerns 14 products.[17] In these cases, indirect emissions are deducted using standard emissions factors (EC, 2018). Québec excludes indirect emissions from its ETS benchmarks. In California, allowance allocation to industrial entities accounts for on-site covered emissions and the emissions associated with purchased electricity and steam but excludes the emissions associated with sold electricity and steam, where emission costs are passed on to the purchasing entities.[18;19] In Korea, benchmarks encompass scope 1 emissions, but separate benchmarks have applied to large consumers' indirect emissions of purchased electricity in the absence of cost pass-through conditions in the regulated electricity market (Kuneman et al., 2021). New Zealand has included indirect emissions from electricity consumption, coal seam methane gas, and fuel oil in product benchmarks since 2012 to compensate industrial consumers for price increases under the ETS (Rontard & Leining, 2021).

Jurisdictions often limit the scope of product benchmarks to direct emissions to encourage efficiency improvements at the source of combustion or production and avoid additional data complexity. To that end, the emissions associated with producing intermediate products are usually covered by individual

[16] Emissions stemming from chemical or physical transformation of materials during production.
[17] These are: refinery products, EAF carbon steel, EAF high alloy steel, iron casting, mineral wool, plasterboard, carbon black, ammonia, steam cracking, aromatics, styrene, hydrogen, syn gas, and Ethylene oxide/ethylene glycols.
[18] California did not calculate initial benchmarks to include the emissions associated with purchased electricity because it was not clear how electrical distribution utilities would set industrial electricity rates under the Cap-and-Trade Program. Since construction of the benchmarks in 2010-2011, it has become clear that carbon costs will be passed to all ratepayers, including industrial entities. To account for this, the California Air Resources Board (CARB) has proposed to calculate new energy- and product-based benchmarks that include purchased electricity for the post-2021 period. These changes would be part of a new regulatory package.
[19] The process to account for the indirect emissions associated with electricity purchases by industrial facilities is separate from the allocation process for direct on-site emissions and emissions associated with purchased steam. Free allocation is provided to electric utilities, and electric utilities pass along a portion of that free allocation value to their industrial customers based on output-based allocation using indirect electricity emissions benchmarks that mirror the direct emissions benchmarks.

product benchmarks distinct from subsequent production activities. For example, coke (intermediate product), sinter ore (intermediate product), and steel (final product) each have their own benchmark in most ETSs. The use of multiple product benchmarks promotes abatement along each segment of the value chain. It also ensures that integrated installations and standalone plants—which source carbon-intensive input materials from offsite facilities—receive a differing number of free allowances per unit of final product that is proportional to their distinct production activities. In this way, a level playing field is ensured.

A shortcoming of this approach appears when carbon costs cannot be freely passed on to downstream producers and consumers – because of international competition. In instances where carbon costs from indirect emissions have *not* been internalized in product prices and lie outside the scope of the benchmark, firms may be incentivized to focus only on the mitigation of direct emissions to reduce their carbon costs rather than emissions abatement along the value chain. Building on the example of steel production, an integrated steel mill will be incentivized to increase its efficiency of coke production as to shorten its allowance shortfall or turn it into a surplus. This is contrary to a standalone plant whose free allocation under the benchmark is not affected by the indirect emissions of coke sourced off-site. However, neither installation faces an incentive to shift to low-carbon substitute materials: the standalone producer is not confronted with a carbon price on its purchased inputs, and the integrated plant may risk losing out from free allowances tied to coke production when shifting to an alternative. Casting wide the scope of the benchmark adjusted for emissions scope can avoid such distortions while retaining a level playing field among producers with different system boundaries. Allowance allocation could instead be based on one uniform product benchmark tied to the final product (i.e., steel) and reduced in proportion to the emissions of carbon-intensive inputs procured off-site and used in the production process (Zipperer et al., 2017). In this vein, low-carbon input or intermediate product substitution does not affect free allocation received under the benchmark, but the use of outsourced carbon-intensive inputs does. Indirect emissions are hence incorporated into the scope of the benchmark to approximate mitigation incentives otherwise set in motion under carbon cost pass-through conditions, albeit based on subsidies (additional allowances) rather than a reduction of indirect carbon costs.[20]

