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The resource, waste and climate conundrum: Essays on the economics of recycling

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THE RESOURCE, WASTE AND CLIMATE CONUNDRUM: ESSAYS ON THE ECONOMICS OF RECYCLING

Résumé en français

Dans cette thèse, je propose une analyse de l'économie du recyclage lorsque trois types d'interactions entre les activités anthropiques et les milieux naturels sont considérés : i) la production/consommation de biens et la disponibilité en matières premières vierges, ii) la production/consommation et l'accumulation de déchets, et, iii) la production/consommation et les émissions carbonées associées. Au cours de quatre chapitres, j'explore le potentiel que représente le recyclage pour le développement d'un modèle économique durable en évaluant ses impacts environnementaux, en analysant les objectifs de politiques publiques et en remettant en question le cadre d'analyse concernant la durabilité.

Recyclage sous contraintes environnementales, climatiques et de ressources

Ce chapitre étudie les possibilités de recyclage d'un secteur industriel limité par les capacités en matière de ressources, de climat et de déchets. Avec les difficultés de développement de marchés et d'industries spécifiques, les intrants recyclés luttent pour concurrencer les intrants ordinaires dans la production, en particulier lorsque des taux de recyclage élevés entraînent des coûts importants. Cependant, j'analyse comment la prise en compte des coûts sociaux liés à l'extraction des ressources, à leur utilisation et à leur fin de vie, pourrait modifier l'arbitrage entre extraction et recyclage. Cette recherche contribue à l'analyse du recyclage à travers les outils de la littérature sur l'économie des ressources.

Pour ce faire, je modélise le cycle de vie complet d'un bien afin de prendre en compte les déchets et les émissions de gaz à effet de serre (GES) provenant à la fois de sa production – à partir de matériaux vierges ou recyclés – et de sa consommation. Le planificateur social choisit les quantités de ressources (vierges ou recyclées) et la part de déchets à valoriser. Dans une première itération, seule l'accumulation des déchets est prise en compte par l'introduction d'une fonction de dommage aux déchets. Dans une seconde itération, les

émissions de GES sont introduites à travers un budget carbone.

J'identifie d'abord les trajectoires optimales d'utilisation des ressources, principalement en fonction de la rareté relative des ressources et de leurs émissions lorsqu'elles sont incluses. Bien que le recyclage soit généralement, et à juste titre, considéré comme une opportunité de réduire l'impact de la consommation sur les ressources primaires et les déchets, je considère également les éventuelles conséquences environnementales négatives du recyclage et je discute des arbitrages qui en découlent. Le recyclage est en effet également une source d'émissions de GES, bien que dans la plupart des cas, elles soient plus faibles que pour les ressources extraites. La structure du modèle (coûts privés linéaires et substitution parfaite) donne des solutions en coin concernant le choix de la ressource. La ressource vierge est exclusivement utilisée pendant une première phase de production. Ensuite, la ressource recyclée peut être utilisée si elle devient socialement plus rentable, soit parce que la rareté de la ressource vierge prévaut, soit parce que les coûts privés favorisent l'utilisation du recyclage, et, soit parce que la différence d'émissions entre les deux ressources est suffisamment grande. Je montre également qu'il peut y avoir une phase de rattrapage de la consommation à la fin du programme du planificateur social.

Je caractérise aussi la stratégie optimale de recyclage et je montre que dans certains cas, le taux de recyclage à travers le temps est en forme de U inversé. Le recyclage est bénéfique à deux égards : il allège le coût de l'accumulation des déchets et permet de produire à partir d'un intrant recyclé. Cela montre la dualité de l'activité de tri des déchets : un objectif pratique de réduction des coûts et un objectif spéculatif d'utilisation potentielle d'intrants recyclés à l'avenir. Lorsque le recyclage est trop coûteux, une phase où le taux de recyclage est nul se produit (bien qu'il soit toujours possible de produire à partir des matériaux recyclés déjà stockés).

Enfin, je discute des implications politiques du modèle en identifiant et en analysant l'ensemble des schémas optimaux de taxes-subventions, et je mets en évidence l'existence d'externalités environnementales standards ainsi qu'une externalité positive liée à l'absence

d'un marché des déchets. Pour implémenter cette solution dit first-best, il faut introduire dans les deux secteurs une taxation basée sur les émissions de GES, pondérée par l'intensité carbone de la branche. Elle s'ajoute à un système de taxation-subvention basé sur les déchets et la reconstitution des stocks, reflétant l'externalité positive.

Impacts sectoriels, sur les ressources et le carbone, de l'augmentation du recyclage du papier et du carton

Dans ce chapitre, j'évalue le rôle du recyclage du papier et du carton dans le secteur forestier, tant du point de vue économique que de celui du climat. La question de savoir si le recyclage du papier et du carton a un impact positif ou négatif sur le carbone est cruciale car le secteur forestier mondial joue un rôle central dans l'atténuation du changement climatique : le secteur du recyclage du papier et du carton est bien développé en France, avec 66% de la production provenant de déchets récupérés. Des impacts complexes en cascade sur la ressource et les autres produits bois suggèrent des impacts économiques et environnementaux incertains. Historiquement, les facteurs de déplacement des GES du recyclage du papier ont été étudiés avec des hypothèses économiques fortes. Cela a conduit à des résultats contradictoires entre les études, en raison des différences dans les méthodes de comptabilisation des émissions, et en raison des limites organisationnelles, spatiales et temporelles du secteur considéré.

Pour cette raison, je modélise l'industrie du recyclage du papier et du carton dans le cadre d'un modèle d'évaluation intégrée de la forêt française, afin de la relier aux autres produits bois et à la ressource forestière. Sur la base de la situation actuelle du secteur forestier français, j'examine l'impact d'une augmentation du recyclage. Pour simuler une augmentation du recyclage, je mets en œuvre une réduction des coûts dans le secteur. Un module économique d'équilibre partiel des produits bois et une structure récursive de la dynamique forestière étendent une analyse du cycle de vie standard avec à la fois des impacts économiques sur d'autres secteurs de produits bois et des variations dans la substitution et

la séquestration du carbone.

En explicitant le modèle, il est clair que les résultats dépendent fortement des hypothèses concernant la substituabilité de la pâte à papier recyclée et de la pâte à papier vierge, alors que les nombreuses études existantes ne tranchent pas sur la question de savoir si c'est la substitution ou la complémentarité qui prévaut (certaines soutiennent qu'avec le temps, la complémentarité prévaut). Empiriquement, les élasticités semblent être faibles et dépendent de l'utilisation finale du produit en papier ou en carton. Les résultats qualitatifs attendus concernant la complémentarité ou la substitution des deux produits sont trouvés, avec respectivement un impact positif ou négatif sur la demande lorsque le recyclage augmente. Les émissions de GES du secteur de la pâte à papier sont en augmentation dans les deux cas, ce qui soulève la question de leur éventuelle atténuation par le reste du secteur forestier. Les effets sur les autres produits bois sont faibles, en raison des faibles élasticités-prix des produits en concurrence pour les ressources en bois et des faibles élasticités croisées entre vierge et recyclé.

Dans l'ensemble, l'impact sur la séquestration nette totale est : positif lorsque les produits à base de pâte sont des substituts forts ; très faible lorsque les produits à base de pâte sont des substituts faibles ; et négatif lorsque les produits à base de pâte sont des compléments. Les effets relatifs sont faibles (entre $0,4$ % et $0,8$ % d'augmentation du bilan carbone du secteur en cas de substitution forte, un scénario très optimiste selon la littérature économétrique). Comme la faible complémentarité entre le papier vierge et le papier recyclé semble être l'hypothèse la plus valable, l'idée suggérant que le recyclage et l'économie circulaire sont une bonne solution contre le réchauffement climatique doit être prise avec précaution : dans cette situation, une augmentation de l'utilisation du recyclage ne s'accompagne pas d'une économie des ressources en bois.

Une évaluation de la réglementation européenne sur le recyclage des batteries des véhicules électriques

Ce chapitre étudie une récente proposition de réglementation européenne visant à favoriser l'émergence d'une industrie du recyclage des batteries en Europe. La mobilité électrique est jugée nécessaire pour réduire les émissions de CO_2 dans le secteur des transports. De plus, les batteries lithium-ion (Li-ion) - les composants centraux des véhicules électriques (VE) - ont fait l'objet de progrès technologiques remarquables ces dernières années. En conséquence, leur coût est passé de 1100 USD par kilowattheure (kWh) en 2010 à 156 USD/kWh en 2020. Cependant, les impacts industriels et environnementaux de la fabrication des batteries lithium-ion sont controversés. Parmi eux, la fin de vie des batteries peut être particulièrement dangereuse. Ajouté à une potentielle rareté de certains métaux (lithium, mais aussi terres rares), le recyclage des batteries usagées apparaît comme une solution séduisante. Pour faire face à ces problématiques, l'Union Européenne (UE) a développé plusieurs initiatives, telles que la European Battery Alliance et la Raw Materials Initiative. La nouvelle proposition de la Commission européenne vise à ajouter un cadre législatif à ces initiatives.

Le cadre réglementaire actuel de l'UE sur les piles et batteries est fixé par une directive de la Commission européenne de 2006. Cette directive vise à fournir des règles communes à travers le marché européen qui limiteraient les impacts environnementaux des déchets de batteries. Cependant, le bouleversement provoqué par l'arrivée des batteries Li-ion, alimenté par le développement de l'électromobilité, a rendu cette ancienne directive obsolète. Une récente proposition réglementaire de la Commission européenne introduit l'obligation d'atteindre une série de seuils minimaux de matériaux recyclés pour les nouvelles batteries qui seront fabriquées après 2030. Les matériaux concernés pour les batteries Li-ion sont le nickel, le cobalt et le lithium. Je discute dans ce chapitre des conditions requises pour que cette obligation soit remplie.

Tout d'abord, je développe un cadre d'analyse qualitative pour discuter des principales

tendances caractéristiques du secteur des batteries et de la manière dont elles devraient affecter l'approvisionnement en matériaux recyclés. Ce cadre d'analyse permet de séparer les différents mécanismes qui déterminent la quantité de matériaux recyclés pouvant être incorporés dans les nouvelles batteries : l'évolution de la demande en batteries, la composition des matériaux liée aux changements technologiques, l'efficacité du recyclage et la probabilité de fin de vie.

Deuxièmement, je calibre un modèle de flux de matières qui effectue des projections de flux de déchets de batteries et leur potentiel de recyclage. En utilisant des données provenant des rapports économiques de l'Agence internationale de l'énergie et de la littérature académique, le modèle indique que la faisabilité des seuils actuellement proposés n'est pas très sensible aux changements d'intensité des matériaux liés aux changements technologiques des batteries, aux efficacités de recyclage ou à l'accélération de la demande. Au contraire, la durée de vie des batteries est le paramètre le plus crucial pour le potentiel de recyclage et le respect des seuils proposés. En effet, la plupart des objectifs fixés par la Commission européenne ne sont pas atteints pour des durées de vie moyennes des batteries supérieures à 10 ans. Une telle hypothèse peut être considérée comme irréaliste compte tenu des progrès technologiques réalisés dans le secteur des batteries, avec des durées de vie plus longues. En outre, un scénario lors duquel la seconde vie des batteries serait développée réduirait considérablement le potentiel de contenu recyclé. Dans ce cas, les batteries des VEs sont remises à neuf lorsqu'elles ne répondent pas aux normes de puissance des VEs et sont utilisées pour le stockage stationnaire.

Les recommandations de politiques publiques se déclinent en deux volets. Premièrement, je recommande d'abaisser les seuils pour améliorer la crédibilité de la réglementation. Bien que les objectifs puissent être modifiés à l'avenir, les acteurs industriels du secteur (fabricants de batteries et de voitures, industrie du recyclage) auraient besoin d'une réglementation lisible et crédible aujourd'hui pour faire les bons investissements et les bons choix technologiques. Deuxièmement, la réglementation devrait intégrer d'autres objectifs qui con-

cernent la durée de vie des batteries. En effet, le développement d'un secteur durable des batteries ne doit pas reposer uniquement sur des objectifs de recyclage et de contenu recyclé. De plus, le fait de ne fixer que des objectifs de contenu élevés pourrait inciter à réduire la durée de vie des batteries, car une durée de vie plus courte permettrait une circulation plus rapide des matériaux dans l'économie, augmentant ainsi l'utilisation potentielle de matériaux recyclés.

Le recyclage dans un cadre durable

Dans ce dernier chapitre, j'étudie la question du recyclage en macroéconomie dans le cadre de la durabilité. En effet, l'étude du recyclage en macroéconomie vise à évaluer comment les boucles fermées de matériaux sont capables d'améliorer la croissance à long terme avec un épuisement optimal des stocks de ressources. Cette analyse peut également prendre en compte la réduction des externalités environnementales grâce au recyclage. En ce sens, le recyclage, et plus généralement le concept d'économie circulaire, sont censés être un moyen de suivre une voie de développement durable. Cependant, l'économie de la durabilité est également liée aux questions d'équité en termes de consommation et d'impacts environnementaux. A ce sujet, la littérature sur le recyclage ne s'appuie pas sur un critère durable ; dans sa grande majorité, elle s'appuie sur l'utilité escomptée, un critère ne favorisant pas l'équité.

Pour explorer le sujet du recyclage dans un cadre de modélisation durable, je développe un modèle de ressources et de déchets résolu à l'aide du critère du maximin. La valeur maximin est définie comme le plus haut niveau d'utilité qui peut être maintenu à l'infini. J'utilise un modèle basé sur le modèle de Dasgupta-Heal-Solow, pour lequel la littérature indique qu'un chemin maximin existe dans le cas de la durabilité faible. Cela correspond au cas où la fonction de production ne dépend pas fortement de la ressource naturelle (pour une fonction de Cobb Douglas, lorsque l'élasticité de la ressource dans la production est inférieure à celle du capital).

Ce modèle simple complexifie l'analyse d'un niveau minimal de consommation durable. Je suis en mesure de caractériser un chemin optimal et ses valeurs fictives, et de discuter de certaines trajectoires à long terme. Des règles plus complexes de gestion des ressources apparaissent via une nouvelle règle d'investissement de Hartwick qui dépend maintenant de l'accumulation des déchets et de l'utilisation du recyclage dans la production. Dans le cas spécifique d'un stock de déchets vidé en temps fini, il est possible de discuter des chemins de consommation explicites len spécifiant les fonctions de production. Dans le cas d'une fonction de production Cobb-Douglas, l'hypothèse d'une élasticité du capital supérieure à l'élasticité de la ressource dans la production est cruciale pour qu'une trajectoire durable existe.

Ce cas spécifique fragilise l'hypothèse que le recyclage puisse permettre des trajectoires durables dans le cas de la durabilité forte. Ce modèle est un premier pas vers l'étude du recyclage à l'aide des outils de l'économie de la durabilité. À ma connaissance, c'est en effet le premier modèle qui étudie le recyclage avec le critère du maximin, et qui met en évidence comment les règles d'investissement optimales sont modifiées.

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* * *

Où s'arrête le jeu, où commence l'art [...]

la récup' ménagère est entrée dans le marché de l'art

et quand je dis marché, je ne dis pas bon marché.

Les glaneurs et la glaneuse - Agnes Varda (2000)

* * *

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An economy of material resources

An historical perspective on a material economy

Global primary material use is expected to double to support world gross domestic product (GDP) growth in the next 40 years (OECD, 2019). These figures remind us that economic development remains widely based on material resources. It may seem outdated in the 21st century, with an ever-increasing presence of a so-called dematerialized economy and a reduction of the share of agriculture and industry in the production apparatus of developed countries.

Historically, economic development prior to the industrial revolution was strongly linked to the exploitation of natural resources. Availability of resources was central in the economic thought, with what is called an "organic regime of growth". The focus on resource availability goes from the agricultural structuration at the core of physiocracy to manufacturing and commercial exchanges that structure the emerging classical economy with Adam Smith (Charbonnier, 2021). However with the introduction of coal in the 18th-19th centuries (the start of the industrial revolution), a transformation of the industrial apparatus allowed for an apparent freeing from the limitations of the organic regime. This freedom came from intensive growth via a significant increase in productivity, as well as an extensive growth in

access to natural resources.

The fatal relationship between economic growth and the finite resource (coal) was already questioned by William Stanley Jevons who anticipated a limit to the techno-industrial superiority of Great-Britain (Jevons, 1865). Jevons introduced a now widely known paradox: gains in productivity which lead to a more sparing use of coal, are offset by more extensive use. Even knowing the paradox, this concern is put aside with the use of coal's modern counterpart, oil, due to the "freedom" from domestic resource constraints for western economies, with an apparent abundance. It took until the second half of the 20th century for concerns regarding natural resources availability to be revived. Those concerns are most famously expressed by the Club of Rome, whose report on the limits to growth showed that physical limits of natural stocks lead to a dead-end (Meadows et al., 1972). The report considers the economic system as a global one that is built on natural capital. This approach is contemporary of Georgescu-Roegen's work on the integration of physical concepts (mainly thermodynamics) into economics in order to reconciliate economic sciences with physics laws that rule the indispensible use of energy and material resources in production (Georgescu-Roegen, 1970). These considerations are added to an ever-growing ecological conscience regarding risks to natural capital: local pollution, disruption of ecosystems and climate change. This unfolding of economic thought lead us to the current observation of a prominent role for natural resources in the economy, from biological to energy to mining resources.

At the same time, certain neoclassical economists have also integrated resource considerations in their models and analysis for years. Solow's classical growth model was enriched by the inclusion of natural resources alongside capital and labour as factors of production. However, limitations of natural resource supply were overcome by the assumption of unbounded material productivity. Unbounded productivity relies on two concepts: substitution and technical progress, both of which are derived from the hypothesis of neoclassical growth models (Daly, 1997; Couix, 2019). This controversy between neoclassical resource

economics and ecological economics gained in interest in the 21st century, as concepts of dematerialized and circular economy became more broadly discussed. The main question is whether a transition towards an economy decorrelated from material constraints can be achieved. In the same way, the history of energy sources is put in terms of transitions while some argue that additions of energy sources is more accurate (Fressoz, 2020).

Material resources in the economy

Material flows in the economy are now accompanied by a broad topic of concerns. The significant growth rate of resource use for metals and non-metallic minerals motivates the monitoring of material flows in the economy. Closer monitoring has revealed a growth of demand driven by demographic and economic growth. The nature of the demand is also changing, with variations across countries and across time as new technologies emerge. A so-called dematerialization of the economy puts away, for instance, material considerations in production, especially in developed countries. However, the material dependency of the economy is still subject to scarcity issues that come from several factors.

First, the risk of physical scarcity of resource stocks has been put forward since the the Meadows report of 1972. However, the notion of scarcity bears several limits as there is a difference between the "existing resource" and the "available stock". The size of the available stock is used to define the size of the reserve and can be described as the currently exploitable resource. The level of exploitable resource is conditioned on the known size of the stock and the current price of the resource that would make extraction economically feasible. This economic dimension allows to define a second stock that includes the geologically available resources that are not economically exploitable. With this first form of criticality lies a trade-off in resource use, which is choosing the specific stock to extract it from, based on the underlying cost. To these stocks should be added potential stocks of resources that are either identified but unexplored, or unidentified but potentially existing. For instance, in Central and South America, estimations of available copper resources are three times higher

than the current reserves, which alread represents half of the world's supply (Seck et al., 2020).

Second, the criticality of material resources also relies on their uneven geographical distribution. This situation leads to geostrategical risks when the resource access can be limited. For instance, lithium reserves are concentrated in South America (Bolivia, Argentina and Chile) while this metal is central to the development of new technologies for the energy transition. Another example is the concentration of rare earth elements; after world supply was dominated by the US who extracted from the single Mountain Pass Mine until the 1990s, China has since become the major supplier with over 90% of the production. The strategical reduction of their extraction through quotas in years 2000s lead to a dramatic increase in prices and subsequent rulings by the World Trade Organization in 2014 (Wan and Wen, 2017). On top of this geopolitical framework, the oligopolistic structure of these resources gives certain actors control of a majority of the resource supply. For instance, Glencore produced 27% of cobalt and Albemarle produced 24% of lithium in 2019 (IEA, 2021b).

Third, environmental degradation can also lead to degraded access to natural resources, resulting in lower quality or higher prices. This is the case for the forest resource which is vulnerable to climate change and subject to over exploitation (Lobianco, Delacote, Caurla and Barkaoui, 2016).

With these considerations, countries and organizations have the incentive to list and monitor critical raw materials (CRMs). To this end, the European Union (EU) established the Raw Materials Initiative to assess the criticality of materials based on the economic importance and the existing risk for their supply. [1] However, this kind of listing often remains arbitrary, as criticality can be defined according to different criteria, methodologies or industrial and strategical priorities being taken into account (Hayes and McCullough, 2018).