The scope adjustment of benchmarks can be taken one step further to account for the effect of byproducts on emissions savings along the value chain or in other sectors (Zipperer et al., 2017). Byproducts of energy-intensive production processes in the form of heat, electricity, or input materials for other industries (e.g., slag, a byproduct of BF steel used for low-carbon cement) can be used on-site or sold and yield emissions savings elsewhere. However, boosting their output may increase direct emissions and hence the marginal cost of primary production under the ETS. Cross-sector and downstream emissions savings potential can be considered by extending the scope of the main product benchmark in line with net emissions savings relative to the displaced carbon-intensive product so long as the latter is covered by free allocation provisions (Zipperer et al., 2017). The producer would thereby receive additional allowances under the benchmark intended to level the playing field of low-carbon

[20] Where considered feasible, the internalization of carbon costs by means of a carbon border adjustment mechanism and reduced free allocation may deliver stronger incentives for abatement.

byproducts with those it ought to replace. The benefits of such an approach must be weighed against additional data requirements and complexity.

Table 2: Abatement incentives per benchmark type

Benchmark type / abatement incentive		Fuel/technology / input switch	Fuel-combustion efficiency	Process efficiency	Substitution of inputs
Product	Scope-adjusted, PBB	Yes	Yes	Yes	Yes
	PBB	Yes	Yes	Yes	No
	Differentiated, PBB	No	Yes	Yes	No
Energy	Heat-based benchmark	Yes	Yes	No	No
	Fuel-based benchmark	Yes	Partly (incentives increase with less frequent updating of activity levels)	No	No

Source: authors' own elaboration

Table 2 summarizes the abatement incentives producers face for each benchmark type. Given their reduced scope for emission reductions, EBBs have mostly been used as fallback option or for specific objectives such as those listed above. Since PBBs strike a balance between broad emission reduction incentives and the need for addressing economic considerations, they have been the benchmark type of choice for most jurisdictions. However, abatement incentives narrow the more differentiated PBBs become.

2.3. Setting the benchmark stringency and updating provisions

Together with production levels compared to baseline activity (see 3.1) and discount factors applied (see 3.2), benchmark stringency determines the proportion of free allowances that entities receive relative to their emissions and how many of them are required to purchase additional allowances or invest in emissions abatement opportunities.[21] The stringency of benchmarks directly affect firm-level decisions through their impact on the effective (or average) cost of carbon internalized in planning decisions. Generally, firms internalize the marginal cost of emissions above the benchmark. Therefore, the more stringent (i.e., lower) the benchmarks are set, the greater the share of emissions subject to the allowance price, and the greater the impact of allowance prices on production costs. Under uniform benchmarks with increasing stringency levels and all else being equal, the average carbon cost (per unit

[21] Installations that perform less efficiently than the benchmark face a shortage and therefore need to increase efficiency in-house or purchase additional allowances. Facilities that perform more efficiently than the benchmark receive more allowances than they need and can sell the excess on the secondary market. Where the free allocation budget declines progressively, as observed in the EU ETS and in California, efficient producers performing below the benchmark might still face a shortfall of allowances provided by free allocation such that the producers may need to acquire additional allowances at auctions and secondary markets. Efficient producers with increasing output levels may also face a shortfall in systems that opt for fixed baseline periods of activity with minimal updating.

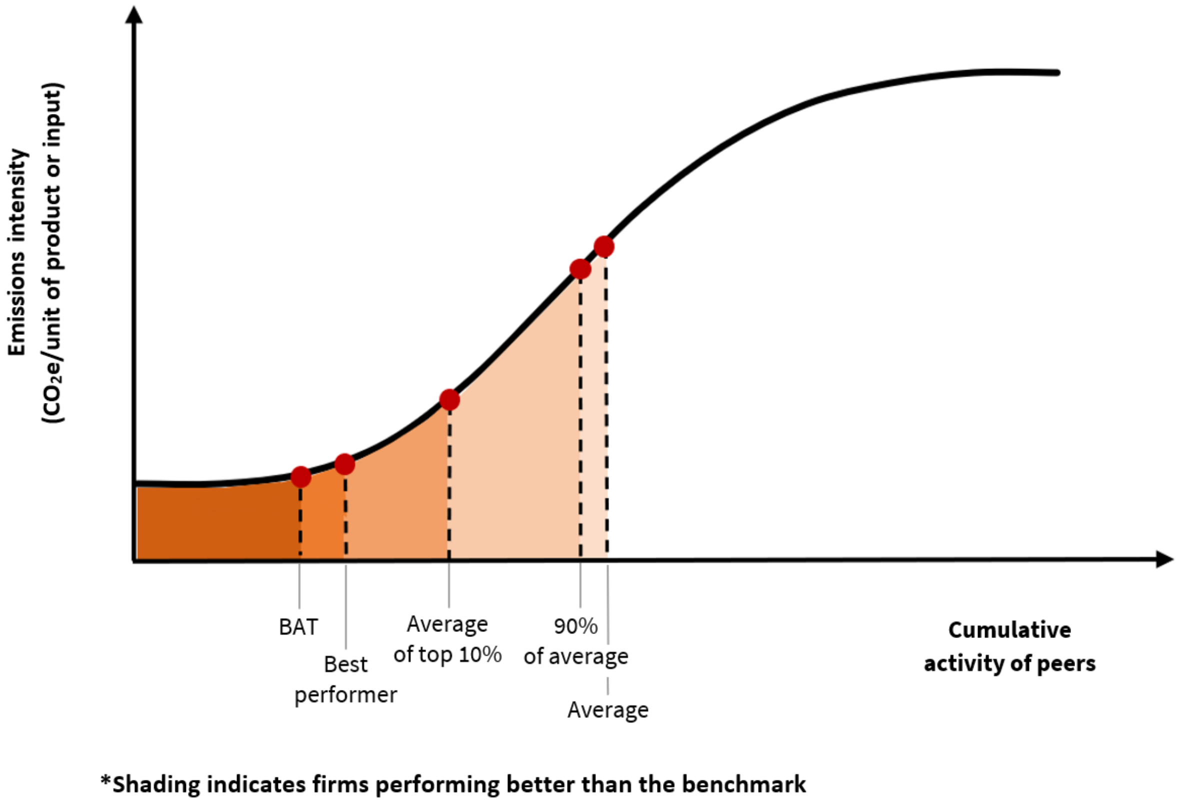
of total emissions) for carbon-intensive producers increases up to the market price of allowances prevailing in the ETS (Flues & van Dender, 2017). How stringent benchmarks are set thus directly plays into the profitability of the available production technologies and in what direction investments are steered.

The impact of benchmarks on effective carbon costs will differ by sector and system. In sectors that receive 100% free allocation at the benchmark or similarly high levels, the benchmark stringency level will be the key determinant of allowance costs internalized in the production process. This often applies to EITE sectors, but also to certain non-EITE sectors covered by output-based allocation and where discount factors play less of a role. In practice, all cap-and-trade systems to varying extent use discount factors to align the free allocation budget with leakage criteria or the cap's trajectory (see 3.2). In the EU, industries not included in the carbon leakage list received 30% free allocation at the benchmark level in 2020, to be phased out completely by 2030. In such cases, benchmark stringency remains important but is not the single driver in determining a firm's average carbon cost.

Across the systems surveyed, benchmark stringency has been determined by taking account of the existing emitters' performance. In New Zealand and Québec, benchmark stringency reflects the average emissions intensity of production facilities. In Korea, the weighted average emissions intensity of producers belonging to the same sub-sector is used, but this may change to the BAT in the fourth trading phase that will commence in 2026 (Kwon & Ritchie, 2021a). In California and the EU, benchmarks are set below the average, reflecting the emissions intensities of highly efficient facilities. Under the EU ETS, benchmark values are based on the average emissions intensity of the top 10% most efficient installations (Article 10a.2, EU Directive 2018/410, 2008/2018). In California, benchmarks are set at 90% of the average emissions intensity of a sector (see Figure 2). How sectors or products are defined furthermore affects the stringency of benchmarks by presetting the range of activity data (i.e., carbon intensities) they are benchmarked against (see 2.1). Differentiated PBBs tend to result in more generous benchmark values for carbon-intensive outputs.