[1] https://ec.europa.eu/growth/sectors/raw-materials/policy-and-strategy-raw-materials_fr

Resource economics play a role in assessing those risks of scarcity. It involves arbitrating in two directions. First, between resource types, as increasing prices can lead to turning away from one supplier of a mineral to another, or even to a different resource substitute. Second, trade-offs across time, as a limited resource supply has to be allocated between generation of consumers. One famous economic result is Hotelling's rule which defines the price of the resource when the economic surplus is maximized across time (Hotelling, 1931). It isolates a scarcity rent, called the Hotelling rate, that increases at the discount rate. Economics also give three indicators to measure resource scarcity. First, the marginal cost of extraction: it takes into account technological change but omits other costs (costs to access to the market, exploration costs,...). Second, the market price: it includes all costs and the scarcity rents, but, it also takes into account other distortions such as economic policies and geopolitical contexts. Third, the scarcity rent, or opportunity cost: theoretically, the most accurate indicator, but, difficult to observe. However, as we will discuss in the following, this trade-off in resource use can be extended by considering externalities linked to their extraction and use (Watari et al., 2020).

Local pollution and social impacts

Relationships between natural resources and the environment are numerous and intervene at all stages of their use in the economy: extraction, transformation and final use.

First, many externalities relate to the extraction of the resource. Metal mining often consumes large volumes of water and can therefore contribute to groundwater pollution. The case of lithium in South America is a characteristic one, as it has been reported that lithium extraction in Chile's Atacama region uses 65% of the region's water tables. Extracting one ton of lithium uses 2,000 tons of water,[2] seriously disrupting local ecosystems and releasing toxic chemicals. Regarding biological resources, exploitation of the forest resource can harm ecosystems via the use of chemical intrants and a lack of diversity in planted species. These

[2]Katwala, A. The spiralling environmental cost of our lithium battery addiction. Wired https://www.wired.co.uk/article/lithium-batteries-environment-impact (2018)

things cause soil and water pollution as well as a reduction in ecosystem services provided by the forest (Gamfeldt et al., 2013). Additionally, these extraction impacts can last long after the activity, thus posing intertemporal threats to deal with.

Second, transformation of resources can rely on heavy industries, with important environmental impacts of their own. In the copper industry, tailing resources have to be stored on site and pose threats in terms of chemicals infiltration in groundwater (acid and heavy metals), as well as a risk of dams breaking. Moreover, metal smelting produces large amounts of metal fumes (arsenic and lead) and health-threatening gases (SO_2)(Li et al., 2017).

As mentioned previously, resource exploitation activities are often geographically localized. As such, polluting industries often relocate to countries where environmental constraints are less stringent (Bernard, 2015). This increases the risk of negative environmental impacts of resource extraction and transformation, as industries are less incentivized to reduce their externalities. This localization also adds up to social impacts of mining activities. A weaker institutional context as well as lesser capacities to enforce law lead to resource management practices with many health and safety issues as well as child labor concerns. A well documented example is the case of cobalt extraction in the Democratic Republic of Congo (DRC) where 70% of the world production came from in 2019 (Seck et al., 2022). Part of these social risks comes from the fact that around 20% of DRC mines are artisanal with workers and local populations confronted to more severe and unregulated environmental pollution. Furthermore, the DRC is subject to political instability leading to regular surges of violence in the region, affecting mining industries and adding to the scarcity risks mentioned before. While the sources of those conflicts come from various causes, presence of resource extraction activities has been highlighted as one of the factors (Fourati et al., 2021; Mukwege, 2021).

These considerations add up to define the trade-offs for resource use. Given the various environmental externalities and negative social impacts of resource extraction and transformation, the analytical framework of economists cannot be limited considering only to the

cost of extracting a resource. Analytical frameworks must also take into account various explicit or implicit costs associated with resource extraction. Additionally, long term climate impacts and end-of-life management of the resource have to be taken into account to properly choose between the different kinds of resources available for production in the economy.

Climate change

In addition to local externalities due to effluents of the production and transformation of natural resources, virgin material consumption has an important impact on climate change. In the case of plastics, the industry contributes to fossil fuel exploitation as oil and gas are crucial for their manufacturing. Up to 6% of EU's consumption of oil and gas is dedicated to the production of plastics (Plastics Europe, 2017). Metal production is also a huge emitter of greenhouse gases (GHG): it has been estimated that around 7 to 8% of the world's energy supply is dedicated to mining and refining metals (UNEP, 2011). Furthermore, 23% of global GHG emissions (2015) are related to some form of material harvesting (UNEP, 2019). For the specific case of aluminum, harvesting it accounts for 1% of global GHG emissions (Gautam et al., 2018). The relevant local energy framework plays an important role in these results, as we can find significant variations across countries with different energy mix. It can varie from 8,000 kg CO2e per ton of produced aluminum for France to 40,000 kg CO2e/t for China.

In a different way, the forest resource is also strongly connected to GHG emissions reduction as it is one of the main carbon sinks, alongside with the oceans, related to many impacts of climate change (Cook-Patton et al., 2020). Harvesting the forest for the production of wood products gives the opportunity for a renew of the forest cover, thus playing an important role in GHG removal. In France, the carbon sequestration potential is estimated between 66.2 and 125.3 $MtCO_2$ per year (15 to 28% of CO_2 equivalent emissions in France in 2019) (Lobianco, Caurla, Delacote and Barkaoui, 2016).

Climate change and resource extraction are also linked through investments needed for the energy transition. Considerable amounts of minerals are required in order to develop the necessary infrastructure to abate GHG emissions, especially in the energy sector (IEA, 2021b). In the future decades, some mineral extraction sectors are expected to be increasingly dedicated to these, for instance lithium, nickel and cobalt for the electrification of vehicles. This observation is also an incentive for countries and international organizations to carefully monitor material supply chains as well as the underlying environmental and social impacts. The previously presented Raw Material Initiative in the EU is one of them.

Waste accumulation

A central concern of material use in the economy is the accumulation of waste. Waste is defined as material discarded after use, or any economic process including extraction and transformation of resources (industrial waste). For instance, we account for the previously mentioned copper tailing and production scraps into the broad definition of waste: materials that are disposed of when they have been used and they do not bear any value for the consumer (solid waste). In that sense, ownership of waste is ambiguous as it tests the boundaries of what is defined by ownership: as defined by Locke, waste land does not have value or an owner from the moment when it is not exploited (Lupton, 2011). However, the definition can be challenged by thinking of waste (in the literal sense) as something that has both an economic value and a destined use. Additionally, waste has a host of of negative externalities to consider and characterize. We will first focus on the latter in this section.

Environmental damage from waste mainly comes from the different management methods. First, landfilling can be the source of groundwater pollution, soil contamination, fire hazards and olfactory nuisances. In addition to these externalities, landfilling can be the source of high costs due the large land areas allocated to waste deposits.[3] Second, waste

[3]Talpin, J. Saturée de déchets, La Réunion veut stopper le tout enfouissement. Le Monde https://www.lemonde.fr/planete/article/2019/12/28/saturee-de-dechets-la-reunion-veut-stopper-le-tout-enfouissement_6024253_3244.html (2019)

incineration generates air pollution and slags that are highly polluting (especially because the presence of heavy metals) while also requiring waste storage capacities. on top of the externalities relative to waste management, we must also add externalities linked to waste outside of the waste management sector. More specifically, we must consider waste that is illegally disposed of, waste that ends up in the environment by mistake or carelessness, or that is produced through the consumption of products that are dispersed in the environment. An iconic example is the case of marine plastic pollution. Driven by the growth of its production (from 0.5 Mt in 2050 to 260 Mt in 2009 and 320 Mt in 2015), plastic now accounts for a stockpile of 8.3 Gt worldwide (McNicholas and Cotton, 2019). It is estimated that now 40% of plastic production is not managed by the time it becomes waste, thus ending up in terrestrial or marine ecosystems. Macroscopic and microscopic waste (after decomposition) are persistent pollutants with sometimes irreversible consequences on the environment. Macro plastics induce impacts on biodiversity and on human activity through thing like lethal ingestion by animals, a beaches clean-up cost, and disruptions to fishing and tourism. The impacts of microplastics are subject to more nuance and uncertainties. For example, microplastics can be ingested by fish and then by humans through fish consumption. As such there are multidimensional health threats. While marine plastic pollution is rapidly increasing in geographically located areas, such as the Great Pacific Garbage Patch (Lebreton et al., 2018), it is hard to assess damages for waste, as the impacts are often scattered across several environments with complex behaviors and synergies. In studying marine ecosystem services, economists have estimated costs ranging from $3,300 to $33,000 per ton of marine plastic (Beaumont et al., 2019).

As for environmental impacts of resource management, we observe a concentration of waste damage in developing countries where environmental regulations are weaker and the cost of labor is cheaper (Bernard, 2015). For instance, South East Asia is now a major importer of waste worldwide. There have been consecutive geopolitical crisis when China strictly restricted conditions for waste importation in 2018. A good example is the ship

dismantling industry, which is concentrated in Asia due to limited regulatory environments (Devaux and Nicolaï, 2020). Social and environmental consequences are numerous: unsafe working conditions, high amounts of toxic effluents damaging the environment, and innumerable health hazards for local populations.

The recycling challenge

Confronted with the joint problems of increasing amounts of waste to manage and a growing demand in primary material, recycling gained economical and political momentum in recent years as part of the concept of circular economy. As opposed to a linear economy, recycling it is part of a wider reflection on how to transform the economy from a unidirectional model based on the "extract-consume-throw away" tryptic to a more sustainable one. As the industrial process that enables to "close the material loop" in the economy, recycling is a key element of the circular economy. However, the circular economy is a broader vision of a societal change that supplements recycling with changes in supply and demand of products (ADEME, n.d.).

Historical perspective

Concerns regarding a linear model first arose from the nuisances related to production residuals, especially in urban territories. These concerns were widely documented as early as the Middle-Age (de Silguy, 1996). Managing them is then a matter of public health and results in the progressive implication of governments: waste collection, paved streets, sewers... However residuals are rapidly considered as useful resources. Starting in the industrial revolution and then picking up in the 19th century, waste becomes the subject of an intense economic activity. An prime example is the paper industry.[4] To meet the demand for paper rags were initially used. Ragpickers form an important social body, greatly influencing au-

[4]We can also cite animal bones collection for many manufactured objects and manure for agricultural fertilization.

thors and painters during the 19th century. Rag workers represented up to 100,000 workers in France (Compagnon, 2017).

The modern conception of waste, especially solid waste, was built around the turn of the 20th century. Near that time, the profession of ragpickers disappears as the paper industry turns towards vegetable cellulose.[5] The modern concept of waste focused on eliminating it from the public (and private) space, with landfilling becoming the common practice, especially from a purely economic sense (Monsaingeon, 2020).

A new transformation of relationships to waste occured in the second half of the 20th century, with environmental issues carving out a path in the public debate. Saturation of landfills and related pollution drove governments of western countries to legislate the waste management sector. Additionally, the development of incineration and recycling spawned out of an attempt combat increased consumption, and the use of plastic packaging. In France, waste sorting emerged locally in the 1970s, with glass container collection, and then generalized in the 1990s to include plastic containers. In 1975, laws and regulations on waste management required local authorities to put in place solid waste collection services. However, elimination remained the first alternative to landfilling, with almost 40% of solid waste being incinerated in 1989 and still 52% ending up in landfills (Lupton, 2011). It wasn't until the development of the concept of a circular economy that recycling gained prominence (in the late 20th - early 21st century).

The circular economy first emerged in the 1990s and it is still widely discussed and contested today. It first became popular in non-academic circles, promoted by non-governmental organisations, and later by businesses and governments. The *Ellen McArthur Foundation* has for instance played an important role in its popularization. One key approach is to consider the use of resource *from cradle to cradle* as formalized by Braungart and McDonough (2002). The idea is to rethink the economic system so that all resources can be recovered after use and reintegrated in the economy or the environment, without any damage. Be-

[5]Bones collection is also replaced by synthetic resins and fertilization now uses chemical fertilizers.

yond the purely economical aspect, the concept now also stands as part of a new societal vision embedded in a sustainable economic model (Arnsperger and Bourg, 2017). One could consider the circular economy as a *mobilizing concept* or an *essentially contested concept*, as it binds many topics of concerns with the objective of creating a new sustainable paradigm everyone agrees on, but without a concrete idea of its feasibility or the path to achieving it (Gallie, 1956; Korhonen, Nuur, Feldmann and Birkie, 2018). While it has been linked to the topic of sustainability, the wide range of research looking into the circular economy can lead academics to consider it not only as an economic theory, but as a new holistic approach for ecological and economical concerns. Moreover, its apparent objective of breaking free from ecological limits through circularity appears to be an attempt to reconciliate physical constraints developed by ecological economics and sustainable growth advocated by neoclassical economics. With the increasing involvement of academic research in the topic, some limits have been pointed out in that sense (Korhonen, Honkasalo and Seppälä, 2018). For instance, it still bears thermodynamic challenges regarding resource and energy uses. From these considerations, developing a circular economy appears more as a tool for sustainable development than a goal in itself (Millar et al., 2019).

Policy framework

Reducing the impact of waste accumulation and promoting recycling is a central political question. An important feature is that it affects a large variety of environmental policies, from the ones specifically dedicated to waste management, to more general ones like carbon pricing. This holistic dimension comes from the several impacts of resource use in the economy, as described before. It gives policy makers a wide range of possibilities when regulating, but it involves complex impact studies when multiple instruments are involved. We will take here the example of France. Since 1975, France has deployed a large range of policies aiming at supporting recycling and lowering the impact of resource use from extraction to waste.

Extended Producer Responsibility (EPR) is the core principle for waste management. In that sense, an economic actor that produces a good is held responsible for the environmental impacts during its entire lifecycle. The objective of EPR is to incite the producer to adopt more environmental friendly usages. In practice, producers organize to form Producer Responsibility Organisations (PRO). PROs subsidize recycling by collecting fees from upstream producers that are set to match recycling costs. Financing waste management can also be completed by taxation by local or central authorities.[6] These policies are complemented by adjustments on the value-added tax (VAT) for collection and sorting services. These solutions are not the textbook pigovian tax advocated by economists, and their efficiency can be criticized. In particular, subsidizing recycling activities appears to be a less efficient policy. A better alternative is putting in place a deposit-refund program. Under such a policy, the consumer bears a supplementary cost only if the good is thrown away (Palmer et al., 1997; Ino, 2011; Acuff and Kaffine, 2013). In France, the approach has recently regained interest as demonstrated by the deposit refund system for plastic bottles which will be implemented under the *anti-gaspillage pour une économie circulaire (AGEC)* law (February 2020). [7]

Although economists have long advocated for direct environmental taxation due to its efficiency regarding the internalization of the social cost of externalities, it has rarely been implemented. As mineral and material industries are carbon intensive sectors from extraction to waste, taxation on GHG emissions can have significant reduction impacts (Acuff and Kaffine, 2013). At the European level, a trading scheme for CO_2 emissions has been implemented, with an forthcoming border adjustment mechanism to avoid carbon leakage outside the EU. Resource transformation is covered by this European Trading Scheme (ETS). However, end-of-life management of resources is subject to many exemptions. In particular, recycling and incineration are not subject to the ETS. On the other hand, the cement clinker

[6] In France, local authorities have different options for financing waste collection and curbside recycling, either independently or related to the provided service https://www.ecologie.gouv.fr/fiscalite-des-dechets.

[7] LOI n° 2020-105 du 10 février 2020 relative à la lutte contre le gaspillage et à l'économie circulaire https://www.legifrance.gouv.fr/jorf/id/JORFTEXT000041553759/

industry, which is an important part of waste incineration, is included, while benefitting an large share of free carbon allowances.

Taxation directly addressing pollution (other than GHG) also exists with the *Taxe Générale sur les Activités Polluantes* (TGAP). Created in 1999, it can be broken down into several components with each one targeting a different effluent. In particular, it includes a tax on waste flowing into landfilling and incineration industries, with adjustments depending on environmental performances (for instance, incineration with co-production of electricity). This TGAP is complemented by a disposal fee for solid waste. Disposal fees are still implemented on a voluntary basis from local authorities. In France in 2021, 6 millions inhabitants are subject to disposal taxation. Based on volume or weight, its goal is to discourage the disposal of non-recyclable goods (Bueno and Valente, 2019). However, economists have pointed out some limitations such as an increase of illicit waste burning or dumping (Fullerton and Kinnaman, 1995), ex-ante opposition to the taxation (Carattini et al., 2018) and the influence of private values and social norms when confronted with individual waste management practices (Viscusi et al., 2011; Czajkowski et al., 2019).

Another widely used type of policy instrument is the use of regulation to impose norms on producers of goods. For instance, with the *loi AGEC*, France introduced a ban on microplastics and a progressive ban of single use plastic products. Given the variety of effects on the environment and their persistence through time, there are difficulties in assessing the cost of plastic pollution and taxing it accordingly. Complementary to EPR mechanisms used to incentivize the production of greener products, norms on their composition are also enforced. New regulations aim at imposing a minimum share of recycled material in new products. Some initiatives have voluntarily originated from private actors at first, and later will become mandatory. For example, in France, the production of PET plastic bottles will require a minimum amount of recycled material after 2025. In addition to the upstream impact, it also forces a demand for recycled material. However this kind of measure can be problematic at an international level. Asymmetries between countries,

due to varying abilities of recycling waste, can lead to large increases of trade instead of changes in environmental performances (Huhtala and Samakovlis, 2002). Finally, policies also target information available for the consumer of the product. It includes reparability indexes for electronic products, sorting instructions and environmental labels. The large variety of existing labels, both from public and private incentives, is a limit regarding their efficiency and drove lawmakers to start standardizing them. Furthermore, many discussions exist regarding the efficiency of social norm nudging, especially when norms are very high to meet (Czajkowski et al., 2019).

Trade-off between resource, waste and climate

Historically, recycling has either been considered as a solution for a waste problem or a resource availability problem. However, recycling cannot separated from a wider analysis that includes the full lifecycle of materials. As detailed before, recycling is part of a path towards sustainability, as long as it is studied under an holistic approach. Resource availability, local pollution, social impacts, climate change and waste accumulation are all topics related to the use of resources in the economy. Therefore the decision to recycle goes from a single trade-off problem on resource stocks and pollutions levels to a multi criteria trade-off.

This multidimensional trade-off problem is at the core of this dissertation. The starting point is a standard stock and flows analysis, where recycling alleviates resource scarcity and allows for less damage from waste accumulation. However, as it has been noted before, welfare losses and gains have to include underlying effects on climate change. In that sense, we have to consider GHG emissions as an element of the trade-off. The objective is to assess the eventual synergies or antagonisms between the fight against climate change and waste, and resource regulations.

Contributions of this dissertation

Scope of the dissertation

In this dissertation, we propose an analysis of the economics of recycling when three types of interactions between anthropogenic activities and natural environments are considered: i) the production/consumption of goods and the availability of virgin raw materials, ii) the production/consumption and accumulation of waste, and, iii) the production/consumption and associated carbon emissions. Over four chapters, we explore the potential that recycling holds for the development of a sustainable economic model by assessing its environmental impacts, analyzing policy objectives and questioning the framework of analysis regarding sustainability.

Description of the chapters

Recycling under environmental, climate and resource constraints [8]

This chapter investigates the recycling opportunities of an industrial sector constrained by resource, climate, and waste capacities. With development difficulties of specific markets and industries, recycled inputs struggle to compete with regular inputs in production, especially when high recycling rates bear significant costs. However, we analyze how accounting for social costs from resource extraction, use and end-of-life, could change the trade-off between extraction and recycling. This research contributes to the analysis of recycling with the instruments of the literature on resource economics.

To do this, we model the full lifecycle of a good to consider the waste and GHG emissions coming from both its production — from virgin or recycled materials — and consumption. The social planner chooses resource quantities (virgin or recycled) and the share of waste to be recovered. In a first iteration, only waste accumulation is taken into account through

[8] This chapter was written in collaboration with Gilles Lafforgue. Published in *Resource and Energy Economics* (2022) https://doi.org/10.1016/j.reseneeco.2021.101278

the introduction of a waste damage function. In a second iteration, GHG emissions are introduced through a carbon budget.

We first identify the optimal trajectories of resources use, mainly depending on the relative scarcity of the resources and on their emissions when included. Although recycling is usually, and correctly, noted as an opportunity to reduce the impact of consumption on primary resources and waste, we also consider the possible negative environmental consequences of recycling and we discuss the resulting trade-offs. Recycling is indeed also a source of GHG emissions, although in most cases they are lower than for extracted resources. The structure of the model (linear private costs and perfect substitution) gives us corner solutions regarding the choice of the resource. Virgin resource is exclusively use during a first phase of production. Then, the recycled resource can be used if it becomes more socially profitable, either because the scarcity of the virgin resource prevails, because private costs favor the use of recycling, and, or because, the emissions difference between both resources is sufficiently large. We also show that there can be a catch-up phase of consumption at the end of the social planner program.

We also characterize the optimal recycling strategy and we show that in some cases, the recycling rate through time is an inverted U-shape. Recycling is beneficial in two ways: it alleviates the cost of waste accumulation and allows for production from a recycled input. This shows the duality of the activity of sorting waste: a practical goal being the reduction of costs and a speculative goal of potentially using recycled input in the future. When recycling is too costly, a phase where the recycling rate is zero occurs (although it is still possible to produce from the already stockpiled recycled material).