Choices surrounding benchmark stringency should consider the long-term impact on low-carbon investment. Where emissions abatement on aggregate keeps pace with the emissions cap and reductions to the allowance budget, benchmark values inform decisions at the firm level on whether to abate now or later by sending a signal on the expected returns on carbon-intensive versus clean technologies. Generous benchmarks can be deployed as a transitionary tool, but they may entail a risk of distorting long-term investment signals. In principle, the strength of the allowance price sets the incentive for low-carbon investment irrespective of benchmark stringency. Where generous benchmarks provide an additional revenue stream for clean technologies (i.e., through selling surplus allowances), stringent benchmarks increase the carbon costs for carbon-intensive production. Both support the profitability of low-carbon vis-à-vis carbon-intensive technologies. Generous benchmarks might even be more effective in promoting investment in clean technologies where it concerns products that compete in international markets. However, they reduce direct cost exposure under the ETS, which can dampen incentives to disinvest in polluting technologies featuring long lifecycles and fixed capital that must be recouped. A clear signal must be sent to industries on the magnitude of adjustments required in the short- and long-term. Setting sufficiently stringent benchmarks from the outset or clearly communicating their gradual tightening aligned with the desired cap trajectory can help avoid locked-in carbon-intensive infrastructure and increased future abatement costs.

Figure 2: Illustrative benchmark curve showing various benchmark stringencies



Source: authors' own elaboration based on (PMR, 2017)

Benchmarks can be fixed (no updating over time) or dynamic (updated at defined triggers, regular intervals, or during broader system performance evaluations) (PMR, 2017). These updates may follow changes in the underlying activity data, for example once the sector at large has significantly reduced the emissions intensity of production or when new technologies have entered the market and the benchmark value no longer reflects the sector's overall performance or the opportunities available for low-carbon investment. Where this is the case, jurisdictions may update benchmark values at the given stringency level or increase the stringency of benchmarks altogether. Doing so enables targeted free allocation at the benchmark level and could avoid triggering discount factors under a declining cap and allowance budget.

Approaches to updating benchmarks vary across the jurisdictions surveyed. In New Zealand, Korea, and the Western Climate Initiative (WCI) jurisdictions of California and Québec, benchmarks are updated on an as-needed basis. To date, California has administratively updated benchmarks: (1) when normal operating conditions in a sector have substantially changed; (2) when the make-up of a sector has substantially changed due to facilities entering or exiting the program; or (3) when CARB has needed to consolidate products to streamline reporting and allocation. However, CARB does not have a mandate to re-evaluate or adjust benchmarks to increase their stringency, and changing benchmarks requires a formal public regulatory process.[22] Québec follows a similar administrative approach for updating benchmarks.

In contrast, for Phase 4 of the EU ETS, the EU has adopted a regular updating mechanism for benchmarks to reflect technological developments. Benchmarks values (i.e., the average top 10% best performers) are updated at five-year intervals based on recent activity data according to which an annual reduction rate between 0.2% and 1.6% is applied to each product benchmark for the next five years. If the annual update rate corresponding to the updated benchmark value falls outside the 0.2-1.6% range, the relevant limit value applies. The Commission has proposed to increase the upper threshold to 2.5% from 2026 along with a revision to the scope and definitions of benchmarks (see 2.1.1).

New Zealand has thus far not updated the benchmarks under its ETS, but may in the coming years to avoid overallocation (Rontard & Leining, 2021). A key advantage of keeping benchmark values constant is that it affords entities a predictable subsidy on low-carbon investment, thereby supporting early action – an advantage that diminishes as benchmarks become more stringent over time. Therefore, the key trade-off to maintaining stringent benchmarks through their regular updating in line with mitigation objectives is the diminished opportunity for firms to mitigate abatement costs through revenues obtained from allowance trade under the ETS.[23] This trade-off can be managed through policy predictability on benchmark design, which enables firms covered under any updating approach to derive credible assumptions about their expected future carbon costs and corresponding investment options.

3 Other factors in determining allocation

3.1. The use of production or activity data

In determining the number of free allowances allocated to each entity, the respective benchmarks must be applied to underlying activity levels. Free allocation can either reflect actual production (**output-based allocation (OBA)**) or be based on a historical baseline period of production (**fixed historical benchmarked allocation**). As it has been practiced, however, jurisdictions using fixed historical benchmarked allocation have allowed for some annual adjustments based on actual production. This is the case in Phase 4 of the EU ETS, which is further explored below. The key distinction is that OBA typically fully accounts for annual changes in levels of production, e.g., without a threshold of changes in activity levels that triggers an increase or decrease in allocation.

OBA is applied in California, Québec, and New Zealand. Under OBA, changes in production are fully compensated by changes in the level of free allocation; hence, OBA provides very strong leakage protection (Meunier et al., 2014). OBA may trigger a behavioral response in firms whereby they treat benchmarks as a focal point for improvements in efficiency (Branger & Sato, 2017). The carbon costs a firm faces stand in direct relation to how efficiently it performs relative to the benchmark and not in

[22] Most of the 88 product benchmarks in California have also remained fixed for the duration of the Program with only a relatively small number updated due to shifts in the operating conditions of the facilities within the sector.
[23] Leakage protection is the other important trade-off not further discussed in this paper.

relation to decisions about overall output. This effect is somewhat diminished under systems that are less responsive to changes in production.

Under OBA, in addition to discount or correction factors (see 3.2), the stringency of the benchmarks also plays a role in ensuring environmental integrity by ensuring that allocation declines in step with the cap. This is because the pool of free allocation can grow more easily in line with industrial output, which in some jurisdictions could pose challenges with exceeding the cap in the absence of measures to control overall levels of allocation. This challenge will likely be greater for jurisdictions with narrower sectoral scope and thus a smaller pool of allowances from which to draw (Acworth et al., 2020). Benchmarks must therefore be stringent enough to achieve climate targets, and discount factors are likely required to ensure overall allocation is in line with the cap trajectory. In addition, since OBA reduces carbon costs for each additional unit of output, it limits increases in product prices stemming from carbon costs, which in turn reduces incentives for downstream abatement (Zipperer et al., 2017). Under OBA, the opportunity cost—which could be passed on at least in part to consumers—is the difference between a firm's emissions intensity and the benchmark, as opposed to the full amount of emissions in a system that does not respond to changes in actual production (Branger & Sato, 2017).

Fixed historical benchmarked allocation was applied in Phase 3 (2013-2020) of the EU ETS, where allocation was based on a PBB and historical activity levels, taking the mean value of annual output during the baseline years of activity.[24] Yearly changes to allocation in Phase 3 were limited to drastic reductions in output, of at least 50% below baseline activity levels. During Phase 4 (2021-2030), the approach to fixed historical benchmarked allocation is more responsive to changes in production, including year-on-year increases or decreases in allocation resulting from changes of more than 15% in activity levels measured based on the average of the previous two years. For Phase 4, the baseline activity levels will also be updated twice to account for changes in production. With more frequent and responsive updating of allocation based on actual production, the EU is moving closer to OBA. The K-ETS also applies fixed historical benchmarked allocation to an expanding number of sectors as grandparenting is phased out.

In setting a baseline for fixed historical benchmarked allocation, using data from a period spanning a predetermined number of years (typically two to three) close to when the ETS was introduced can strike the balance between effective carbon leakage production and availability of data (PMR, 2017). Since under a fixed historical benchmarked approach allocation does not fully account for changes in production, firms might decide to reduce output to reduce emission liabilities and, where possible, increase prices. This may induce demand-side abatement but may come at the cost of windfall profits for industrial producers (PMR & ICAP, 2021). If facilities were to limit output due to incentives under fixed historical benchmarked allocation, deadweight losses may also occur from resulting mismatches between supply and demand. At the same time, a weaker link between allocation and production means that leakage protection may not be particularly well calibrated, with installations receiving more than they need when output falls, and less than they need when output increases.

[24] Operators were free to choose between the years 2005-2008 or 2009-2010.