Finally, we discuss the policy implications of our model by identifying and analyzing the set of optimal tax-subsidy schemes, and we highlight the existence of standard environmental externalities as well as a positive externality linked to the absence of a market for waste. To do this first best solution, a taxation based on GHG emissions must be introduced in both sectors, weighed by the carbon intensity of the branch. This comes in addition to a tax-

subsidy system based on waste and stock replenishment, reflecting the positive externality.

Sectoral, resource and carbon impacts of increased paper and cardboard recycling [9]

In this chapter, we assess the role of paper and cardboard recycling on the forest sector, both from an economic and carbon perspective. The question of whether paper and cardboard recycling has a positive or negative carbon impact is crucial as the global forest sector plays a central role in climate change mitigation: the paper and cardboard recycling sector is well developed in France, with 66% of the production coming from recovered waste. Complex cascading impacts on the resource and other wood products suggest unclear economic and environmental impacts. Historically, GHG displacement factors of paper recycling have been studied with strong economic assumptions. This has led to contradictory results between studies, due to differences in accounting methods of emissions, and due to the organisational, spatial and temporal boundaries of the sector considered.

For that reason, we model the paper and cardboard recycling industry within an existing integrated assessment model of the French forest, in order to relate it to other wood products and forest resource. Based on the present situation of the French forest sector, we examine the impact of increased recycling. To simulate increased recycling, we implement a cost reduction in the sector. A partial equilibrium module of wood products and a recursive structure of the forest dynamics expands a standard Life Cycle Analysis (LCA) with both economic impacts on other wood-product sectors and variations in carbon substitution and sequestration.

In formulating the model, it is clear that results are highly dependant on assumptions regarding the substitutability of recycled and virgin pulp, while many existing studies do not settle on whether substitution or complementarity prevails (some argue that with time, complementarity prevails). Empirically, elasticities appear to be low and depend on the

[9] This chapter was written in collaboration with Philippe Delacote and Antonello Lobianco. FAERE working paper (2021) and submitted to *Environmental Modeling and Assessment* (2021)

final use of the paper or cardboard product. Expected qualitative results regarding the complementarity or substitution of both products are found, with respectively a positive or negative impact on demand when recycling increases. GHG emissions of the pulp sector are always increasing, thus raising the question of their possible mitigation through the rest of the forest sector. Effects on other wood products are small, due to small price-elasticities of products competing for timber resources and small cross-elasticities between virgin and recycled.

Overall, the impact on total net sequestration is: positive when pulp products are strong substitutes; very small when pulp products are weak substitutes; and, negative when pulp products are complements. Relative effects are small (between 0.4% and 0.8% in the additional carbon balance of the sector when strong substitutes, a very optimistic scenario according to the econometric literature). As weak complementarity between virgin and recycled paper seems to be more accurate, the idea suggesting that recycling and circular economy is a good solution against global warming should be taken cautiously: in this situation, an increase in the use of recycling does not come along with savings of wood resources.

An assessment of the European regulation on battery recycling for electric vehicles [10]

This chapter investigates the design of a recent regulatory proposal aimed at favoring the emergence of a battery recycling industry in Europe. Electric mobility is deemed necessary to cut CO_2 emissions in the transport sector. Moreover, Lithium-ion (Li-ion) batteries - the central components of electric vehicles (EVs) - have been subject to remarkable technological progress in the recent years. As a result, their costs have fallen from USD 1100 per kilowatt-hour (kWh) in 2010 to USD 156/kWh in 2020. However, the industrial and environmental impacts of lithium-ion battery manufacturing are controversial. Among them, the end-

[10] This chapter was written in collaboration with Quentin Hoarau. Published in *Energy Policy* (2022) https://doi.org/10.1016/j.enpol.2021.112770

of-life of batteries may be particularly hazardous. Added to a potential scarcity of some metals (lithium, but also rare earth materials), recycling of used batteries appears as an appealing solution. To address these issues, the EU has developed several initiatives, such as the European Battery Alliance and the Raw Materials Initiative. The new proposal of the European Commission intends to bring a legislative framework to these initiatives.

The current EU regulatory framework on batteries is currently established by a directive from the European Commission in 2006 (Council of European Union, 2006). This directive aims at providing common rules across the EU market that would limit environmental impacts of battery wastes. However, the disruption of Li-ion batteries fueled by the development of electromobility has rendered the former directive outdated. A recent regulatory proposal from the European Commission introduces the obligation to attain a series of minimum thresholds of recycled materials for the new batteries to be manufactured after 2030. The relevant materials for Li-ion batteries are nickel, cobalt and lithium. We discuss the conditions required for that obligation to be fulfilled.

First, we develop a framework to qualitatively discuss the main characteristic trends of the battery sector and how these should affect sourcing from recycled materials. It allows us to separate the different mechanisms driving the amount of recycled material that can be incorporated in new batteries: the change in battery demand, material composition linked to technological changes, recycling efficiency, and end-of-life probability.

Second, we calibrate a material flow model that projects battery wastes and their recycling potential. Using data from economic reports of the International Energy Agency and academic literature, the model indicates that the feasibility of currently proposed thresholds is not very sensitive to changes of material intensities from battery technology shifts, recycling efficiencies, or the faster uptake of demand. On the contrary, battery lifetimes are the most crucial parameters for recycling potential and meeting proposed thresholds. Indeed, most of the targets given by the European Commission are not met for average battery lifetimes beyond 10 years. Such an hypothesis can be considered as unrealistic given the

technological progress in the battery sector, with longer lifetimes. Moreover, a scenario with a developed second life of batteries would dramatically lower the potential for a recycled content. In this case, EV batteries are being refurbished when they do not meet EV power standards and are used for stationary storage.

Our policy recommendations are therefore twofold. First, we recommend lowering the thresholds to improve the credibility of the regulation. Although targets might be changed in the future, industrial actors of the sectors (battery and cars manufacturers, recycling industry) would need readable and credible regulations today to make the right investments and technology choices. Second, the regulation should integrate other objectives that address the lifetime of batteries. Indeed, developing a sustainable battery sector should not rely on recycling and recycled content targets only. Moreover, only setting high content targets could give incentive to decreasing battery lifetimes, as shorter lifetimes would allow a faster circulation of material in the economy, thus increasing the potential use of recycled material.

Recycling in a sustainable framework [11]

In this last chapter, we study the question of recycling in macroeconomics with the scope of sustainability. Indeed, the macroeconomics of recycling aim at assessing how closed material loops are able to improve long term growth with optimal depletion of resource stocks. Such analysis can also take into account the reduction of environmental externalities through recycling. In that sense, recycling, and more generally the concept of a circular economy, are expected to be a means for following a sustainable development path. However, sustainability economics also relates to equity issues in terms of consumption and environmental impacts. On this matter, literature on recycling does not rely on a sustainable criterion; in its vast majority, it relies on discounted utility, a criterion that does not favour equity and is thus unsustainable.

To explore the topic of recycling in a sustainable modeling framework, we develop a resource and waste model to be solved using the maximin criterion. The maximin value is

[11] Ongoing working paper

defined as the highest level of utility that can be sustained forever. We use a model based on the Dasgupta-Heal-Solow model, for which literature states that a maximin path exists in the case of weak sustainability. It corresponds to the case when the production function is not strongly dependent on the natural resource (for a Cobb Douglas function, when the elasticity of the resource in production is lower than the one of capital).

This simple model adds complexity to the analysis of a minimum sustainable consumption level. We are able to characterize an optimal path and its shadow values, and discuss some long-run analysis. More complex resource management rules are reflected by a new Hartwick investment rule which now depends on waste accumulation and the use of recycling in production. In the specific case of a waste stock emptied in finite time, it is possible to discuss explicit consumption paths when specifying production functions. In the case of a Cobb-Douglas production function, the assumption of an elasticity of capital higher than an elasticity of resources in production is crucial for a sustainable path to exist.

This specific case weakens the idea of recycling being able to allow sustainable paths in the case of strong sustainability. This model is a first step towards the study of recycling using the tools of sustainability economics. To our knowledge, it is indeed the first model that studies recycling with the maximin criterion, and highlights how optimal investment rules are modified.

CHAPTER 1

RECYCLING UNDER ENVIRONMENTAL, CLIMATE AND RESOURCE CONSTRAINTS

* * *

With development difficulties of specific markets and industries, recycled inputs struggle to compete with regular inputs in production, especially when high recycling rates bear significant costs. However, accounting for social costs from resource extraction, use and end-of-life of products, could change the trade-off between extraction and recycling. Although recycling is usually, and correctly, noted as an opportunity to reduce the impact of consumption on primary resources and waste, we also consider in this chapter the possible negative environmental consequences of recycling and we discuss the resulting arbitrations. We discuss the policy implications, and we highlight the existence of standard environmental externalities as well as a positive externality linked to the absence of a market for waste.

* * *

1.1 Introduction

The early economic studies on secondary materials long focused on recycling's ability to save extracted resources (UNEP, 2019). More recently, however, economists have begun to consider recycling's ability to mitigate waste pollution and greenhouse gas (GHG) emissions. With this new view, the concept of a circular economy, in which recycling is one of the cornerstones, appeared as a solution for a more sustainable economical model, as formalized by Braungart and McDonough (2002) in *Cradle to Cradle*. This concept has generated a significant amount of "grey literature" through many non-governmental organizations like the *Ellen McArthur Foundation*. It can be noted that a large concentration of the grey literature, as well as of the academic literature, occurs in Europe and Asia. Both of these regions have many academic and institutional initiatives in this field (de Jesus and Mendonça, 2018).

In this chapter, we contribute to the literature on circular economy by examining the impact of recycling in an industrial sector facing environmental, resource and climate constraints. In this context, we must consider three different balances: a material balance in order to examine the saving of natural resources and the reduction of waste accumulation; a carbon balance for the topic of climate change; and, an economic balance for the evolution of consumption. Hence, we see environmental objectives of recycling going in three different directions: the saving of natural resources while its shortage could lead to economic difficulties; the reduction of waste accumulation that is costly to manage for both public and private entities, and poses a threat to the environment; the fight against climate change and especially the reduction of GHG emissions. A good illustration of these features can be found in the metal sector: scarcity of critical metals raises the question of their sustainable use and end of life management, as well as the emissions rates of their lifecycle (UNEP, 2011).

One intended specificity of this chapter is to extend the study of recycling by considering

climate change as an additional externality. Recycling can indeed harm the environment, in particular because of the extra energy required by this activity and its resulting potential GHG emissions. This additional externality leads to new arbitrations we describe in the model: in most cases, recycling is a way of reducing the use of resources with a high carbon footprint (ADEME and FEDEREC, 2017), but recycling is still the source of GHG emissions. It has already been highlighted that circularity and environmental issues are connected in an industrial sector, with for instance used tires (Lonca et al., 2018). In France, studies of ADEME and FEDEREC (2017), ADEME and Bio by Deloitte (2017) and ADEME (2019) focused on quantifying different impacts of recycling in terms of GHG emissions, showing that industrial processes are often highly carbon intensive compared to recycling industries (see Table 1). However, in the end, recycling does not appear to be the ideal clean substitute to regular production: recycling produces its own emissions; there is a need in initial production from a regular source; and, recycling comes with a cost (UNEP, 2011). These environmental considerations contribute to the current policy framework, with many countries implementing emissions reduction targets that affect polluting industries.

	PET Plastics	Aluminum	Glass	Copper	Scrap/Steel
Virgin material	3 270	7 803	923	1 445	2 211
Recycled material	202	562	409	1 304	938

Table 1.1: Examples of emissions rates in France (in kg CO_2e/t)

Note: Values obtained from ADEME and FEDEREC (2017) and from bilans-ges.ademe.fr

We do not pretend to calibrate policies to implement, but we try to analytically highlight the cross-effects and eventual synergies between climate change mitigation, preservation of resources, and the limitation of waste disposal. To do that, we develop a dynamic model of resource use where a final manufactured good can be produced from a virgin exhaustible resource or from recycled materials, these two inputs being taken as perfect substitutes.

In order to clearly understand their various effects, we successively introduce two negative environmental externalities. First, we examine waste accumulation that harms the economy through a specific damage function, and then we add cumulative GHG emissions that are constrained by an exogenous carbon budget. Consequently, we observe that a third source of (positive) externality must also be considered as long as there is no waste market, as consumption of the final good provided a waste stream which can be reused thanks to an endogenous effort of recycling.

In this framework, we characterize the main properties of the optimal trajectories of the model. In particular, we discuss the merit order in using each type of resource, depending on the relative scarcity of resources and their emissions rates. We analyze the optimized recycling rate curve through time and we show that, under some conditions, it can be an inverted U-shape. We also discuss the respective dynamics of resource use, which sometimes can result in a catch-up phase of consumption at the end of the program. Last, we show that in a competitive market economy, this optimal outcome can be implemented by a set of tax-subsidy schemes and we discuss their policy implications depending on the identity of the tax payer or the subsidy beneficiary. The introduction of a waste market allows to account for the positive externality linked to production that relaxes resources restriction thanks to recycling. This externality comes from the dynamic framework of the model and has not been considered by the static economic literature on recycling.

The rest of this chapter is organized as follows. Section 1.2 provides a brief review of the literature on recycling. Section 1.3 characterizes the basic model and describes the different possible scenarios of consumption and recycling. Section 1.4 considers the introduction of GHG emissions through an emissions ceiling.[1] Section 1.5 studies the decentralized equilibrium outcome and discusses policy implications. Section 1.6 concludes.

[1]We also propose three other extensions in appendix: accounting for a scrap value for waste, decoupling emissions between collection and transformation in the recycling branch, limiting the capacity of the recycling sector.

1.2 Related literature

Our work can be related to three strands of literature on recycling: resource scarcity, waste management, and environmental policies.

Early studies from the 70s-80s already tackled resource scarcity. For instance, Smith (1972) puts forward social costs linked to waste accumulation and stock diminution. He focuses on the dynamics of waste when recycling is under consideration, and he shows that there is a trade-off between private costs (labor, material) and social costs (waste accumulation, resource depletion). Such dynamic models informed the first economic guidelines motivating recycling. Waste accumulation issues were added to the topic through various models intending to find the optimal level of pollution in an economy (see for instance Plourde (1972), Forster (1973), or Hoel (1978)). Later, the work of Chakravorty et al. (2006) and Chakravorty et al. (2008) focused on the order of resource extraction and gave many insights on situations where resource depletion induces pollution, that changes the extraction order. However, these studies do not include recycling in their model.

An important part of the relevant literature was later developed around the topic of using green policies to promote recycling. Palmer and Walls (1997) use a static micro-economic model to analyze the effects of diverse economic incentives such as subsidies, waste tax and deposit-refunds. This approach gives many policy insights but only takes into account waste and recycling activities. The work started by Palmer and Walls (1997) was then expanded to include environmental effects associated with recycling and resource extraction (Fullerton and Kinnaman, 1995; Palmer et al., 1997). Going further in this type of analysis, Walls and Palmer (2001) integrate life-cycle aspects of production and consumption and discuss optimal policy instruments. However, they only include an effluent linked to a third, non-material, input in production. Thus they do not differentiate between recycling and extraction. In a later work, Acuff and Kaffine (2013) add carbon emissions to the model with a direct link to the input choice and show that the objective of reducing GHG is also a

strong incentive to increase recycling, and that green policies can be implemented with this goal.

These articles add a significant contribution regarding public intervention linked to recycling activities. However this kind of static analysis omits the dynamic aspects of resource stocks mentioned above. A further analysis is needed to examine the arbitration between environmental externalities and resource depletion. For instance, extending the Acuff and Kaffine (2013) model (initially being an extension of the Palmer and Walls (1997) model) to a dynamic system.

An example of this can be found in Huhtala (1999). This is one of the few to analyze the optimal use of an exhaustible resource while considering issues of waste accumulation, resource depletion, and pollutant emissions at the same time. She describes the best arbitration of labor between recycling and primary production, and designs a fitting tax-subsidy scheme to achieve it under a balanced budget. This work is complemented by different recent studies on resource economics, with for instance Pittel et al. (2010) who model a decentralized economy with a recycling activity and highlight the market failure resulting from the absence of a market for waste, despite their economical value. They provide an optimum by setting up a market for waste and subsidizing recycling activities.

This dynamic approach is also the perspective of Di Vita (2001) and Di Vita (2007) who assess the possibility of an economic and welfare growth under a material constraint, thanks to investments in recycling. Sorensen (2017) also considers a recycling technology in a Ramsey model that alleviates externalities due to resource extraction and consumption. These articles share the use of optimal control theory, but propose different models to represent a circular economy. Additionally, Di Vita (2001) and Di Vita (2007) propose models that do not respect a material balance in the economy in contrast to the physical reality of the use of secondary materials. With these propositions, they arrive to the conclusion that recycling allows a stationary growth path. An alternative modeling approach is given by Boyce (2012) who chooses to specify a recycling stock separate from accumulated waste. He examines the

dynamic of this stock when there is perfect substitution between virgin and recycled material and he describes economic trade-offs between the two material to manufacture a final good.

The results of all these studies are in part driven by the substitutability characteristics of the production function. Models often use perfect substitution for virgin and recycled uses (Boyce, 2012; Hollander and Lasserre, 1988; Hoogmartens et al., 2018). While being a strong hypothesis, it remains relevant in our case of metal recycling with efficient processes (Villalba et al., 2002, 2004). While metal recycling has been a motivated topic for a long time (Ayres, 1997; Sigman, 1995), more recent concerns connected to technical changes and the energy transition have begun to be included. The essential place in modern technological applications taken by some materials (such as rare earth or lithium) also raises the question of how to bare the storage costs inherent to the industry as they are not evenly allocated worldwide (Rosendahl and Rubiano, 2019; Ba et al., 2020). As such, one could consider in-use material as a stock of resource (Batteries Europe, 2020).

1.3 A basic optimal recycling model

1.3.1 Setup of the model

We consider an industry that is managing a natural resource stock from which a final consumption good can be produced. We set an exogenous time-limit T to the management program, corresponding to the horizon of exploitation of the resource, before a shut down of the mine (Lappi and Ollikainen, 2019).

The industry can produce a quantity q of final good from two different inputs: the virgin resource and a recycled material, of relative quantities v and r. We assume perfect substitution between these two inputs, involving that $q(t) = v(t) + r(t)$ for any time t. We also pose that waste is a one-to-one co-product of the final consumption good. Thus, $q(t)$

also denotes the instantaneous flow of waste, before any potential recycling process.[2]

Consuming q units of final good provides a gross surplus $u(q)$ to the final user, where $u(.)$ follows the standard hypotheses of utility functions: of class $\mathcal{C}^2$, increasing ($u' > 0$), concave ($u'' < 0$) and verifying the Inada conditions, *i.e.* $\lim_{q\to 0^+} u'(q) = +\infty$ and $\lim_{q\to\infty} u'(q) = 0$. To account for the previously stated hypothesis of perfect substitution in the production function, we assume the utility drawn from the consumption of a good made out of virgin material is the same as that taken from the consumption of a good made out of recycled material. [3]

Input flows v and r come from two different primary sources, a virgin stock and a recycled stock. We denote the average delivery costs of these two resources as c_v and c_r, respectively. These average delivery costs include the extraction, production and transportation costs, all of which are assumed to be constant through time.

The industry is initially endowed with a stock $V_0 > 0$ of virgin resource. We suppose that recycling has never happened by the past, meaning that the initial stock of recycled material is zero: $R_0 = 0$. At any time t, the current levels of virgin and recycled stocks $V(t)$ and $R(t)$ are governed, respectively, by the following extraction processes:

$$\dot{V}(t) = -v(t) \quad , \quad V(0) = V_0 \tag{1.1}$$

$$\dot{R}(t) = -r(t) + \beta(t)q(t) \quad , \quad R(0) = 0\,, \tag{1.2}$$

where β, $\beta \in [0, 1]$, represents the endogenous recycling intensity (*i.e.* the share of production which is collected, sorted and incorporated into the recycled stock after use).[4] However,

[2] An alternative would be to consider that a constant share of the final good is destroyed by consumption, and then unrecoverable. In that case, consuming a quantity q of good would release a flow γq of waste, with $\gamma < 1$ being the waste content rate of the manufactured good. This would not change significantly the analysis but does not respect the physical law of material balance.

[3] Note that when focusing on the use of different resources, perfect substitution is a common simplifying assumption in the literature (Amigues et al., 2011; Boyce, 2012; Chakravorty et al., 2006, 2008).

[4] While β does not exactly fit the common definition, we will also refer to it as "recycling rate" throughout the rest of the chapter. Moreover, as we model recycled resource and waste as two different stocks, β must be a control variable in order to have an endogenous recycling. A similar approach is used by Boyce (2012)

note that recovering waste from the flow q and using it in production are two different actions that are only correlated by the cost arbitration. As a matter of fact, it can be optimal to use a recycled input in production while deciding not to redirect waste into the recycled stock ($\beta = 0$), as long as this stock is not empty.

Recovering material from waste is costly and this cost is added to the other private costs related to the recycled input. The recycling cost per unit of waste is given by the function $f(\beta)$, such that $f(0) = 0$, $f'(\beta) > 0$ with $\beta \in [0, 1[$, $f'(1) = +\infty$ and $f''(\beta) > 0$. Recycling the share β of the waste flow q thus costs $qf(\beta)$. The convexity of $f(.)$ means decreasing returns in the recycling technology, which reflects the technical difficulties to retrieve all material produced and consumed. This assumption, combined with $\lim_{\beta \to 1} f'(\beta) = +\infty$, implies that full recycling is not attainable. There should always be a part of the waste that cannot be recovered, at any cost. It can be due to fatal transformation of the original resource during the manufacturing process or due to a very dispersive use of the resource which renders it technologically impossible to retrieve all of the resource after consumption.[5]

The remaining share of final good which could not be collected and recycled yields a waste stream $(1 - \beta)q$ that accumulates.[6] We denote $W(t)$ as the resulting stock of waste at time t, with W_0 as the initial stock. This waste stock is reduced by a bio-decomposition, or another natural resorption mechanism, at a constant and exogenous rate α, $\alpha \in (0, 1)$. The dynamics of W are then given by:

$$\dot{W}(t) = (1 - \beta(t))q(t) - \alpha W(t), \quad W(0) = W_0\,. \tag{1.3}$$

as he considers a sorting cost and assigns to the social planner the choice of sorting waste or not (a binary decision). This choice of modeling two stocks allows to represent the material balance constraint in our economy.

[5]This hypothesis reflects the dispersive use of many materials as well as complex applications at mass production scales. In fact, thermodynamic limitations to recycling can involve important economic costs, or even impracticability, thus forbidding perfect recycling (Ayres, 1999; UNEP, 2011).

[6]Note that the share $(1 - \beta)$ of the waste stream that is not redirected to the recycled stock at instant t cannot be recycled later in the program.

Finally, waste accumulation generates local environmental degradation that harms the economy through a damage function $D(W)$, with $D'(W) > 0$ and $D''(W) \geq 0$. In order to simplify the analysis, we will consider a linear damage function, with a constant marginal damage denoted by $c_W > 0$.

1.3.2 Central planner program and optimal conditions

The objective of the central planner is to determine the resource extraction and recycling paths that maximize the discounted sum of net surpluses over the planning horizon.[7] Formally, denoting the time discount rate as $\rho > 0$, we want to solve the following problem:[8]

$$\max_{\{v,r,\beta\}} \int_0^T [u(v+r) - c_v v - c_r r - qf(\beta) - D(W)]\, \mathrm{e}^{-\rho t} dt\,, \tag{1.4}$$

subject to the dynamic constraints (1.1)-(1.3) and to the non-negativity constraints on v, r and β. The Hamiltonian can be written as:

$$\begin{aligned}\mathcal{H} = u(v+r) - c_v v - c_r r - qf(\beta) - D(W) + \lambda_V(-v) + \lambda_R[\beta(v+r) - r] \\ - \lambda_W[(1-\beta)(v+r) - \alpha W]\,,\end{aligned} \tag{1.5}$$

where λ_V, λ_R and $-\lambda_W$ are the co-state variables attached to the virgin stock, the recycled stock and the waste stock, respectively.[9]

[7] Remind that, here, the ending time T is exogenous and finite. An alternative would be to let T free. However, as the industry evolves in a finite world where none renewable resource is available and because of Inada conditions, it would be optimal to postpone T as much as possible: $T \to \infty$.

[8] In order to simplify notations, we will hide the time subscript whenever it is convenient and clear from the context.

[9] As waste accumulation generates a negative externality through a damage function, its shadow value is non-positive. For the purpose of simplifying the notations, we introduced a positive shadow cost by considering formally $-\lambda_W$ as co-state variable.

The optimal solution must satisfy the following first-order conditions:

$$u' \leq c_v + f(\beta) + \lambda_V - \beta\lambda_R + (1-\beta)\lambda_W, \quad (= \text{ if } v > 0) \tag{1.6a}$$

$$u' \leq c_r + f(\beta) + \lambda_R - \beta\lambda_R + (1-\beta)\lambda_W, \quad (= \text{ if } r > 0) \tag{1.6b}$$

$$qf'(\beta) \geq (\lambda_R + \lambda_W)q, \quad (= \text{ if } \beta > 0) \tag{1.6c}$$

$$\dot{\lambda}_V = \rho\lambda_V \tag{1.6d}$$

$$\dot{\lambda}_R = \rho\lambda_R \tag{1.6e}$$

$$\dot{\lambda}_W = (\rho+\alpha)\lambda_W - D'(W)\,, \tag{1.6f}$$

completed by the transversality conditions:

$$\lambda_V(T)V(T)e^{-\rho T} = \lambda_R(T)R(T)e^{-\rho T} = \lambda_W(T)W(T)e^{-\rho T} = 0\,. \tag{1.7}$$

Conditions (1.6a) and (1.6b) compare the gross marginal surplus of using each type of resource with its full marginal cost and state that it is optimal to use the resource when both are equal. The full marginal cost (FMC) of each input v or r is composed of: the delivery cost (c_v or c_r) of the input; the unitary cost $f(\beta)$ of recovering material from the waste stream; the scarcity rent (λ_V or λ_R) of the resource stock; the social marginal cost $-\beta\lambda_R$ of replenishing the recycled stock (a negative cost, this stock being a good for society); and, the shadow marginal cost $(1-\beta)\lambda_W$ of waste accumulation.

We can already determine the motivation of recycling as implemented in this resource model: recycling relaxes the resource constraint by replenishing the recycled stock and reducing waste accumulation. However, the choice of using recycled inputs in production here only relaxes pressure on virgin stock. Damage due to waste accumulation is not impacted, as both inputs have the same impact on waste after consumption. Waste reduction is controlled by the choice of the recycling rate β.

The optimal path for the recycling behavior is obtained from (1.6c) by comparing the

marginal cost $f'(\beta)$ per unit of final good with the marginal social gain $(\lambda_R+\lambda_W)$ of recycling. Indeed, for a given material stream q, increasing the share of recycled material by $\Delta\beta$ allows to increase the stock of recycled resource, whose marginal shadow value is λ_R, by quantity $\Delta\beta q$, and to reduce waste generation by this same unit, thus saving the marginal shadow cost λ_W of the waste stock.

Equations (1.6d) to (1.6f) rule the dynamics of the system. (1.6d) and (1.6e) illustrate, in particular, the Hotelling rule which state that the scarcity rents of the two resources grow at a rate equal to the discount rate ρ. Hence we can write: $\lambda_V(t) = \lambda_{V0}e^{\rho t}$ and $\lambda_R(t) = \lambda_{R0}e^{\rho t}$, with $\lambda_{V0} \equiv \lambda_V(0)$ and $\lambda_{R0} \equiv \lambda_R(0)$.

In (1.7), we can replace λ_V and λ_R by the above expressions, and V and R by the solutions of the differential equations (1.1) and (1.2), respectively. Assuming that neither resource stock is abundant, *i.e.* $\lambda_{V0} > 0$ and $\lambda_{R0} > 0$, the transversality condition implies that both stocks must be fully depleted at the terminal time T:

$$\int_0^T v(t)dt = V_0 \tag{1.8}$$

$$\int_0^T [\beta(t)v(t) - (1-\beta(t))r(t)]dt = 0\,. \tag{1.9}$$

Equation (1.6f), combined with the transversality condition (1.7), gives us the trajectory of the shadow cost of waste accumulation:[10]

$$\lambda_W(t) = \int_t^T D'(W)e^{-(\rho+\alpha)(s-t)}\mathrm{d}s\,. \tag{1.10}$$

We see here that the shadow cost of waste accumulation is equal to the intertemporal sum of the flows of marginal damages, discounted at rate $(\rho+\alpha)$ since waste is not only a flow, but

[10]Solutions of equations (1.3) and (1.6f) are, respectively: $W = e^{-\alpha t}[W_0 + \int_0^t (1-\beta)qe^{\alpha s}ds]$ and $\lambda_W = e^{(\rho+\alpha)t}[\lambda_{W0} - \int_0^t D'(W)e^{-(\rho+\alpha)s}ds]$. Replacing W and λ_W by these expressions into the transversality condition (1.7), we get: $\lambda_{W0} = \int_0^T D'(W)e^{-(\rho+\alpha)t}dt$. The initial value of the waste social cost is independently given by conditions on waste, only taking into account the dynamic characteristics α of the stock and its cost $D(W)$ for society.

also a stock that shrinks at rate α. With a constant marginal damage c_W, this expression can be simplified as follows:

$$\lambda_W(t) = \frac{c_W}{\rho + \alpha}\left[1 - \mathrm{e}^{-(\rho+\alpha)(T-t)}\right] . \tag{1.11}$$

This implies that λ_W is decreasing through time: $\dot{\lambda}_W(t) = -c_W \mathrm{e}^{-(\rho+\alpha)(T-t)} < 0$. In fact, this social cost accounts for the future damage due to waste. As the program approaches its end, the constraint has less impact.[11]

From first-order conditions (1.6a) and (1.6b), we can infer that the optimal solution is one or more consecutive phases of production from one input or the other, with, in parallel, a specific path for rate β ruled by equation (1.6c). Perfect substitution and fixed private costs do not allow simultaneous use of each input. Following the display of the model, we will be studying the switch from one input to the other, the needed initial conditions, and the optimal choice regarding the redirection of the waste flow to the recycled stock.

1.3.3 Arbitration on resources use

In order to get the optimal consumption path, we have to compare the full marginal costs of using each specific input (FMC_i, with $i \in \{v; r\}$), given by the right-hand side of (1.6a) and (1.6b). We define ΔFMC as their difference:

$$\Delta FMC \equiv FMC_v - FMC_r = c_v - c_r + (\lambda_{V0} - \lambda_{R0})\mathrm{e}^{\rho t} . \tag{1.12}$$

Examining the sign of this expression gives us arbitration at stakes when applying Herfindhal least-cost principle analysis (Herfindahl, 1967). It can be noted that neither recycling nor the waste stock come into play in the trade-off. In fact, products made out of

[11] We discuss in the appendix the case where the damage caused by the left-over waste stock $W(T)$ still harms the economy after the terminal date T.

recycled or virgin materials have the same impact on waste accumulation and the activity of waste recovery does not depend on the origin of the input flow. In this basic version of the model, the relative competitiveness of each type of resource is simply driven by the differential of resource extraction costs and by their relative scarcity. The relative magnitude of these two effects can lead to a switch between inputs, mathematically materialized by the change in sign of ΔFMC before the end of the program. Formally, if $\Delta FMC < 0$ then we produce from the virgin input, and if $\Delta FMC > 0$ then we produce from the recycled input.

From this analysis, Proposition 1 below describes the main characteristics of the optimal sequence of input use.

Proposition 1. *Regardless of the recycling rate $\beta(t)$, the optimal path is such that:*

1. *It must start with a phase of production from extraction of virgin resource in all scenarios.*
2. *There can be at most one switch of input during the program. Let us call this time of switch $\tilde{T}$, whose existence within $[0;T]$ is not guaranteed. Then, if it exists within $[0;T]$, $\exists! \tilde{T} > 0$ such that $\Delta FMC(t) < 0$ for any $t < \tilde{T}$ and $\Delta FMC(\tilde{T}) = 0$.*
3. *When such a switching time $\tilde{T}$ exists, it is defined by:*

$$e^{\rho \tilde{T}} = \frac{c_r - c_v}{\lambda_{V0} - \lambda_{R0}}. \tag{1.13}$$

Proof. (1.) For any $t \in [0, \tilde{T}[$, if $v(t) = 0$ then, from (1.2), $R(\tilde{T}) = -\int_0^{\tilde{T}} (1-\beta) r ds \leq 0$ which is not possible. (2.) Differentiating (1.12) with respect to time yields $\Delta \dot{FMC} = \rho(\lambda_{V0} - \lambda_{R0}) e^{\rho t}$. As this expression is constant in sign, ΔFMC is monotonic and there can be at most one switch of inputs during the optimal program of the model. (3.) This is simply the solution of $\Delta FMC(t) = 0$. □

The first result highlights the need for initial conventional production: recycling can only be achieved when there has been enough extracted input consumed, and then collected

from the waste stream. Also, by the assumption of the recycled stock being empty at the beginning ($R_0 = 0$), there must always be a first phase of virgin production to fill the recycled stock before being able to use it. Thus, ΔFMC must be initially negative, which requires $\lambda_{V0} - \lambda_{R0} < c_r - c_v$. The second result implies that, if it exists, the time of the switch $\tilde{T}$ is the solution of the following equation $c_v + \lambda_V(\tilde{T}) = c_r + \lambda_R(\tilde{T})$. A single instant solution to $\Delta FMC = 0$ as given by (1.13) also means that a simultaneous use of virgin and recycled resource cannot occur. Different scenarios can occur, depending on the sign of $(c_r - c_v)$. Proposition 2 below summarizes them.

Proposition 2. *Regardless of the recycling rate path $\beta(t)$, we can summarize the different optimal scenarios to the following ones:*

1. *If extracting input from the recycled stock is costlier than from the virgin stock ($c_r > c_v$):*
   (a) *We only use the virgin resource if this resource is relatively more abundant than the recycled one ($\lambda_{V0} < \lambda_{R0}$);*
   (b) *We use the virgin resource up to time $\tilde{T}$, with $\tilde{T} \leq T$, and then the recycled resource if the later is relatively more abundant than the former ($\lambda_{V0} > \lambda_{R0}$);*
2. *If extracting input from the virgin stock is costlier than from the recycled stock ($c_r < c_v$), we use the virgin resource up to time $\tilde{T}$ and then the recycled resource. In this case, the virgin resource must also be relatively more abundant than the recycled one ($\lambda_{V0} < \lambda_{R0}$).*

Proof. The different optimal paths can be drawn from the positivity conditions for ΔFMC. Moreover, in case 1(b), as the transversality condition guarantees that the virgin resource must be exhausted at the terminal date T, the switch to the recycled resource occurs no later than T: $\tilde{T} \leq T$. □

In scenario 1(a), using the recycled resource does not show any interest as the virgin resource is both less expensive and relatively more abundant than the recycled one. In

scenario 1(b), the virgin stock exhibits relative scarcity that can be relaxed by using the recycled resource, despite its higher extraction cost. Last, scenario 2 illustrates the case where the recycled resource is cheaper than the virgin one. In this case, the only way to justify an initial phase of virgin resource use, which is required to provide recycled waste, is to assume that the virgin stock is relatively abundant.

As we separate marginal delivery cost c_r and waste recovery cost $f(\beta)$, we do not assume at first any relationship between delivery costs c_i. However, the difference in maturity between these two materials can be observed as the recycling branch is usually not favored when social costs are not internalized, leading to the reasonable hypothesis $c_r + f(\beta) > c_v$ [12]. However we do not assume anything more as differences between sectors could lead to recycling costs relying on delivery ($c_r >> f(\beta)$ thus $c_r > c_v$) or on collection ($f(\beta) >> c_r$ thus potentially $c_v > c_r$).

At the end, the optimal path of resource consumption is characterized by the triplet of endogenous elements $\left\{\lambda_{V0}, \lambda_{R0}, \tilde{T}\right\}$, which is determined as the solution of the three equations-system (1.8), (1.9) and (1.13). For the remaining of the section, we will consider the most interesting case where both resources are successively used, which means that the switching time $\tilde{T}$ exists within $]0, T[$. Hence, λ_{V0} and λ_{R0} must satisfy the following additional condition:

$$1 < \frac{c_r - c_v}{\lambda_{V0} - \lambda_{R0}} < \mathrm{e}^{\rho T} \,. \tag{1.14}$$

1.3.4 Recovering waste

The optimal path of the recycling rate β is determined from equation (1.6c). In order to simplify the analysis, we define the function $\Phi_\beta \equiv \lambda_R + \lambda_W$ as the marginal social gain of the recycling effort (both in avoiding accumulation of waste and replenishing the stock of

[12] Di Vita (2007) precises that an industry of secondary materials with the same properties as virgin inputs would bear very high cost, not sustainable with current market prices. With the exemple of lithium, Rosendahl and Rubiano (2019) use higher initial costs for extraction than recycling.

recycled resource). Recycling intensity results from the arbitration between its marginal cost and this marginal gain. Formally, $\beta = 0$ when $\Phi_\beta \leq f'(0)$, and $\beta \in]0;1[$ such that $\Phi_\beta = f'(\beta)$ when $\Phi_\beta > f'(0)$. Remark that there can be recycling $\beta > 0$ while the virgin resource is not used: recycling only alleviates the pressure on waste accumulation in this case.

The optimal recycling program can be described by having three material moments with characteristics detailed in proposition 3. The principle material moment is time $\hat{T}_\beta$ which represents the time of maximal recycling rate $\beta(t)$. This moment is accompanied by two other material moments $\underline{T}_\beta$ and $\overline{T}_\beta$ which represent the start and finish of recycling, respectively. Formally we specify $\underline{T}_\beta$ and $\overline{T}_\beta$ such that $\beta(t) = 0$ if $t < \underline{T}_\beta$ or $t > \overline{T}_\beta$. Note that we restrained our analysis to the case where both V and R are extracted, meaning that there must be a phase of recycling during the program.

Proposition 3. *Regardless of the resource extraction regime:*

1. *There can be zero, one or two phases where the recycling rate β saturates and is zero. These phases can either be:*
   - *(a) No saturation phase (we always recycle);*
   - *(b) One phase at the beginning or the end of the program, such that $\beta(t) = 0$ when $t \in [0;\underline{T}_\beta]$ or $t \in [\overline{T}_\beta;T]$, where the values of times $\underline{T}_\beta$ or $\overline{T}_\beta$ are the solution in $[0;T]$ of equation $\Phi_\beta(t) = f'(0)$;*
   - *(c) Two phases at the beginning and the end of the program, such that $\beta(t) = 0$ when $t \in [0;\underline{T}_\beta]$ and $t \in [\overline{T}_\beta;T]$, where the values of times $\underline{T}_\beta$ and $\overline{T}_\beta$ are the solutions in $[0;T]$ of equation $\Phi_\beta(t) = f'(0)$ with $0 < \underline{T}_\beta < \overline{T}_\beta < T$.*
2. *When the recycling rate $\beta(t)$ is not always zero, it reaches a maximum at the time $\hat{T}_\beta$*

defined as follows:

$$\hat{T}_\beta \begin{cases} = 0 & , \textit{ if } \mathrm{e}^{(\rho+\alpha)T} \leq \frac{c_W}{\rho\lambda_{R0}} \\ = T & , \textit{ if } \frac{c_W}{\rho\lambda_{R0}} \leq \mathrm{e}^{\rho T} \\ \in]0,T[\textit{ and s.t. } \mathrm{e}^{\alpha\hat{T}_\beta} = \frac{\rho\lambda_{R0}}{c_W}\mathrm{e}^{(\rho+\alpha)T} & , \textit{ if } \mathrm{e}^{\rho T} < \frac{c_W}{\rho\lambda_{R0}} < \mathrm{e}^{(\rho+\alpha)T}\,. \end{cases} \tag{1.15}$$

Proof. As the cost function associated to the recycling rate $f(\beta)$ is increasing and convex: from (1.6c), the proof of proposition 3 comes from the study of the shape of function Φ_β, illustrated by Figure 1.1(a). We can study the dynamics of β by studying the evolution of Φ_β:[13]

$$\Phi_\beta(t) = \lambda_{R0}\mathrm{e}^{\rho t} + \frac{c_W}{\rho+\alpha}\left(1 - \mathrm{e}^{-(\rho+\alpha)(T-t)}\right)\,.$$

Function $\Phi_\beta(t)$ defined on $\mathbb{R}^+$ is proved to be increasing from $\Phi_\beta(0) = \lambda_{R0} + \frac{c_W}{(\rho+\alpha)}[1-e^{-(\rho+\alpha)T}]$ to its maximum $\Phi_\beta(\hat{T}_\beta)$, and then declining, with $\lim_{t\to\infty}\Phi_\beta(t) = -\infty$.

(1.) We can see that the equation $\Phi_\beta(t) = f'(0)$ has zero solution when $\forall t \in \mathbb{R}^+, \Phi_\beta(t) < f'(0)$, one solution when $f'(0) \leq \Phi_\beta(0)$ and two solutions otherwise.[14] We can determine $\underline{T}_\beta$ and $\overline{T}_\beta$ as the results of this equation, when they exist in $[0;T]$.

(2.) The existence of $\hat{T}_\beta$ in the boundaries of the program $[0;T]$, as illustrated by Figure 1.1(b), directly follows the study of the equation $\Phi'_\beta(t) = 0$ and gives the results of the second item of Proposition 3. □

It follows from this analysis that as the optimal path of $\beta(t)$ is defined by the path of function $\Phi_\beta(t)$, the optimal recycling rate is inverted U-shaped in t when positive. Also, the permanent arbitration with the marginal social benefit of recycling can induce phases with

[13] For the purpose of this analysis, we extend the definition of $\Phi_\beta(t)$ to $\mathbb{R}^+$, and then discuss the existence of solutions in $[0,T]$.