3.2. The use of discount factors

Discount factors are applied either to differentiate the level of free allocation for specific sectors or facilities based on carbon leakage risk, or to lower the level of free allocation to reflect declining emissions caps (e.g., cap decline factors or the cross sectoral correction factor in the EU ETS).

In determining which sectors are at risk of carbon leakage, ETS jurisdictions to date have used two main indicators, emissions intensity[25] and trade exposure,[26] either in isolation (EU ETS Phase 3) or in combination (New Zealand, EU ETS Phase 4, California, Québec, and Korea). The EU has a binary assessment of carbon leakage, with all industrial activities above the threshold of leakage risk receiving 100% free allocation at the respective benchmark regardless of their degree of emissions intensity and trade exposure. Other jurisdictions have a tiered assessment of carbon leakage risk and apply what is commonly referred to as an assistance factor for different levels of emissions intensity and trade exposure. New Zealand uses two tiers of leakage risk (moderate and high), applying an assistance factor of 60% and 90% respectively to facilities' overall allocation. Québec uses three tiers (low, medium, and high), with assistance factors of 90%, 95%, and 100% respectively for the 2021-2023 period.[27] California has in theory three tiers but in practice applies the same assistance factor of 100% owing to state legislation and ongoing concerns about leakage risk.

Cap decline factors for free allocation are used to bring allowance allocation in line with the general cap trajectory. Cap decline factors are applied in California and Québec. New Zealand also has provisions for cap decline factors, although these have not yet been used. WCI jurisdictions have differentiated the use of cap decline factors across emission sources. In Québec they did not apply to process emissions until 2020, but for the 2021-2023 period, a 0.5% annual decline factor is applied to process emissions. In California, activities with over 50% of total emissions from industrial processes, high emissions intensity, and a high leakage risk classification are subject to a more moderate cap decline factor such that by 2031, allocation will reach about 75% of its level at the beginning of the program.[28]

In the EU ETS, where the share of free allocation ("industry share") is fixed in proportion to the overall cap, a uniform cross-sectoral correction factor (CSCF) is used to ensure that the sum of free allowances is equivalent to the industrial sector cap, although the EU includes pool of allowances representing 3% of the total allowances to act as a buffer to avoid triggering the CSCF. If the bottom-up calculation of free allocation based on the relevant rules for all eligible installations exceeds the "industry share", the CSCF is applied by the same proportion to reduce allocation for all installations that are not electricity generators. The CSCF was applied in Phase 3 of the EU ETS. It not necessary in the first half of Phase 4 (2021-2025), but it may be applied in the latter half of Phase 4, if necessary. The K-ETS also has a correction factor to reduce overall allocation if it exceeds what is available under the cap.

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[25] *Emissions intensity* is a measure of how strongly the carbon price affects a specific sector or firm. It can be measured as the volume of emissions created per unit of output, revenue, value added, or profit. Sectors are classified as emissions intensive if the selected metric rises above a set threshold.

[26] *Trade exposure* reflects sectors' exposure to international competition and is used as a proxy to determine whether carbon costs can be passed on to end consumers. Systems have either done a qualitative assessment of whether international trade takes place (New Zealand) or measured this in terms of intensity (EU ETS, California, Québec, and Korea).

[27] Québec has an additional fourth tier for electricity and steam generators with fixed-price contracts before 2008.

[28] By 2031, the cap decline factor for all other activities will reach about half of its level at the beginning of the program.

Alongside benchmark design, choices on the use of production data and discount factors will impact the overall level of free allocation that entities will receive. While these choices are aimed more squarely at addressing leakage concerns and ensuring the integrity of the cap as allowance budgets decline, they may also affect abatement incentives and interact with benchmark design, since they ultimately impact the average carbon costs of covered entities. Jurisdictions should therefore carefully consider the use of product data and discount factors as part of their free allocation policy along with benchmark design.