[14] We intentionally omit the specific case when there is a tangential solution for the equation, such as $\max_{\mathbb{R}^+}\Phi_\beta = f'(0)$.

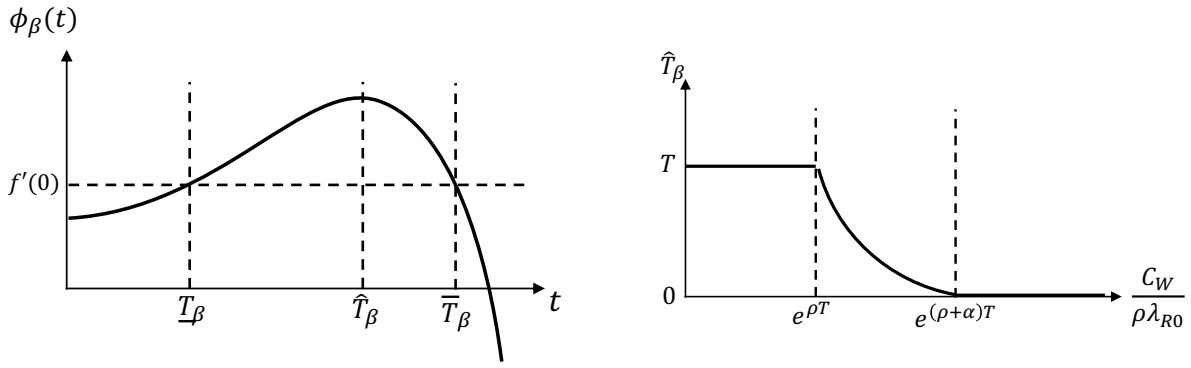


Figure 1.1: (a) Saturation of β and (b) Maximum time $\hat{T}_\beta$

$\beta = 0$. When this phase comes first, it means that the resource constraint on stock R is not high enough to start recycling. The social planner only initiates it after $\underline{T}_\beta$. On the other hand, there can also be a final phase without recycling as the shrinking social cost of waste accumulation makes it no more optimal. This is visible from the evolution of maximum time exposed in (1.15). Given the inverted U-shape of $\beta(t)$, a maximum instant $\hat{T}_\beta$ is given by the comparison between the sum of discounted marginal waste damage $\frac{c_W}{\rho} = \int_0^\infty c_W e^{-\rho t} dt$, and the initial scarcity of the recycled resource λ_{R0}. For high damage wastes (or a low constraint on the recycled resource), the maximum of recycling occurs at the beginning.

In section 1.3.5, we will consider the most interesting case where the recycling rate $\beta(t)$ reaches a maximum at instant $\hat{T}_\beta$, corresponding to the third case of (1.15), where $0 < \hat{T}_\beta < T$.

1.3.5 Dynamics of resource flows

The objective is now to superimpose the respective dynamics of resource use and recycling that have been previously analyzed separately. For that we consider a phase during which $v > 0$ (and $r = 0$) or $r > 0$ (and $v = 0$). We look at the dynamics of $v(t)$ and $r(t)$ by examining the time derivatives of conditions (1.6a) and (1.6b). Using (1.11), for any value

of β, we obtain:

$$\dot{FMC}_v = \left[\rho(\lambda_{V0} - \beta\lambda_{R0}) - (1-\beta)c_W \mathrm{e}^{-(\rho+\alpha)T}\mathrm{e}^{\alpha t}\right]\mathrm{e}^{\rho t}, \tag{1.16}$$

$$\dot{FMC}_r = (1-\beta)\left[\rho\lambda_{R0} - c_W \mathrm{e}^{-(\rho+\alpha)T}\mathrm{e}^{\alpha t}\right]\mathrm{e}^{\rho t}. \tag{1.17}$$

Remark that, defining $\Phi_v(t) \equiv [1-\beta(t)]\mathrm{e}^{\alpha t} + \beta(t)\mathrm{e}^{\alpha \hat{T}_\beta}$, with $\hat{T}_\beta$ such that $e^{\alpha \hat{T}_\beta} = \rho\lambda_{R0}e^{(\rho+\alpha)T}/c_W$, we can rewrite (1.16) as follows:

$$\dot{FMC}_v = \left[\frac{\rho\lambda_{V0}}{c_W}\mathrm{e}^{(\rho+\alpha)T} - \Phi_v\right] c_W \mathrm{e}^{\rho t}\mathrm{e}^{-(\rho+\alpha)T}. \tag{1.18}$$

Proposition 4. *The full marginal cost FMC_v (resp. FMC_r) of producing with the virgin resource (resp. recycled) can reach a maximum at time $\hat{T}_v$ (resp. $\hat{T}_r$).*

1. *A maximum for FMC_v exists if $\Phi_v(0) < \frac{\rho\lambda_{V0}}{c_W}\mathrm{e}^{(\rho+\alpha)T} < \Phi_v(T)$. It is determined by:*

$$\Phi_v(\hat{T}_v) = \frac{\rho\lambda_{V0}}{c_W}\mathrm{e}^{(\rho+\alpha)T}.$$

2. *If FMC_r is non-monotonous, it reaches a maximum at time $\hat{T}_r = \hat{T}_\beta$, with $\hat{T}_\beta$ being characterized by (1.15).*
3. *If the $\hat{T}$'s exist, then $\hat{T}_v < \hat{T}_r$ (resp. $>$) if $\lambda_{V0} < \lambda_{R0}$ (resp. $>$).*

Proof. (1.) The full marginal cost FMC_v will reach an extremum when $\dot{FMC}_v$ as given by (1.18) equals 0. In addition, Φ_v is strictly increasing: $\Phi_v'(t) = \left(\mathrm{e}^{\alpha\hat{T}_\beta} - \mathrm{e}^{\alpha t}\right)\dot{\beta} + (1-\beta)\mathrm{e}^{\alpha t} > 0$ for any t, as $\hat{T}_\beta$ is the maximum instant of β. Then, if $\dot{FMC}_v = 0$, the extremum of FMC_v is a maximum. (2.) From (1.17), and given that $\beta(t) < 1\ \forall t$, it comes immediately that $\dot{FMC}_r(t) = 0 \Leftrightarrow t = \hat{T}_\beta$. Since $\rho\lambda_{R0} - c_W\mathrm{e}^{-(\rho+\alpha)T}\mathrm{e}^{\rho t}$ is decreasing in t, the extremum is a maximum. (3.) From the characterization of $\hat{T}_v$ given by the first part of the proof, we can

write: $e^{\alpha\hat{T}_v} - e^{\alpha\hat{T}_\beta} = \left(\frac{1}{1-\beta(\hat{T}_v)}\right)\left[\frac{\rho\lambda_{V0}}{c_w}e^{(\rho+\alpha)T} - \beta(\hat{T}_v)e^{\alpha\hat{T}_\beta}\right] - e^{\alpha\hat{T}_\beta} = \left(\frac{1}{1-\beta(\hat{T}_v)}\right)\frac{\rho(\lambda_{V0}-\lambda_{R0})}{c_W}e^{(\rho+\alpha)T}$.
Hence, $\text{sign}(\hat{T}_v - \hat{T}_\beta) = \text{sign}(\lambda_{V0} - \lambda_{R0})$. □

Analyzing possible outcomes for FMC curves and considering their characteristics described in Propositions 1, 2 and 4 allows us to determine the typical trajectories of the gross marginal surplus $u'(q)$. From (1.6a) and (1.6b), we know that $u'(q) = FMC_v(t) \leq FMC_r(t)$ for $t \leq \tilde{T}$ and $u'(q) = FMC_r(t) < FMC_v(t)$ for $t > \tilde{T}$. Given the possible shapes of the FMC's as described in Proposition 4, the time dependent marginal gross surplus curve can thus be either always increasing, or always declining, or an inverted U-shape. In this last case, the maximum of $u'(q)$ can be attained before the switching time $\tilde{T}$, after it, or precisely at that time.

Table 2 summarizes all these possible cases depending on whether the recycled material is more expensive (second column) or cheaper (third column) than the virgin resource, and characterizes the conditions on the dates $\hat{T}_v$ and $\hat{T}_r$ for their occurrence. Notice that, as we focus on the scenario of a successive use of both resources, *i.e.* such that $\tilde{T} \in (0, T)$, if $c_r - c_v > 0$ (resp. $<$), then we must have $\lambda_{V0} - \lambda_{R0} > 0$ (resp. $<$) from condition (1.14), which implies $\hat{T}_v > \hat{T}_r$ (resp. $<$) from Proposition 4.3. This last result allows for reducing the number of possible trajectories.

By concavity of u, the dynamics of resource extraction q, either from the virgin stock or from the recycled stock, is the inverse of those of the marginal surplus as characterized in Table 2. When the gross marginal surplus curve is increasing, the Hotelling effect prevails due to the scarcity of both resources (*i.e.* λ_{V0} and/or λ_{R0} large), and the extraction path must be decreasing. However, as previously explained, this scarcity effect on the resource stocks can be counterbalanced by a recycling effect aiming at increasing the resource use in order to provide raw materials to be recycled, which corresponds to phases where $u'(q)$ is decreasing through time.

Shape of $u'(q)$	If $c_v < c_r$ (and $\hat{T}_r < \hat{T}_v$)	If $c_r < c_v$ (and $\hat{T}_v < \hat{T}_r$)
Always increasing	$T \leq \hat{T}_r$	$\tilde{T} \leq \hat{T}_v$ and $T \leq \hat{T}_r$
Increasing and then decreasing with a maximum attained after $\tilde{T}$	$\tilde{T} < \hat{T}_r < T$	$\tilde{T} < \hat{T}_v < \hat{T}_r < T$
Increasing and then decreasing with a maximum attained at $\tilde{T}$	$\hat{T}_r \leq \tilde{T} \leq \hat{T}_v$	impossible
Increasing and then decreasing with a maximum attained before $\tilde{T}$	$0 < \hat{T}_v < \tilde{T}$	$0 < \hat{T}_v < \hat{T}_r < \tilde{T}$
Always decreasing	$\hat{T}_v \leq 0$	$\hat{T}_v \leq 0$ and $\hat{T}_r < \tilde{T}$

Table 1.2: Possible shapes of the gross marginal surplus and conditions of occurrence

As the trajectory of u' is possibly inverted U-shaped, the total flow of material can increase at the end of the program (after time $\hat{T}_i$). This constitutes a catch-up phase: the lower pressure on waste (as the end of the program approaches) and a higher appreciation of the replenishment of the recycled stock counterbalance depletion of the virgin stock. The constraint $\hat{T}_\beta = \hat{T}_r < \hat{T}_v$ follows when $\hat{T}_\beta$ comes before the end of the program and the scarcity of V is more prominent than R. A catch-up phase with an increasing production comes after the start of a phase when the interest in recovering waste is lowered $\dot{\beta} < 0$, meaning that the social cost of waste accumulation declines more than the social cost of the resource.

1.4 Accounting for climate change impacts

1.4.1 Optimal program with greenhouse gas emissions

We now take into account another environmental externality that comes from GHG emissions due to both virgin and recycled resource uses. We assume that input flows v and r

respectively contribute at rates δ_v and δ_r to these emissions.[15] Then, the cumulative GHG emissions $E(t)$ at any time t evolves as:

$$\dot{E}(t) = \delta_v v(t) + \delta_r r(t), \quad E(0) = 0\,. \tag{1.19}$$

Contrary to waste accumulation which creates local damages, GHG emissions are a global externality which do not directly harm the industry. However, for this last externality to be binding in our model, we define a GHG stabilization cap beyond which damages are supposed to be too high to be supported on a global scale. Such cumulative emission targets are set by international environmental efforts in order to curb global warming. We suppose in our model that this global target has been divided between different countries, leading to national emissions objectives (or budgets) that can be dispatched between industries. We set this sectoral carbon budget to $\bar{E}$ so that, at any time t, the cumulative emissions cannot be higher:

$$E(t) \leq \bar{E}, \forall t\,. \tag{1.20}$$

As we consider neither carbon-free alternative resources nor processes of reducing cumulative emissions, we can no longer use inputs, virgin or recycled, once the cap is reached. With the Inada conditions, it forces $q = 0$ as late as possible, simplifying hypothesis (1.20) to a final state condition. Hence, the exogenous time-limit T of the model also corresponds to climate objectives of the industrial sector and its associated emissions ceiling, involving the constraint:

$$E(T) \leq \bar{E}\,. \tag{1.21}$$

[15] We suppose for the moment that the contribution of the recycling activity represented by rate β to total emissions is included in rate δ_r given for the use of the recycled input. In appendix, we show how relaxing this assumption modify the results.

Compared to the set of optimal conditions obtained in the basic model without GHG emissions, only (1.6a) and (1.6b) are modified as shown in (1.22a) and (1.22b) and an additional condition (1.22c) on GHG dynamics is introduced. In this condition, $-\lambda_E$ is the co-state variable associated to the dynamics of GHG emissions:

$$u' \leq c_v + f(\beta) + \lambda_V + \delta_v \lambda_E - \beta\lambda_R + (1-\beta)\lambda_W, \quad (= \text{ if } v > 0) \tag{1.22a}$$

$$u' \leq c_r + f(\beta) + \lambda_R + \delta_r \lambda_E - \beta\lambda_R + (1-\beta)\lambda_W, \quad (= \text{ if } r > 0) \tag{1.22b}$$

$$\dot{\lambda}_E = \rho\lambda_E \Leftrightarrow \lambda_E = \lambda_{E0} e^{\rho t}, \tag{1.22c}$$

Conditions (1.6c)-(1.6f) are unchanged. The transversality condition (1.7) also remains true but it must be expanded to include:

$$\lambda_E(T)(\bar{E} - E(T))e^{-\rho T} = 0. \tag{1.23}$$

The new FMC expression for each input now also includes the social marginal cost λ_E of cumulative emissions weighted by the carbon intensity of each input, δ_v and δ_r. We can see that beyond relaxing the resource constraint and reducing waste accumulation, recycling can also be motivated by potentially reducing GHG emissions when the virgin input is more polluting than the recycled one ($\delta_v > \delta_r$).

1.4.2 Relative scarcity of the stocks and budget

Transversality conditions (1.7) and (1.23) state that at the final time T, stock V (resp. R and $\bar{E} - E$) is either a scarce resource, with $V(T) = 0$ (resp. $R(T) = 0$ and $E(T) = \bar{E}$), or it is an abundant resource and its shadow value is always zero, and thus $\lambda_{V0} = 0$ (resp. $\lambda_{R0} = 0$ and $\lambda_{E0} = 0$). From (1.1), (1.2) and (1.19), final stocks are linked with the following

equation:

$$E(T) = \delta_v(V_0 - V(T)) + \delta_r \left(\int_0^T \beta(s)q(s)\mathrm{d}s - R(T) \right) . \tag{1.24}$$

Therefore, a full exhaustion of all stocks means that the exogenous decision on a carbon budget perfectly meets the emissions potential of both resources and accounts for the optimal trajectory of recycling. This politically-hard-to-design scenario can then be ruled out.

For the problem to be meaningful, we place our model in a situation where the carbon budget is set to answer a pressing climate constraint, where there should be significant changes of behaviours in the economy. In other words, we assume that the carbon budget $\bar{E}$ to be small enough, as compared with the initial endowment in carbon-emitting resources, so that it will be exhausted no later than time T. As there is no carbon-free option available in this economy, it is then optimal to postpone as much as possible the exhaustion of the budget, $E(T) = \bar{E}$, which implies a strictly positive shadow cost of emissions: $\forall t,\ \lambda_E(t) = \lambda_{E0}e^{\rho t} > 0$.

However, exhausting the virgin resource would mean $\delta_v V_0 \leq \bar{E}$ from equations (1.1) and (1.19). This situation is similar to what we would call a "business as usual" scenario, where imposing a carbon budget has no impact on resource exhaustion on a finite period. For this reason we should consider the virgin resource to be relatively abundant, meaning that $V(T) > 0$ and, from (1.7), that $\lambda_V(t) = 0\ \forall t$. These observations lead us to the following assumption:

Assumption 1. *The terminal state of the model with climate change is such that the carbon budget is saturated $E(T) = \bar{E}$ and the virgin resource is abundant $V(T) > 0$.*

1.4.3 Arbitration on resources use

FMCs and their difference are now expressed with the social cost of GHG and without the scarcity rent on the virgin resource:

$$FMC_v = c_v - \beta\lambda_R + \delta_v\lambda_E + (1-\beta)\lambda_W + f(\beta) \tag{1.25}$$

$$FMC_r = c_r + (1-\beta)\lambda_R + \delta_r\lambda_E + (1-\beta)\lambda_W + f(\beta) \tag{1.26}$$

$$\Delta FMC \equiv FMC_v - FMC_r = -(c_r - c_v) + [(\delta_v - \delta_r)\lambda_{E0} - \lambda_{R0}]\mathrm{e}^{\rho t}\,. \tag{1.27}$$

The change of inputs now also depends on the GHG emissions difference. Proposition 1 remains overall the same, except for a new eventual time of input switch which is now defined by:

$$\mathrm{e}^{\rho\tilde{T}} = \frac{c_r - c_v}{(\delta_v - \delta_r)\lambda_{E0} - \lambda_{R0}} \tag{1.28}$$

Different scenarios depending on the costs and emissions rates of resources exist and are summarized with Proposition 5:

Proposition 5. *Regardless of the recycling rate path $\beta(t)$, we can summarize the different optimal scenarios under a carbon budget to the following:*

1. *If extracting input from the recycled stock is costlier than from the virgin stock ($c_r > c_v$):*
   - *(a) If the use of the virgin resource emits less than the recycled one ($\delta_v < \delta_r$), or if the scarcity of the recycled resource weighs more than the emissions difference, we use only the virgin resource;*
   - *(b) If the virgin resource is more polluting ($\delta_v > \delta_r$), at a level enabling $\tilde{T}$ to exist as defined in (1.28) (i.e. $(\delta_v - \delta_r)\lambda_{E0} > \lambda_{R0}$) and provided that $\tilde{T} \leq T$, we use the virgin resource up to time $\tilde{T}$;*
2. *If extracting input from the virgin stock is costlier than from the recycled stock ($c_r < c_v$),*

we use the virgin resource up to time $\tilde{T}$ and then the recycled resource. In this case, the emissions difference must also weigh less than the scarcity of the recycled resource ($(\delta_v - \delta_r)\lambda_{E0} < \lambda_{R0}$).

Proof. The different optimal paths can be drawn from the positivity conditions for ΔFMC. □

The arbitration on resources use is significantly influenced by the carbon constraint, as the scarcity of the GHG budget also leads to a switch in inputs, while the virgin resource remains abundant. However, the reasoning on recycling rate β developed earlier remains valid, as equation (1.6c) does not include the social cost of carbon. Last, as previously mentioned, we restrict the analysis to the case of an interior switch of inputs, *i.e.* $\tilde{T} \in (0; T)$, and when recycling is non-monotonous, *i.e.* $0 < \hat{T}_\beta < T$.

1.4.4 Dynamics of resource flows

While no closed-form expression for the consumption trajectories of resources can be determined analytically in this program, first-order conditions can give us some insights on the different phases of production (as seen previously) and the fluctuations of the material flows. For that we consider a phase during which $v > 0$ (and $r = 0$) or $r > 0$ (and $v = 0$). We look at the dynamics of $v(t)$ and $r(t)$ by examining the time derivatives of conditions (1.6a) and (1.6b). Using (1.11), we obtain for any value of $\beta(t)$:

$$F\dot{M}C_v = \left[\rho\delta_v\lambda_{E0} - \rho\beta\lambda_{R0} - (1-\beta)c_W e^{-(\rho+\alpha)T}e^{\alpha t}\right]e^{\rho t}, \tag{1.29}$$

$$F\dot{M}C_r = \left[\rho\delta_r\lambda_{E0} + \rho(1-\beta)\lambda_{R0} - (1-\beta)c_W e^{-(\rho+\alpha)T}e^{\alpha t}\right]e^{\rho t}. \tag{1.30}$$

For this we define functions Φ_v and Φ_r as follow:

$$\Phi_v(t) \equiv [1-\beta(t)]\mathrm{e}^{\alpha t} + \beta(t)\mathrm{e}^{\alpha \hat{T}_\beta}\,, \tag{1.31}$$

$$\Phi_r(t) \equiv [1-\beta(t)]\mathrm{e}^{\alpha t} - [1-\beta(t)]\mathrm{e}^{\alpha \hat{T}_\beta}\,, \tag{1.32}$$

where, from (1.15), $\hat{T}_\beta$ is such that $e^{\alpha \hat{T}_\beta} = \frac{\rho \lambda_{R0} e^{(\rho+\alpha)T}}{c_W}$ (which is, if it exists in $]0;T[$, the instant when recycling reaches its maximum).

Proposition 6. *The full marginal cost FMC_v (resp. FMC_r) of producing with the virgin resource (resp. recycled) can reach a maximum at time $\hat{T}_v$ (resp. $\hat{T}_r$).*

1. *A maximum exists if $\Phi_v(0) < \frac{\rho\delta_v\lambda_{E0}}{c_W}\mathrm{e}^{(\rho+\alpha)T} < \Phi_v(T)$ (resp. $\Phi_r(0) < \frac{\rho\delta_r\lambda_{E0}}{c_W}\mathrm{e}^{(\rho+\alpha)T} < \Phi_r(T)$). These instants correspond to maxima of the FMCs and are determined by:*

$$\Phi_v(\hat{T}_v) = \frac{\rho\delta_v\lambda_{E0}}{c_W}\mathrm{e}^{(\rho+\alpha)T} \text{ and } \Phi_r(\hat{T}_r) = \frac{\rho\delta_r\lambda_{E0}}{c_W}\mathrm{e}^{(\rho+\alpha)T}.$$

2. *If true, we have:*

$$\hat{T}_\beta < \hat{T}_r < \hat{T}_v \text{ if } c_v < c_r\,,$$
$$\hat{T}_\beta < \hat{T}_r \text{ and } \hat{T}_v < \hat{T}_r \text{ if } c_r < c_v\,.$$

Proof. (1.) The full marginal cost FMC_v will reach an extremum when $\dot{FMC}_v = 0$. Using (1.29) and (1.31), this condition becomes: $\dot{FMC}_v = \left[\frac{\rho\delta_v\lambda_{E0}}{c_W}\mathrm{e}^{(\rho+\alpha)T} - \Phi_v\right] c_W \mathrm{e}^{\rho t}\mathrm{e}^{-(\rho+\alpha)T} = 0$. In addition, Φ_v is strictly increasing: $\Phi_v'(t) = \left(e^{\alpha\hat{T}_\beta} - e^{\alpha t}\right)\dot{\beta} + (1-\beta)e^{\alpha t} > 0$ for any t, as $\hat{T}_\beta$ is the maximum instant of β. Then, if $\dot{FMC}_v = 0$, the extremum of FMC_v is a maximum. The same reasoning is applied to FMC_r, with $\dot{FMC}_r = \left[\frac{\rho\delta_r\lambda_{E0}}{c_W}\mathrm{e}^{(\rho+\alpha)T} - \Phi_r\right] c_W \mathrm{e}^{\rho t}\mathrm{e}^{-(\rho+\alpha)T}$.

(2.) First, from the expression of Φ_r, condition $\Phi_r(\hat{T}_r) = \frac{\rho\delta_r\lambda_{E0}}{c_W}\mathrm{e}^{(\rho+\alpha)T}$ implies that $\hat{T}_\beta < \hat{T}_r$.

Next, when $c_v < c_r$, from (1.27), we can write $\Delta \dot{FMC} = \rho[\Delta FMC + (c_r - c_v)]$. As $\tilde{T}$ exists, we know that there is a second phase during which the recycled input is used, meaning $\Delta FMC > 0$ for $t > \tilde{T}$, which involves $\Delta \dot{FMC} > 0$. This result implies $[(\delta_v - \delta_r)\lambda_{E0} - \lambda_{R0}] > 0$. As it is independent of time, we can generalize with $\Delta \dot{FMC} > 0$, $\forall t$. Hence $\dot{FMC}_v > \dot{FMC}_r$. By rewriting expressions of $\dot{FMC}_i$ with Φ_i ($i \in \{v; r\}$), which are proved to be increasing functions of time, thus $\dot{FMC}_i > 0$ before $\hat{T}_i$ and $\dot{FMC}_i < 0$ after. This gives us that $\hat{T}_r < \hat{T}_v$.

Else, when $c_r < c_v$, expression (1.28) gives us that $[(\delta_v - \delta_r)\lambda_{E0} - \lambda_{R0}] < 0$. When used in the expression of $\Delta \dot{FMC}$, it leads to $\dot{FMC}_r > \dot{FMC}_v$, hence $\hat{T}_v < \hat{T}_r$. □

From Proposition 6, we can induce the same type of results on the dynamics of the gross marginal surplus and the dynamics of the resource extraction as those contained in Table 2. Whatever the relative cost of each input, we obtain a time trajectory of the resource extraction that is either strictly increasing, or strictly decreasing or U-shaped. Hence, as in the basic model, a catch-up phase can occur at the end of the program, during which the total flow of material increases. However, this phase is not driven anymore by the relative scarcity of each resource stocks (the virgin stock is assumed to be abundant), but rather by the relative carbon footprint of each resource and by the carbon budget constraint on cumulative emissions with respect to the environmental damage due to waste accumulation.

1.5 Decentralization and policy implications

In order to analyze environmental policy tools used to promote recycling and the reduction of carbon emissions, there is a need to discuss a decentralized version of our model. To decentralize the model, transfer functions for the different actors in the economy are introduced. These functions correct for the externalities previously developed in this chapter.

The basis of the decentralized model is the extended model developed in Section 1.4 that includes GHG emissions, as it offers a richer discussion on combined externalities and allows for the observation of different mixes of policies.

As a basic structure, we assume that the economy is composed of four agents: the final consumer, the producer of the manufactured good, the virgin resource sector, and the recycled resource sector which manages both the recycling activity and the exploitation of the recycled stock.[16] The property rights of each resource stocks are correctly defined so that each extracting sector is owner of its reserves. These agents can take actions on four markets, assumed to be perfectly competitive: the virgin resource market (price p_v), the recycled resource market (price p_r), the manufactured good market (price p_q), and the waste market (price p_w), provided that such a market exists. Initially, we omit the waste market and we introduce it in a second iteration of the decentralized model.[17]

The policy-maker can influence private decisions on each market thanks to a set of monetary transfer functions, denoted by $T_C(\cdot)$, $T_P(\cdot)$, $T_V(\cdot)$ and $T_R(\cdot)$, to the consumer, the producer, the virgin resource sector and the recycling sector, respectively.

1.5.1 Equilibrium in the absence of a waste market

The consumer determines the quantity of final good q they will consume in order to maximize her instantaneous surplus function $S_C \equiv [u(q) - p_q q + T_C(q)]$.[18] The producer of the manufactured good chooses the quantity of inputs v and r in order to maximize instantaneous profits $S_P \equiv [p_q q - p_v v - p_r r + T_P(v, r)]$ subject to the technological constraint $q = v + r$. The virgin resource owner chooses the extraction rate v maximizing profits

[16] We can obtain similar results with a less developed model, where the production sector is omitted. In this case, the final consumer directly consumes the two types of resources which are perfect substitutes for each other.

[17] Note that by considering this market economy, we only apply a partial decentralization process, in the sense that we do not develop explicitly the financial market. We assume that private agents discount their monetary flows at the rate ρ. Moreover, we do not include transaction and transportation costs here.

[18] There is no dynamic budget constraint, as we do not develop the financial market.

during an exogenous finite time T, expressed by $S_V \equiv \int_0^T [p_v v - c_v v + T_V(v)] e^{-\rho t} dt$ subject to (1.1). Last, the objective of the recycling sector is to determine both the share of final consumption good to be recycled β, and the recycled resource extraction r that maximize $S_R \equiv \int_0^T [p_r r - c_r r - q f(\beta) + T_R(r, \beta)] e^{-\rho t} dt$ subject to (1.2). Note that we directly focus here on interior values of β (the conditions for corner solution has been discussed previously in the central planner problem).

We introduce notations λ_i^* and $\tilde{\lambda}_i$ ($i \in \{R; E; W\}$) that refer respectively to the co-state variables of the central planner problem and the co-state variables of the decentralized problem.

Proposition 7. *Given λ_i^* $i \in \{E; W\}$, the shadow costs for an optimal trajectory, the set of policy instruments that restores optimality in the absence of a market for waste must satisfy the following conditions:*[19]

$$\frac{1}{q}\frac{\partial T_R(r,\beta)}{\partial \beta} = \lambda_W^* \tag{1.33}$$

$$T_V'(v) + \frac{\partial T_P(v,r)}{\partial v} + T_C'(q) = \pi(\beta) - \delta_v \lambda_E^* - \lambda_W^* \tag{1.34}$$

$$\frac{\partial T_R(r,\beta)}{\partial r} + \frac{\partial T_P(v,r)}{\partial r} + T_C'(q) = \pi(\beta) - \delta_r \lambda_E^* - \lambda_W^* \,, \tag{1.35}$$

with $\pi(\beta) \equiv \beta f'(\beta) - f(\beta) \geq 0$ for any β.

Proof. Maximizing surplus functions of the consumer, the producer, the resource owner and

[19] Here, as we follow a partial equilibrium approach, we do not consider the budget balance equation of the policy-maker. In particular, there is no reason at all to suppose that the net sum of all these monetary transfers must be equal to zero.

the recycling sector, we get the following first-order conditions:

$$u'(q) \leq p_q - T'_C(q)\,, \quad (= \text{ if } q > 0) \tag{1.36a}$$

$$p_q \leq p_v - \frac{\partial T_P}{\partial v} \text{ and } p_q \leq p_r - \frac{\partial T_P}{\partial r}\,, \quad (= \text{ resp. if } v > 0 \text{ or } r > 0) \tag{1.36b}$$

$$p_v \leq c_v + \tilde{\lambda}_V - T'_V(v)\,, \quad (= \text{ if } v > 0), \text{ with } \tilde{\lambda}_V = \tilde{\lambda}_{V0} e^{\rho t} \tag{1.36c}$$

$$p_r \leq c_r + \tilde{\lambda}_R - \frac{\partial T_R}{\partial r}\,, \quad (= \text{ if } r > 0), \text{ with } \tilde{\lambda}_R = \tilde{\lambda}_{R0} e^{\rho t} \tag{1.36d}$$

$$f'(\beta) \geq \tilde{\lambda}_R + \frac{1}{q}\frac{\partial T_R}{\partial \beta}\,, \quad (= \text{ if } \beta > 0). \tag{1.36e}$$

Given Eq. (1.36a)-(1.36d), the market-clearing conditions that characterize an equilibrium are:

$$u'(q) = c_v + \tilde{\lambda}_V - T'_V(v) - \frac{\partial T_P}{\partial v} - T'_C(q) \tag{1.37a}$$

$$u'(q) = c_r + \tilde{\lambda}_R - \frac{\partial T_R}{\partial r} - \frac{\partial T_P}{\partial r} - T'_C(q),\,. \tag{1.37b}$$

Additionally, we include condition (1.36e) to characterize the equilibrium. With these conditions, we can then compare them to the first-order conditions for optimal interior solutions in the central planner model (conditions (1.22a), (1.22b) and (1.6c)) to get equations (1.33) to (1.35). □

Simple economic policies can be illustrated when considering linear and additive separable transfer functions, *i.e.* tax-subsidy schemes: $T_C(q) = T_C \times q$, $T_P(v, r) = T_{Pv} \times v + T_{Pr} \times r$, $T_V(v) = T_V \times v$ and $T_R(r, \beta) = T_{Rr} \times r + T_{R\beta} \times \beta q$. In this case, first-best implementation requires four policy instruments.

The first instrument is a unit subsidy $\pi(\beta)$ (always non-negative given the properties of function $f(.)$) on the flow of produced/consumed goods. For the case of an interior solution ($\beta > 0$), from (1.6c), this subsidy can be rewritten as follows: $\pi(\beta) = [\beta q \Phi_\beta - q f(\beta)]/q$. This expression reads as the average net social gain of recycling the share β of the flow q of final good (remind that Φ_β is the marginal social benefit of the recycling effort and $f(\beta)$ is

the recycling cost per unit of output). The policy-maker must implement such a subsidy in order to correct for the positive externality generated by the waste production for free in the absence of waste market. It stems from the absence of a market for waste and the dynamic framework of the model. Recycled input accumulation is indeed not accounted for in the static modeling literature, thus not finding this externality. To the best of our knowledge, only the dynamic work of Pittel et al. (2010) highlights this market failure.

The second instrument is a unit subsidy of value λ_W^* on the flow of recycled good βq. It is required due to the avoided waste accumulation. This subsidy is always attributed to the recycling industry.

The third and fourth instruments are unit taxes corresponding to the two environmental negative externalities: GHG emissions and waste accumulation.

To integrate the GHG emissions externality in the model, we define the first unit tax using (1.34) and (1.35) to write: $T_{Rr} + T_{Pr} - T_V - T_{Pv} = (\delta_v - \delta_r)\lambda_E^*$. In this expression, λ_E^* is a uniform carbon tax that is applied to carbon content rate of each primary material. This tax can be imposed either to the producer of the final good or to the extraction sectors.

To integrate the waste accumulation externality in the model, we define the second unit tax of value λ_W^* that focuses on the potential waste generation from production/consumption of the final good q. This tax can be imposed either to the producer of the final good or to the consumer.

Observing the two monetary transfers associated with waste accumulation (instruments two and four), we can notice that when aggregated together, they represent a single tax of value λ_W^* on the effective waste accumulation $(1 - \beta)q$.

Table 1.3 illustrates various possibilities of who carries the burden or receives the benefit of these instruments.[20] Note that all these options are revenue-equivalent for the policy-maker.

[20] As usual, the transfer of the tax burden between producers and consumers may depend on the price elasticity of demand and supply functions. It is not the purpose here to develop this aspect.

Transfers to:	**Example 1**	**Example 2**	**Example 3**
Final consumer (T_C)	$\pi(\beta)q$	$(\pi(\beta) - \lambda^*_W)q$	$-\lambda^*_W q$
Producer (T_P)	$-\lambda^*_W(v + r)$	$-\lambda^*_E(\delta_v v + \delta_r r)$	$-\lambda^*_E(\delta_v v + \delta_r r)$
Virgin resource (T_V)	$-\lambda^*_E \delta_v v$	0	$\pi(\beta)v$
Recycling sector (T_R)	$\lambda^*_W \beta q - \lambda^*_E \delta_r r$	$\lambda^*_W \beta q$	$\pi(\beta)r + \lambda^*_W \beta q$

Table 1.3: Examples of policy-mix

These examples show how the carbon tax should be distributed between the producer (T_P) and resource managers (T_V and T_R). The implementation of this tax is the subject of a wide range of literature, and is not the focus of this chapter. While existing policies are often developed to favor the recycling industry, our model acknowledges the level of the GHG emissions of the sector, and advocates for an appropriate carbon tax on this industry. The tax does not replace the existing support for the recycling industry. These tax-subsidy support schemes remain as they address other issues unrelated to GHG emissions.

It is interesting to note that whatever the transfer structure, there is a subsidy for the recovered flow of material for the recycling sector. The combination of this subsidy with a taxation on products is often promoted in economic literature as a deposit-refund scheme (Fullerton and Kinnaman, 1995; Choe and Fraser, 1999). Additionally, this combination accounts for the specific characteristics of the industrial sector like emissions rates, private and social costs of waste management (Acuff and Kaffine, 2013). Although the system suggested by our model is still hardly implemented, there are several subsidy schemes for recycling industries and several taxation schemes on waste.

In the examples of Table 1.3, taxation relies on the global flow of material subject to potential waste damage. Extended Producer Responsibility (EPR) systems are an example where producers have to assume financial responsibility for the potential environmental damage of their products during their whole life-cycle (through an eco-contribution). In

example 2 and 3, the cost of potential waste damage is assumed by the final consumer. We can note that even if the tax is applied to the producer, it will eventually be reflected on the price of the product and the burden of the tax will be assumed by the consumer. In existing examples, EPR instruments are implemented by industrial sector, in varying sizes. In France in 2020, Citeo (an agency organizing EPR for packaging industries) fixes a baseline level for eco-contributions of 16,53 ct€/kg for paper and cardboard, and 28,88 ct€/kg for light PET (colourless polyethylene terephthalate).[21] Note that almost 80% of these amounts are redistributed to the recycling sector (collecting, sorting and resource production), thus covering the subsidy part of our monetary transfers (Citeo, 2019).

In the case of example 3, note that subsidy $\pi(\beta)$ is applied to resource producers instead of the final consumer. This reflects the prospective activity of recycling, as consumed resources have the potential to be recycled resources or waste.

1.5.2 Existence of a waste market

Let us now assume that the final consumer can sell its flow of waste q at price p_w on a specific market. The consumer's surplus must now include the revenue from the waste sale $p_w q$: $S_C = [u(q) + (p_w - p_q)q + T_C(q)]$. The new expression of the intertemporal surplus of the recycling sector is $S_R = \int_0^T [p_r r - c_r r - q f(\beta) - p_w q + T_R(r, \beta, q)] e^{-\rho t} dt$, which must be maximized subject to (1.2). The private marginal cost of recycling now also includes purchasing waste. Behaviors of the other agents are unchanged.

Proposition 8. *Given λ_i^* $i \in \{E; W\}$, the shadow costs for an optimal trajectory, the set of policy instruments that restores optimality with a market for waste must satisfy the following*

[21] This amount is completed by a unit based amount and a bonus-penalty system depending on the eco-design of the product.

conditions:

$$\frac{1}{q}\frac{\partial T_R}{\partial \beta} = \lambda_W^* \tag{1.38}$$

$$T_V'(v) + \frac{\partial T_R}{\partial q} + \frac{\partial T_P}{\partial v} + T_C'(q) = -\delta_v \lambda_E^* - (1-\beta)\lambda_W^* \tag{1.39}$$

$$\frac{\partial T_R}{\partial r} + \frac{\partial T_R}{\partial q} + \frac{\partial T_P}{\partial r} + T_C'(q) = -\delta_r \lambda_E^* - (1-\beta)\lambda_W^* \,. \tag{1.40}$$

Proof. Programs of the final good producer and of the virgin resource extracting sector are not affected by the existence of a waste market. Thus, Eq. (1.36b) and (1.36c) still hold. Additionally, the first-order conditions (1.36d) and (1.36e) are unchanged for the recycling sector, but another condition appears for this sector (1.41a) and the condition for the consumer is changed (1.41b):

$$u'(q) + p_w = p_q - T_C'(q)\,, \tag{1.41a}$$

$$f(\beta) + p_w = \beta\tilde{\lambda}_R + \frac{\partial T_R}{\partial q}\,. \tag{1.41b}$$

When comparing these equations with the first-order conditions for optimal interior solutions in the central planner model given by (1.22a), (1.22b) and (1.6c), we get equations (1.38) to (1.40). □

As expected, the waste market allows to fully internalize the positive externality associated with the waste generation by the final consumer.

1.6 Conclusion

In this chapter, we provided a discussion on optimal use of resources when climate change and waste are both additional constraints for the social planner. Depending on initial characteristics of the industrial sector, the optimal path can be divided into phases of virgin and recycled production, with a potential switch of inputs if recycling becomes more socially

profitable. This change in the sector intervenes when the scarcity of the virgin resource prevails and/or private costs also favour the use of recycling. When introducing a climate constraint and a difference between emissions rates, a change of inputs can also occur when this difference is sufficiently large, especially compared to the fixed difference between private delivery costs of inputs. In parallel, the social planner will develop a recycling (or waste recovering) strategy, such as it alleviates the cost of waste accumulation and allows for production from a recycled input. This shows the duality of the activity of sorting waste: a practical goal being the reduction of costs and a speculative goal as the use of recycled input can potentially occur in the future. This leads to inverse U-shaped recycling programs, where it is optimal to increase recycling at first and then decrease it. Moreover, when the marginal cost of recycling is too high, it leads to phases where the recycling rate is zero (although it is still possible to produce from the existing stock of recycled input at the same time). In parallel, under certain conditions, production shows catch-up phases at the end of the program, as full marginal costs start decreasing (involving an increase in production). Our model gives a better understanding of the complementary nature of developing climate change objectives and addressing resource scarcity issues. The model addresses the fact that recycled inputs, while often a cleaner option for production also delaying scarcity, still come with GHG emissions.

While the central planner program gives many insights regarding optimal recycling activities, it does not give suitable results for policy making. The decentralized model of the sector allows to specify the need for public intervention. This new model highlights a possible tax/subsidy scheme in order to implement a first-best solution. To do this first best solution, a taxation based on GHG emissions must be introduced in both sectors, weighed by the carbon intensity of the branch (the carbon intensity is higher for virgin extraction in most cases). This comes in addition to a tax-subsidy system based on waste and stock replenishment which are already demonstrated in Extended Producer Responsibility programs currently implemented for certain products.

Our model is mainly focused on the case of metal industries, where strong GHG emissions assymetries and perfect substitution are relevant hypothesis. In the appendix we address other constraints on recycling: the existence of a scrap value for waste, a decoupling of GHG emissions between collection of waste and transformation into an input, and a limited capacity for the stock of recycled input. Extensions toward other industries would require relaxing some hypothesis used here. For instance, loss of quality is in fact an important topic for the academic research on circular economy, especially when the efficiency of a process is studied (Figge et al., 2018). This is, for instance, the case in the paper and cardboard industry, when each recycling loop lowers the quality of the pulp and only allows a limited amount of cycles depending on the needed quality (usually 7 and 8 cycles are technically possible, but fiber is rather used 3.5 times on average in Europe (ERPC, 2015)). Lafforgue and Rouge (2019) provide the basis for understanding the optimal production path when technological progress in recycling allows for higher quality in recycled inputs.