4 Concluding remarks and future prospects

Benchmarking is a tool that can effectively protect firms in an ETS from carbon leakage while setting incentives to reduce emissions. It offers ETS jurisdictions the advantage of severing the link between a firm's emissions and the allowances it receives as part of free allocation. While this provides significant advantages over grandparenting based on historical emissions, benchmarking is complex, and the way it is designed has implications for the effective carbon costs and abatement incentives that regulated entities face.

How products are defined and the extent to which similar products are differentiated—and are thereby at risk of violating the principle of "one product, one benchmark"—will affect abatement incentives and administrative complexity. A jurisdiction may have valid reasons for having multiple benchmarks for very similar products, but this could favor more emissions-intensive producers and distort the signal for low-carbon investment. Setting a wide scope or boundary of the emissions included in the benchmark (e.g., sources of indirect emissions) will increase the number of abatement options and may enhance incentives where the costs of the activity have not been internalized. However, this significantly increases complexity and is not widely practiced in ETSs. Lastly, jurisdictions take a variety of approaches on the stringency of benchmarks, which plays a direct role in determining net carbon costs and the profitability of available production technologies, as well as in how to update benchmark values.

Energy-based benchmarks may serve as a suitable fallback option for sectors where activities are hard to group or production data is not yet available, but these are often replaced by PBBs as ETS experience evolves, since they offer a wider set of abatement options across the production process as opposed to only specific segments. For some jurisdictions, EBBs may remain in place in some circumstances, owing to factors such as a limited number of comparable facilities in a sector.

Together with benchmark stringency and discount factors, which production data a jurisdiction uses has strong implications for the effective carbon costs that a producer faces, with choices between recent or actual data and historical baselines that are less responsive to changes in output. Despite their potential cost implications, discount factors are a useful tool to control overall levels of free allocation in line with the cap trajectory or to help target allocation based on entities' different leakage risks.

Benchmarks are well suited to driving production efficiency and, if defined broadly, in levelling the playing field across technologies. Increasing their stringency over time in a predictable manner allows jurisdictions to strengthen incentives for emissions abatement and target free allocation within the constraints of declining allowance budgets. However, there are limits to what free allocation can achieve in terms of the deep industrial decarbonization that is required under net-zero targets. Deeper stages of decarbonization will require large-scale uptake of innovative technologies and processes as well as strong price signals for the consumers of industrial materials. Each of these will eventually rely on greater carbon cost pass-through in product prices (Acworth et al., 2020). More stringent benchmarks can facilitate a gradual shift toward full auctioning, which would be more compatible with the longer-term demands of deep decarbonization in that they provide an undistorted price signal. However, energy-intensive industries may continue to face constraints in their ability to pass on the costs of mitigation in their product prices owing to leakage risks. Jurisdictions may therefore need to consider broader reforms and complementary policies to carbon pricing, including border carbon adjustments and increased financial support for innovation, to put emissions-intensive industries on a trajectory consistent with net zero as they transition from free allocation.

Such policies are already under development in some jurisdictions, especially in the EU. In addition to a carbon border adjustment mechanism, the EU and member states are pursuing additional support for innovation through carbon contracts for difference—essentially a feed-in-tariff for industrial sectors—and placing conditionality on free allocation such that covered entities must invest in energy efficiency measures identified in mandatory audits or equivalent measures to receive full allocation (European Commission, 2021).[29] Benchmarks may be relevant for jurisdictions pursuing border carbon adjustments in terms of setting default values for the embedded emissions of imported goods, given the relative ease of setting default values based on available data. Québec is similarly seeking to increase incentives for investment while boosting funding for industrial sectors in its provincial budget. Under consideration is a policy that would reduce free allocation to industrial emitters by a yearly percentage but reserve the revenue from the sale of these allowances at auction for them in the form of low-carbon investment support.

Meanwhile, carefully designed technology-neutral benchmarks can buttress the long-term investment signals under the ETS while supporting industry so long as allowance budgets permit.

[29] This does not apply to investments with pay-back periods beyond five years. Costs of investment must be proportionate. The reduction in the event of non-compliance with the provision would be 25% of allocation.