1.A Appendix

Our model can be extended to account more precisely for existing environmental externalities. First, we assume the existence of a scrap value for the waste stock after the program. Second, we decouple the GHG emissions of the recycling process into those related to the collection of waste and to those related to production of the recycled input. To end, we consider the limited capacity of the recycling sector by imposing a cap on the cumulative flows of recycled materials.

1.A.1 Scrap value for waste

Here, we assume that waste accumulation is no longer damaging from time T onward: $D(W(t)) = 0, \forall t \geq T$. From (1.3), given that extraction/consumption flows are zero, the remaining stock of wastes evolves as: $W(t) = W(T)e^{-\alpha(t-T)}$, for $t \geq T$. A more realistic

approach consists of attaching a scrap value Ω to $W(T)$ in case where the damage caused by the left-over waste stock remains after the end of the exploitation period of time:

$$\Omega(W(T)) = \int_T^\infty D\left(W(T)e^{-\alpha(t-T)}\right)e^{-\rho t}dt. \tag{1.42}$$

This scrap value must be added to the value function (1.4) of the social planner program, which slightly modifies the tranversality condition relative to stock W:

$$\left[\lambda_W(T) - \Omega'(W(T))\right]W(T)\mathrm{e}^{-\rho T} = 0\,. \tag{1.43}$$

Using (1.6f), (1.42) and (1.43), the shadow cost of the waste stock at any point in time can be expressed as:

$$\begin{aligned}\lambda_W(t) &= e^{(\rho+\alpha)t}\left[\Omega'(W(T))e^{-(\rho+\alpha)T} + \int_t^T D'(W)e^{-(\rho+\alpha)s}ds\right]\\ &= \int_t^T D'(W)e^{-(\rho+\alpha)(s-t)}ds + e^{-\rho T}\int_T^\infty D'(W)e^{-(\rho+\alpha)(s-t)}ds\,.\end{aligned} \tag{1.44}$$

At any time t, the shadow marginal cost of waste contains two components: the sum from t to T of the marginal damage, discounted at rate $(\rho+\alpha)$ to reflect the marginal absorption process of the stock of waste by the environment, and the value at time T of the sum from T onward of the discounted marginal damage, i.e. the scrap value term. With constant marginal damage, this shadow cost becomes:

$$\lambda_W(t) = \frac{\theta}{(\rho+\alpha)}\left(1-(1-e^{-\rho T})e^{-(\rho+\alpha)(T-t)}\right)\,, \tag{1.45}$$

and we still have $\dot{\lambda}_W < 0$. The scrap value intervenes by adding an exogenous factor $1-\mathrm{e}^{-\rho T} \in]0,1[$ to the decreasing term of the shadow cost. The remaining shadow cost for waste at the end of the program is the marginal scrap value $\frac{c_W}{\rho+\alpha}\mathrm{e}^{-\rho T} = \Omega'(W(T))$, instead of zero for the model previously discussed.

This additional assumption does not change the change the arbitration in the model, as the first-order conditions (1.6a)-(1.6f) remain the same. However, the optimal recycling rate follows a different time path, according to the following proposition.

Proposition 9. *When the recycling rate $\beta(t)$ is not always zero and considering a scrap value for the waste stock damage, it reaches a maximum at time $\hat{T}_\beta$. This instant can take the following values:*

$$\hat{T}_\beta = \begin{cases} 0 & , \text{ if } \frac{\mathrm{e}^{(\rho+\alpha)T}}{(1-\mathrm{e}^{-\rho T})} \leq \frac{c_W}{\rho\lambda_{R0}} \text{ or } \lambda_{R0} = 0 \\ T & , \text{ if } \frac{c_W}{\rho\lambda_{R0}} \leq \frac{\mathrm{e}^{\rho T}}{(1-\mathrm{e}^{-\rho T})} \\ \frac{1}{\alpha}\left[\ln\left(\frac{\rho\lambda_{R0}}{c_W(1-\mathrm{e}^{-\rho T})}\right) + (\rho+\alpha)T\right] \in]0,T[& , \text{ if } \frac{\mathrm{e}^{\rho T}}{(1-\mathrm{e}^{-\rho T})} < \frac{c_W}{\rho\lambda_{R0}} < \frac{\mathrm{e}^{(\rho+\alpha)T}}{(1-\mathrm{e}^{-\rho T})}\,. \end{cases} \tag{1.46}$$

Proof. Now the marginal benefit of recycling is:

$$\Phi_\beta(t) = \lambda_{R0}\mathrm{e}^{\rho t} + \frac{c_W}{\rho+\alpha}\left(1 - \left(1-\mathrm{e}^{-\rho T}\right)\mathrm{e}^{-(\rho+\alpha)(T-t)}\right)$$

Following the same analysis as the general case, we get Proposition (9). □

1.A.2 Decoupling emissions of the recycling branch

Another possible extension of the model is to consider a decoupling of the emissions related to the recycling process. In that case, parameter δ_r only concerns the transformation and use of the recycled input, while we add a constant rate δ_β for the activity of collecting and sorting from the waste flow: $\delta_\beta\beta(t)q(t)$. This leads to the following new dynamic equation for emissions accumulation:

$$\dot{E}(t) = [\delta_v + \beta(t)\delta_\beta]v(t) + [\delta_r + \beta(t)\delta_\beta]r(t),\ E(0) = 0\,. \tag{1.47}$$

Practically speaking, this modification in the model does not change the arbitration between the two resources. This is because the term relative to this type of emissions disappears with the Full Marginal Costs difference, the same way it does for terms relative to waste accumulation, recycled stock replenishment and the cost of recycling. However, there is an impact on the first-order condition relative to recycling, with a new function Φ_β:

$$\Phi_\beta = \lambda_R + \lambda_W - \delta_\beta \lambda_E \tag{1.48}$$

This more complex expression of the marginal social gain of recycling can be easily reinterpreted: we now recycle to replenish stock R and alleviate the cost of stock W but it costs emissions to the industrial sector, at rate δ_β. Developing this expression, we obtain: $\Phi_\beta = (\lambda_{R0} - \delta_\beta \lambda_{E0})\mathrm{e}^{\rho t} + \frac{c_W}{\rho+\alpha}\left[1 - \mathrm{e}^{-(\rho+\alpha)(T-t)}\right]$. The same analysis as before can be done for the evolution of the recycling rate. However we can also add the following proposition:

Proposition 10. *It is never optimal to start recycling under the two following conditions:*

- *There is a higher initial social cost for emissions than the depletion of the recycled stock, i.e.* $\lambda_{R0} \leq \delta_\beta \lambda_{E0}$*;*
- *Damage of waste accumulation is such that* $\frac{c_W}{\rho+\alpha} \leq f'(0)$.

In this case, the recycled resource is never used, $\lambda_{R0} = 0$.

Proof. Given the new expression of Φ_β, we see that it is decreasing when $\lambda_{R0} \leq \delta_\beta \lambda_{E0}$. As $\lim_{-\infty} \Phi_\beta = \frac{c_W}{\rho+\alpha}$, equation $\Phi_\beta = f'(\beta)$ does not have a solution when $\frac{c_W}{\rho+\alpha} \leq f'(0)$, proving the necessary condition of the proposition. As there is no recycled stock initially, we never use the recycled resource. Note that a more constraining condition for the absence of recycling is $\Phi_\beta(0) \leq f'(0)$. □

By adding an emission term for recycling activities, we can highlight a simple condition on the parameters of the model for which recycling is never optimal. If the social cost of

GHG emissions relating to the recycling activity is too high, or if the avoided cost of waste accumulation is too low, recycling is never optimal. Finally, note that we also observe the catch-up phase for the use of inputs with this extension of the model.

1.A.3 Limiting the capacity of the recycling sector

Optimal program with a cap on $R(t)$

In this final iteration of the model, the recycling sector faces an upper limit regarding the size of the recycled stock R. This limit suggests that the speculative strategy for recycling is bounded. One can view this limit as the implicit storage cost of the recycled resource. To analyze this concept mor individually, we do not include the previously analyzed GHG ceiling. For this extension, we introduce a cap $\bar{R}$ such as $R(t) \leq \bar{R}$. This new constraint leads to a new first order condition on stock R, introducing the associated co-state variable χ:

$$\dot{\lambda}_R = \rho\lambda_R + \chi \quad \text{with} \quad (\bar{R} - R(t))\chi(t) = 0 \quad \text{and} \quad \chi(t) \geq 0\,. \tag{1.49}$$

The other optimal conditions refer to the basic case developed in Section 1.4. The optimal sequence of use has now the following characteristics:

Proposition 11. *Regardless of the recycling rate $\beta(t)$, the optimal path with a cap on the recycled stock is such that:*

1. *While the recycled stock is not full, it follows the optimal program described in Proposition 1, with a sequential order of use.*
2. *When recycled stock is full, two scenarii can occur:*
   (a) *If there is no recycling, we only use the virgin resource;*
   (b) *If there is some recycling, there can be a stationary regime for R where the two resources are used simultaneously such as $\beta v = (1-\beta)r$.*

Proof. While the cap $\bar{R}$ is not reached, equations from the general problem hold, especially $\dot{\lambda}_R = \rho\lambda_R$, thus proving 1. When $\bar{R}$ is reached, meaning a maximum for R, we have $\beta v = (1-\beta)r$ from equation (1.2) (it can be only instantaneous and $R(t)$ starts decreasing again). Then this equality for stationarity on R proves 2. □

The cap on the recycling stock offers the possibility of a stationary regime, thus with a simultaneous use of the two resource. At each instant, the recycling rate allows to perfectly compensate the depletion of R. However, this should not be interpreted as a sustainable scenario, as $\beta < 1$ gives an asymmetry in the material flows: if recycling is low ($\beta < 0,5$) virgin production prevails, the opposite otherwise.

Stationary regime for stock $R(t)$

As seen before, the stationary regime for stock $R(t)$ is defined by a simultaneous production from both inputs, and a ratio v/r governed by the evolution of the recovery rate β. From Proposition 11 and the relationship between flows, we know that this ratio grows when rate β declines: less recovery in recycling means that stock R is less exploited (r diminishes) and eventually consumption is sustained by higher extraction (v increases).

To analyze further, let T^* be the starting point of this stationary regime. From the new FOC on stock R given by equation (1.49) and its associated transversality condition, we get expression for the shadow value of the recycled stock: $\lambda_R(t) = -\int_t^T \chi(s)\mathrm{e}^{-\rho(s-t)}ds$. This cost is negative, as scarcity is not at stake here ($R(T) = \bar{R} > 0$) and the recycling stock is subject to an upper limit (similar to what was done for the GHG emissions stock). The model during the stationary regime is now governed by the following equations, after instant T^*:

$$\forall t \geq T^*, \quad c_v + \lambda_{V0}\mathrm{e}^{\rho t} = c_r - \int_t^T \chi(s)\mathrm{e}^{-\rho(s-t)}ds \tag{1.50}$$

$$f'(b) = -\int_t^T \chi(s)\mathrm{e}^{-\rho(s-t)}ds + \frac{c_W}{\rho+\alpha}\left(1-\mathrm{e}^{-(\rho+\alpha)(T-t)}\right). \tag{1.51}$$

Equation (1.50) comes from the simultaneous use of the resource, meaning that both marginal costs are equal. One necessary condition as seen here is that the cost for the recycled input c_r should be higher than the cost for the virgin input c_v when added to its scarcity rent λ_V. When expressed at the end of the program, it states that both should be equal. In that sense, either the virgin resource is scarce ($\lambda_{V0} > 0$) or both marginal delivery costs are equal.

Another necessary condition for simultaneous use of the resources is that the recycling activity occurs during the whole stationary phase ($\beta > 0$). Equation (1.51) expresses the equality between cost and marginal benefit of recycling as described earlier. The optimal recycling program can be precised as follow:

Proposition 12. *If the optimal path of the program enters a stationary regime for stock R, the optimal recycling strategy is to always recycle during this phase, and if β reaches a maximum during this phase, $\hat{T}_\beta$ is defined by:*

$$T^* < \hat{T}_\beta < T \quad \textit{such that} \quad (c_r - c_v) = \frac{c_W}{\rho} \mathrm{e}^{-\alpha(T-\hat{T}_\beta)}. \tag{1.52}$$

Proof. From the equality of FMCs (1.50) expressed at instant T of the program, we get $\lambda_{V0} = (c_r - c_v)\mathrm{e}^{\rho T}$. This equality replaces λ_R in the FOC for β, and is used in equality $f'(\beta) = \Phi(\beta)$. The maximum is reached at $\hat{T}_\beta$ when $\dot{\Phi(\beta)} = 0$, thus giving equation (1.52). □

During this specific phase, we have an exogenous expression for the maximum of recycling. This expression (1.52) shows that this instant occurs at a later time when the cost difference $c_r - c_v$ is relatively significant. On the other hand, when the damage of waste accumulation is relatively significant, $\hat{T}_\beta$ happens earlier. This prompts early recycling as we do not have a scarcity of R and the shadow cost of waste diminishes.

CHAPTER 2

SECTORAL, RESOURCE AND CARBON IMPACTS OF INCREASED PAPER AND CARDBOARD RECYCLING

* * *

The question of whether paper and cardboard recycling has a positive or negative carbon impact is crucial as the global forest sector plays a central role in climate change mitigation. In this chapter, with the help of an integrated assessment bio-economic model, we assess the effects of increased paper and cardboard recycling on the resource and the carbon balance of the forest sector. We show that these results are strongly linked to the hypothesis of substitution or complementarity between recycled and wood-pulp. We find increased emissions at the pulp sector level, while effects on other wood products are small. The impact on total net sequestration is positive when pulp products are substitutes, negative when complements.

* * *

2.1 Introduction

2.1.1 Context and motivations

Studying the underlying dynamics of the development of recycling in in chapter one brings to consider the different externalities linked to resource extraction and disposal. Historically, resource depletion and waste accumulation have been the principal motivation for the development of a circular economic model, mainly through recycling programs. However, recent developments led to a more holistic approach of environmental issues, as local and global pollution have become a focal point of environmental studies. The study of the circular economy is now strongly connected to climate change challenges, as a redefinition of our linear economic model could have impacts on greenhouse-gas emissions (GHG) (ADEME and FEDEREC, 2017; UNEP, 2019; ADEME, 2019).

This chapter focuses on the specific case of paper and cardboard recycling in France. We assess its effect on the forest sector with economic and carbon perspectives. We show that results are strongly linked to the hypothesis of substitution or complementarity between recycled and wood-pulp. In the case of substitutes, we account for a reduction of the carbon balance on the global forest sector, mainly driven by the sequestration mechanism of the forest and increased demand in fuel wood. However when products are complementary, increased recycling leads to more emissions.

More generally, the paper and cardboard recycling sector is well developed in France, with 66% of the production coming from recovered waste (ADEME and Bio by Deloitte, 2017). Besides, the paper pulp industry is directly connected to the global forest sector, which plays a crucial role in climate change mitigation. Unlike other recycling sectors like metals or plastics, there is little to no substantial difference in GHG emissions between recycled and wood-pulp industrial process (ADEME and FEDEREC, 2017; ADEME, 2019). Besides, complex cascading impacts on the resource and other wood products suggest unclear economic and environmental impacts of paper and cardboard recycling. Our objective is to

assess the effect of recycling by expending the usual scope of the study to the forest sector. The main goal here is to investigate wether climate change mitigation mechanisms of the forest sector offset or not the small difference in climate impact between recycled and virgin pulp.

Our contribution here is the sectoral analysis of a recycling industry and its impacts in terms of GHG emissions. Instead of a standard accounting for GHG displacements using the LifeCycle Assessment methodology (LCA), we also use an Integrated Assessment Model (IAM)(the French Forest Sector Model - FFSM++)[1] providing prospective scenarios of the French forest sector. Using an IAM allows us to account for the whole forest sector with a bioeconomic perspective, including dynamic mechanisms. Different economic hypothesis on the new recycling branch as well as a carbon accounting module allows us to give the climate impact of increased recycling at different scopes.

2.1.2 Related literature

The economic literature tackled the topic of recycling very early with a focus on social costs associated to waste accumulation and resource depletion, in order to find an optimal level of pollution (Hoel, 1978; Smith, 1972). Later on, some papers, like our first chapter, invested the topic of pollutant emissions and recycling: with a dynamic scope and a sectorial model like Huhtala (1999); with an endogenous growth model like Di Vita (2004); and, through a static micro-economic approach like Acuff and Kaffine (2013). These theoretical papers often isolate costs as the main barrier for development of a recycling industry, and accounting for environmental externalities can make this development more desirable. Considering recycling is also central for companies, with the influence of the reusability on the role of post consumer collection (Tsiliyannis, 2008), and solving supply chain problems (Vahdani et al., 2012). However, another central question when dealing with a recycled product, is the substitution with virgin material, especially for paper and cardboard.

[1] https://ffsm-project.org/wiki/en/home

The econometric literature highlights this issue of substitution, although studies do not appear to give a clear-cut result. Lee and Ma (2001) show that recycled and wood-pulp could be substitutes, however they find very low values for cross-elasticities that are not statistically significative. Some studies also show complementarity between inputs, with pulp industries hardly substituting materials in their transformation process (Lundmark and Olsson, 2015; Lundmark and Söderholm, 2003). A literature review from Mansikkasalo et al. (2014) shows that we can find variations between countries, that could be driven from technological differences. They also show that inputs can be substitutes or complements. Those heterogenous results can be explained by the wide diversity of products within the paper and cardboard industry. Beyond purely market reasons, these results can also be addressed by examining technical difficulties of swapping inputs in pulp production processes (ERPC, 2015). These considerations encouraged us to explore different scenarios where recycled and wood-pulp could be complementary or substitutable products. This question of substitution is actually a crucial point yet to be addressed in the circular economy literature. An undesirable scenario would be that recycling contributes to circular economy by creating a rebound effect in production when recycled products poorly substitute to others or even are complements, thus fulfilling an unsustainable goal (Zink and Geyer, 2017).

Beyond these investigations on substitution, econometric studies show low values for price elasticities of the demand for recycled pulp (Deadman and Turner, 1981; Edgren and Moreland, 1990; Lee and Ma, 2001). This low value is also found for the supply of recycled pulp (Mansikkasalo et al., 2014). However, differences can be found between the several use of transformed pulp, showing the difficulty to compute a composite value describing recycled pulp markets (Lundmark and Olsson, 2015; Lundmark and Söderholm, 2003). Another key parameter describing recycled pulp markets is the level of substitutability with, in this case, other wood products. Early studies tackle this question showing that wood-pulp is a complement to solid-wood (Newman, 1987).

These econometric considerations aside, GHG displacement factors of paper recycling

have often been studied with strong economic assumptions. Early assessments from Byström and Lönnstedt (1997) do not show that increased recycling could be environmentally friendly, while only taking into account the pulp sector. One paper from Merrild et al. (2009) gathers estimations on GHG emissions for different recycling technologies. By extending the analysis to the boundaries of the forest resource, they are able to take into account sequestration and substitution mechanisms. However, they assume substitution with wood-pulp and that unused wood is consumed for energy as a substitute for fossil fuel. They get then a high environmental performance of recycling, without taking into account the complexity of cascading effects in the forest sector. On the other hand, a French study from ADEME and FEDEREC (2017) uses LCA[2] without feedbacks and competition in the rest of the forest sector. By comparing GHG emissions of virgin and recycled production, including avoided end-of-life, their static analysis concludes that paper and cardboard recycling emits the same or even more than producing wood-pulp. This study is extended in ADEME (2019) and finally shows GHG gains when adding assumptions on wastepaper anaerobic degradation. Finally, an Input/Output model applied to the Dutch case by Nakamura (1999) shows decreases in emissions while taking into account wide inter-industry effects, but not economic effects, being a quantity based model.

Including the forest sector in the analysis of the GHG impact of wood products recycling seems crucial as its relationship with climate change mitigation is consequential. It draws on two different mechanisms we take into account in this study. First, sequestration relies in the growth of forests in order to capture additional carbon and reduce GHG concentration. It depends strongly on climatic factors of the forest, thus varies around the world and is likely to be affected by ongoing and future climate variations (Cook-Patton et al., 2020). In addition to this environmental factor, management practices of the forest are also central as strategies on collected and replanted biomass can have significant impacts (Lobianco, Caurla, Delacote and Barkaoui, 2016). Second, the substitution mechanism consists in using wood

[2]ISO 14040 and ISO 14044

instead of other products in industrial process. The GHG balance is impacted through different activities including replacing fossil fuel energy with fuel-wood, using less energy for manufacturing forest products instead of non-wood products as well as storing carbon in products (Sathre and O'Connor, 2010; Churkina et al., 2020). One theoretical analysis that includes recycling is carried out by Tatoutchoup and Gaudet (2011) and shows a reduction of the stock of trees, while not accounting for specific uses of wood and assuming perfect substitution between recycled and virgin products. Following this approach, Tatoutchoup (2016) computes an optimal recycling rate maximizing the forest area and the social net benefit. However in both papers they do not extend the analysis to formally compute the carbon balance of the model. Besides, econometric results stated before incite us to alleviate the hypothesis of perfect substitution and explore other possibilities.