5 Annex

Table 3: Overview of free allocation approaches across key jurisdictions

Jurisdiction	Industrial classification	Benchmark design						Benchmark stringency
		Benchmark	No. of PBBs	Alternative benchmark approach	Reference period	No. years in reference period	Specified in legislation or regulation	
EU	NACE (4-digit)	(tCO_2)/t product	52	Energy-based benchmarks, process benchmarks (as fall back)	2016-2017	2	Yes	Average of 10 % of the most efficient installations within a (sub-)sector, based on 2016–2017 activity data. Values adjusted for technological progress on a yearly basis, with annual reduction rate (0.2 % to 1.6 %) determined for each.
California (WCI)	NAICS (6-digit)	(tCO_2)/t product	28 (2012) 88 (2018)	Energy-based benchmarks (as fall back)	2008-2010[30]	3	Yes	90% of average or "best in class"
Québec (WCI)	NAICS (6-digit)	(t CO_2)/t product (weighted average of process emissions, combustion emissions, and other emissions)	76	Energy-based benchmarks (as fall back)	2007-2010[31]	4	Sector level benchmarks: Yes; Facility specific benchmarks: No[32]	Average performance
Korea	Own classification	(t CO_2)/t product and (t CO_2)/t raw material input	18	Energy-based benchmarks (fall back)	2017-2019	3	Yes	Capacity-weighted average
New Zealand	Own classification	Allocative baseline	44	n/a	2006-2009	3	Yes[33]	Average performance

30 If these years are deemed not representative, an alternative reference period may be used.
31 For sites where 2007-2010 data is not available, a minimum of 3 reference years, excluding start-up year, are used.
32 Referred to as "reference units" in the legislation
33 Referred to as "allocation baselines" in the legislation.

Table 3 (continued): Overview of free allocation approaches across key jurisdictions

Jurisdiction	Production / Activity Data	Discount Factors	
	Reference period	Carbon Leakage / Assistance Factors	Cap Adjustment Factors (CAF)
EU	Historical activity level (2005-2008 or 2009-2010 for Phase 3; 2014-2018 and 2019-2023 for Phase 4)	100% of benchmark for sectors on the Carbon Leakage List (CLL); declining from 80% to 30% for sectors not on the CLL throughout Phase 3 (2013-20) and to 0% by 2030.	Linear reduction factor for the cap as a whole (1.74% in Phase 3, 2.2% in Phase 4); reflected in the cross-sectoral correction factor for free allocation sectors. Linear reduction factor subject to change as part of 2030 EU ETS revisions.
California (WCI)	Annual production verified through Mandatory Reporting Program	Three categories: high, medium, and low. Assistance factors for each category determined by leakage risk and were originally envisioned to decline starting in the second period (2015-2017) for all but the high-risk category. Subsequent regulatory decisions and a legislative mandate from Assembly Bill 398 in 2018 resulted in fixing assistance factors for all risk categories at 100% because of ongoing concerns about emissions leakage risk.	Cap adjustment factor (CAF) for "standard activities" that declines annually in proportion to the overall caps; activities with over 50% of total emissions from process emissions, high emissions intensity, and a high leakage risk classification are subject to a more moderate CAF.
Québec (WCI)	Annual production	2013-20: Assistance factors used for the transition from Québec's green levy. 80-100% for combustion emissions, declining 1-2% annually. 100% for fixed process emissions, non-declining. 2021-23: between 90%-100% for all industrial sectors, 60% for electricity production in narrow cases.	Declining CAF for combustion emissions, fixed for process emissions.
Korea	Historical activity level	100% free allocation for sectors as determined by carbon leakage index for Phase 3 (2021-2025) as well as certain public sector services. Allocation = *Benchmark value (tCO_2e/t) x historic activity level (t) x correction factor x carbon leakage factor.* The carbon leakage factor is 1.0 for sectors exposed to significant risk; for non-EITE sectors, it is 0.9.	Cap reduction determined on a per-allocation-phase basis. Phase 3 constitutes a 4.7% decrease in emissions compared to Phase 2. Cap reduction reflected in cross-sectoral correction factor.
New Zealand	Annual production	Two tiered: 90% for highly emissions intensive eligible industrial activities; 60% for moderately emissions intensive eligible activity	1 % annual reduction specified in legislation, but not yet triggered.

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