These observations incited us to study the carbon impact of recycling while considering market interactions in the global forest sector. Such a study is carried out with an integrated assessment model representing the dynamic evolution of the french forest sector. A nice review of such models is carried out by Riviere et al. (2020), investigating their thematic focus, noticing a rise in climate change related topics. A similar model, the Global Forest Products Model, already shows that recycling has little impact on other wood products, however it does not compute GHG displacement factors (Buongiorno et al., 2003). A study on a smaller scale by Pieratti et al. (2019) using multi-criteria decision analysis with circular economy principles underlines under-optimal management practices for a single forest, while the global GHG balance is positive thanks to fossil fuel substitution. However they do not examine the specific effect of recycling. D'Amato et al. (2020) stress that LCA assessments of forest-based economy, while they very often include climate change impacts, lack waste and circularity considerations.

Our model here, FFSM++, has provided a wide stream of literature regarding long-term assessments of the sector, with market, resource or climate outlooks. Introduced by Caurla et al. (2010), it already produced results on the introduction of climate change mit-

igation policies and their possible crossed-effects (Caurla, 2012; Caurla, Delacote, Lecocq and Barkaoui, 2013; Caurla, Delacote, Lecocq, Barthès and Barkaoui, 2013). Climate impacts and risk aversion are also a useful scope for long-term assessments of the French forest dynamics, with Lobianco, Delacote, Caurla and Barkaoui (2016) showing overall stability of the volume stocks. Finally, FFSM++ also allows the observation of substitute products, with the introduction of crossed-elasticities (Caurla, Delacote, Lecocq, Barthès and Barkaoui, 2013; Petucco et al., 2019). We present here an extension of this model that links pulp production to waste accumulation and recycling.

The remaining of this chapter is structured as follows. Section 2.2 presents the extension we added to FFSM++ and the simulation strategy to analyse recycling. Section 2.3 describes the impact of increased recycling in the model, with a focus on carbon at different scales of the sector. Section 2.4 concludes.

2.2 Adding a paper recycling loop to a forest sector model

2.2.1 Modular structure of FFSM

FFSM is designed to explore the dynamics of markets and resources, as well as policies and their economic and environmental impacts[3]. Each year, the model computes prices and quantities for different primary and secondary wood products based on a Samuelson (1952) spatial equilibrium. This way, trade of products is represented on the national level, using prices in different regions, as well as on the international level, using Armington (1969) substitution model. Sauquet et al. (2011) investigate this last point with the FFSM framework, on the specific product sawnwood. This economic model is completed by a resource module representing the dynamics of the forest sector, and a carbon accounting module in

[3]See Caurla (2012) for a full description of the model, and `https://ffsm-project.org/wiki/en/doc/home#published_articles` for any further extension.

order to assess the emissions balance of the sector. Carbon accounting consists in reporting sequestration mechanisms (through wood products, inventoried and extra forest biomass) and emissions (direct forest operations, energy and material substitution for transformed wood products with coefficients from the already existing literature) described in detail in Lobianco, Caurla, Delacote and Barkaoui (2016).

The dynamic recursive structure of FFSM is described in Figure 2.1, where previous year harvest and forest dynamics give the availability of the resource to the market module. The detailed description of the market module is given by Figure 2.2, with a supply in primary products upstream (roundwood and industrial wood) and a demand in products of first transformation downstream. The interface between production and consumption consists in the transformation industry, with an input-output logic (see Figure 2.2).

The addition of a recycling loop consists here in the creation of a pair of primary and transformed products. Recovered waste of the previous year (from both recycled and virgin pulp consumed in the model) is reprocessed into recycled pulp, linked to the demand in wood-pulp through complementarity or substitution.

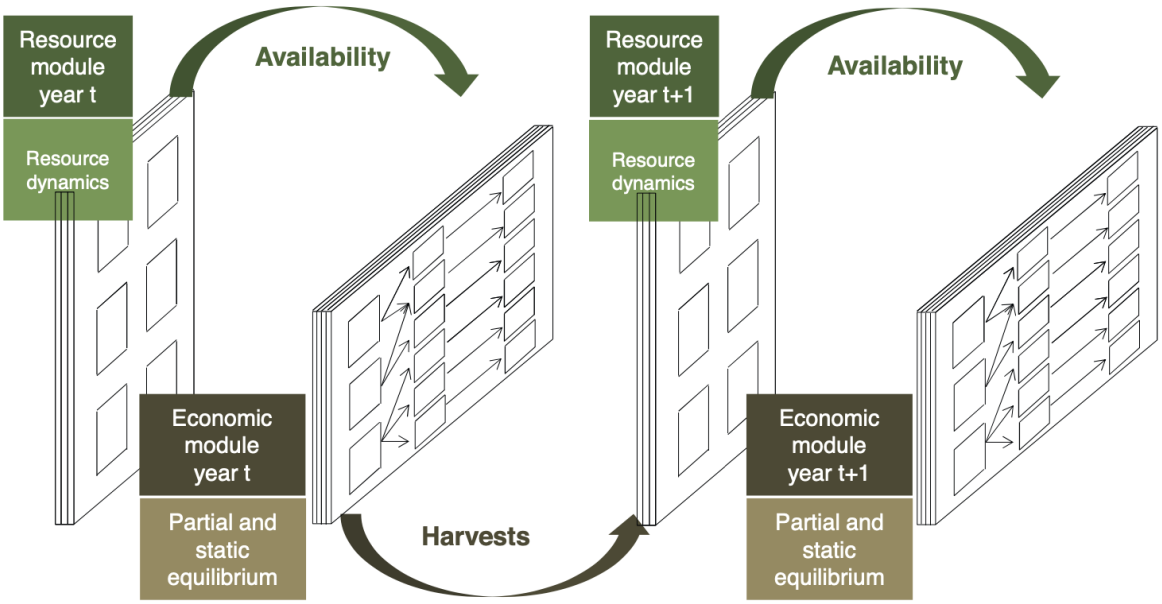


Figure 2.1: Recursive structure of FFSM

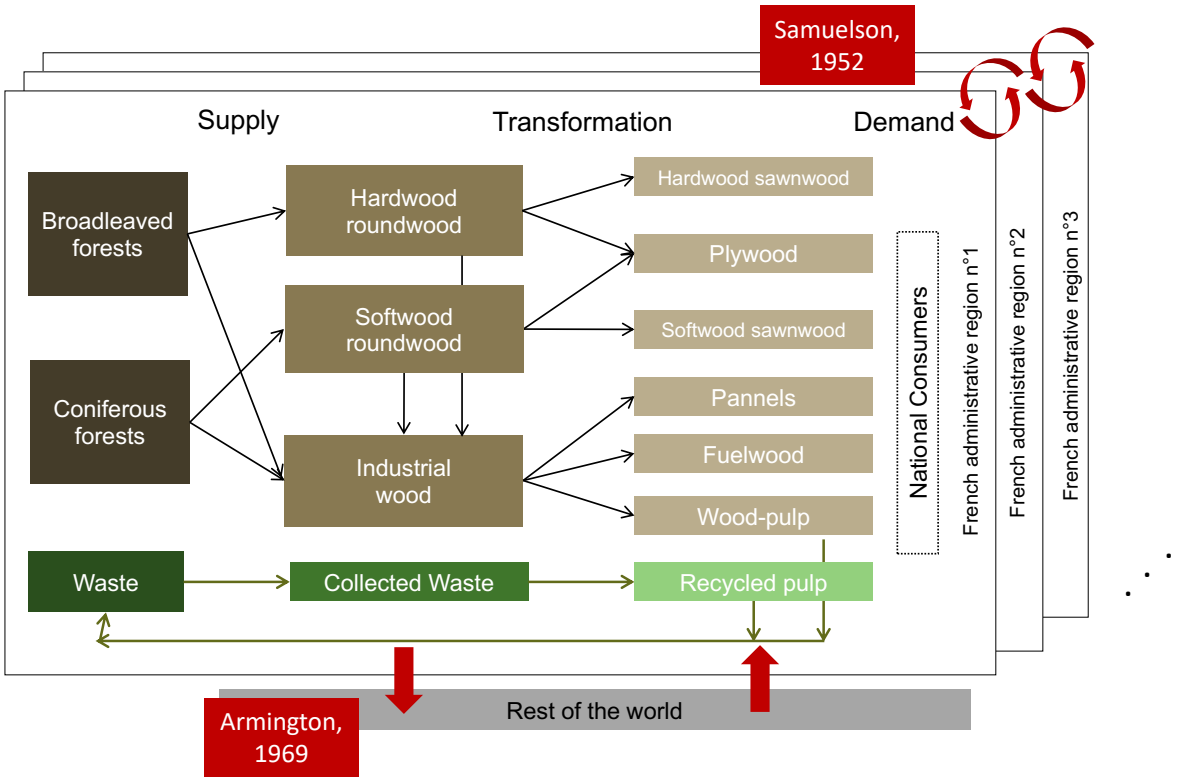


Figure 2.2: Products structure of FFSM

2.2.2 Dynamics of pulp recycling

The regional demand for recycled pulp is similar to demands for other transformed products introduced in FFSM (Caurla et al., 2010), expressed as a composite demand $D_{recy,i,t}$ (for region i at year t) with a substitution between foreign and local shares of the total demand. Besides the recursive dependency with supply and prices, we introduce a cross elasticity between the recycled pulp and wood-pulp:[4]

$$D_{recy,i,t} = D_{recy,i,t-1} \left(\frac{P_{recy,i,t}}{P_{recy,i,t-1}} \right)^{\sigma_{recy}} \left(\frac{\frac{P_{recy,i,t}}{P_{pulp,i,t}}}{\frac{P_{recy,i,t-1}}{P_{pulp,i,t-1}}} \right)^{\eta_{recy,pulp}} \tag{2.1}$$

Where

[4]See Caurla, Delacote, Lecocq, Barthès and Barkaoui (2013).

- σ_{recy} is the price elasticity of demand;
- $P_{recy,i,t}$ is the price of composite recycled pulp, in region i at year t;
- $P_{pulp,i,t}$ is the price of the substitute (or complementary) product, wood-pulp in region i at year t;
- $\eta_{recy,pulp}$ is the cross elasticity of demand.

A similar and symmetric cross-elasticity $\eta_{pulp,recy}$ in the demand function for wood-pulp is introduced.

We also introduce a composite supply of recovered waste $S_{waste,i,t}$ [5] depending on its price and the stock of waste paper and cardboard that could be recovered for recycling purposes:

$$S_{waste,i,t} = S_{waste,i,t-1} \left(\frac{P_{waste,i,t}}{P_{waste,i,t-1}} \right)^{\epsilon_{waste}} \left(\frac{W_{i,t}}{W_{i,t-1}} \right)^{\beta_{waste}} \tag{2.2}$$

Where:

- $P_{waste,i,t}$ is the price of composite recovered waste paper and cardboard, in region i at year t;
- $W_{i,t}$ is the volume of waste cardboard and paper that can be recovered;
- ϵ_{waste} is the price elasticity of supply;
- β_{waste} is the stock elasticity of supply.

From the dispersive use of material and the technological limits associated to recycling and recovery, stock $W_{i,t}$ is a share of the total pulp (from wood and recycled) that has been produced at $t-1$. Following a common assumption in literature concerning recycling of resources, we do not account for the duration of use of the pulp produced that would induce

[5] see Caurla et al. (2010) for the detailed construction of the composite supply of primary products

lagged effects in the model (Palmer et al., 1997; Huhtala, 1999). This means that each year, paper is produced, consumed and then recycled or disposed of the following year.

$$W_{i,t} = \gamma_{i,t}(D_{pulp,i,t-1} + D_{recy,i,t-1}) \tag{2.3}$$

Where:

- $\gamma_{i,t}$ is the share of pulp (recycled and wood) consumed in region i at year $t-1$ that can be recovered in region i at year t;
- $D_{pulp,i,t-1}$ is the demand in wood-pulp in region i at year $t-1$;
- $D_{recy,i,t-1}$ is the demand in recycled pulp in region i at year $t-1$;

Recovered waste paper and cardboard are transformed by a recycling industry at a constant cost of production c_{recy} for each unit of recycled pulp produced. This cost enters in the surplus expression, as the model is solved through a static Samuelson spatial equilibrium maximizing the surplus function (detailed at length in Caurla (2012)).

2.2.3 Calibration and simulations strategy

The model is run through a numerical solver with a calibration relying on previous calibrations detailed in the FFSM literature (Caurla et al., 2010; Caurla, 2012). Values on supply and demand parameters come from sectorial analysis of the French forest sector (Lenglet et al., 2017; Montagné and Niedzwiedz, n.d.) and the recycling sector (Copacel, 2016; ADEME and Bio by Deloitte, 2017). Price elasticities of supply and demand come from the relative literature mentioned above (Lee and Ma, 2001; Buongiorno et al., 2003; Mansikkasalo et al., 2014).

As the literature shows conflicting results for the cross-elasticity between recycled and wood-pulp, three cases are designed with different values to examine the sensitivity to this

parameter. As there seems to be a common consensus toward low values (Lee and Ma, 2001; Lundmark and Olsson, 2015), we introduced low substitution ($\eta = -0,2$) and low complementarity ($\eta = 0,2$) between transformed products. A third case is also explored with an optimistic hypothesis on substitution, considering a high negative cross-elasticity ($\eta = -1$). As we do not investigate asymmetries between those products, we use the same crossed elasticity $\eta = \eta_{recy,pulp} = \eta_{pulp,recy}$. The paper and cardboard industry encompasses a wide variety of different products, which creates various types of substitutability/complementarity between virgin and recycled pulp. Thus the elasticity we use should be understood as a synthetic indicator of those many types of relationships for the various products within the industry.

GHG emissions linked to the transformation activity of the pulp industry are included in the carbon balance module. Based on the LCA of ADEME and FEDEREC (2017), we assign to the recycling sector emissions of collection, sorting and avoided end of life. Besides, there is a difference between cardboard and paper recycling (cardboard pollutes more): we take into account this difference by creating a weighted coefficient relying on the share of French production, around 40% paper and 60% cardboard (ADEME and Bio by Deloitte, 2017; Lenglet et al., 2017). The resulting GHG coefficients are 350 kg CO_2eq/t for wood-pulp and 530 kg CO_2eq/t for recycled pulp. Note that these coefficients relate to the paper/recycled paper industry and do not account for effects on the rest of the forest sector, which is in fact the purpose of this work.

Our objective in this paper is to assess the effect of paper recycling on the forest sector, and especially its GHG balance. To this end, we test four scenarii with different recycling intensities: a baseline, lower increase in recycling, medium increase and higher increase. For these increases we use the proxy of the transformation cost c_{recy}, as shown in Table 2.1. This change in costs of the recycling process can come from different sources: a technological breakthrough making the recycling technology broadly available (Lafforgue and Rouge, 2019) or policy choices focused on subsidizing recycling (Palmer et al., 1997) As explained above,

we set out this simulation in three different cases: low complementarity, low substitutability and high substitutability, and we compare scenarii with `baseline`.

Scenario name	`baseline`	`low`	`med`	`high`
c_{recy} (€/m^3)	70	50	30	10

Table 2.1: Tested recycling scenarii

2.3 Impact of paper recycling on the carbon balance

The model is solved numerically over a period of 67 years (from 2013 to 2080). It computes supply/demand/prices of products, as well as resource dynamics and a detailed carbon accounting. In the following part, yearly results, averaged over the simulation period, are given.

2.3.1 Impacts on the pulp industry

What we observe for the pulp sector is the impact on production, given a cost reduction. This impact is direct for the recycling industry, and indirect for wood-pulp. The wood-pulp industry is indeed affected through the cross-elasticity between the two products, with a higher demand in the case of complementarity, a lower with substitution. However, in any case, this absolute change in wood-pulp demand is smaller than the absolute increase of demand for recycled pulp. This is illustrated in Table 2.3 showing the evolution in demand for pulp between `baseline` and `med` in 2030. For this reason, Table 2.2 shows that emissions in the pulp industry are always higher with a cost reduction: increased recycling prevails. Besides, from the same reasoning, we show that increasing substitution (from $\eta = -0,2$ to $\eta = -1$) leads to a smaller emissions difference.

One interpretation of complementarity comes from paper and cardboard industries introducing fixed proportions of wood-pulp and recycled pulp in their transformed products, according to the use and the level of quality they look for. There is indeed a lower quality of

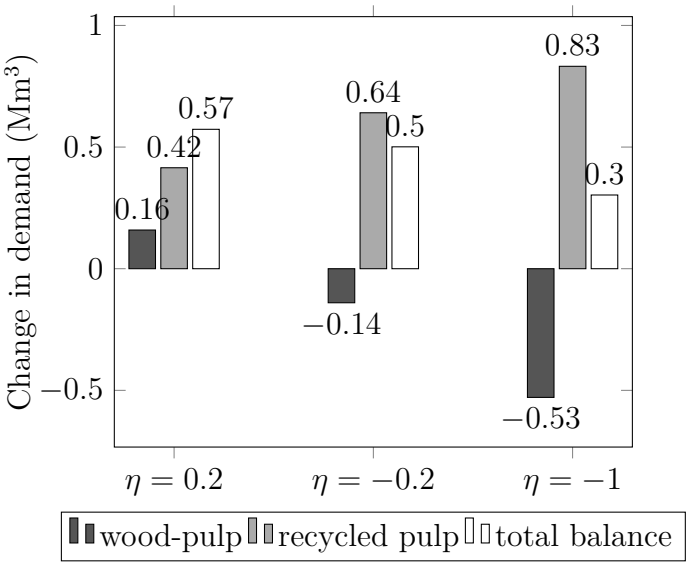


Figure 2.3: Changes in demand for pulp between `baseline` and `med` for substitutability ($\eta < 0$) and complementarity ($\eta > 0$) in 2030

Scenario	`low`	`med`	`high`
Emissions variations ($MtCO_2eq$) of the transformation industry			
$\eta = 0,2$	0,140 (+4%)	0,231 (+6%)	0,315 (+8%)
$\eta = -0,2$	0,127 (+3%)	0,225 (+6%)	0,315 (+8%)
$\eta = -1$	0,105 (+3%)	0,192 (+5%)	0,265 (+6%)

Table 2.2: Changes in emissions linked to transformation in the pulp industry (compared with scenario `baseline`) - yearly average for 2013-2080

the fiber for each recycling process, thus imposing a limit of cycles and use (ERPC, 2015). With this hypothesis, recycling fosters the demand for wood-pulp. In contrast, when products are substitutes, introducing recycled pulp implies a stronger competition for wood-pulp, thus decreasing the quantities. High substitution also reflects technical possibilities, where recycled pulp has an equivalent quality and consumption pattern as wood-pulp.

The purpose of our analysis here is to go beyond this straightforward economic result of higher direct GHG emissions. Structural links between the pulp industry and the rest of the forest sector are expected to induce economic and carbon impact when recycling is more prevalent. These results will figure out wether the wood sector can alleviate the increased climate impact of the pulp industry.

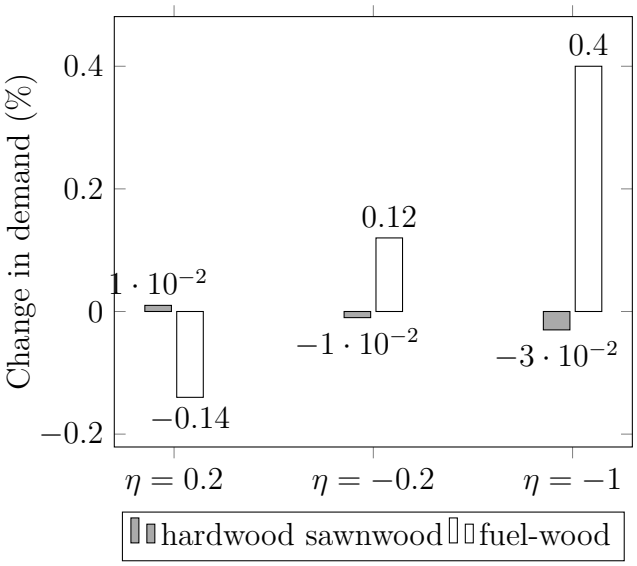


Figure 2.4: Relative changes in demand for other wood products between `baseline` and `med` in 2030

2.3.2 Impacts on other forest products

The structure of the model, depicted in Figure 2.2, shows the link between the pulp industry and the rest of the forest products sector. Wood-pulp is indeed a transformed product of industrial wood, also used for pannels manufacturing and energy purposes. This induces a competition for the resource (the primary product). For this reason we can expect an effect of increased recycling on those products. At a wider scale, industrial wood competes with round-wood primary products for the forest resource. We can also expect an impact on these products. However, results from the work of Buongiorno et al. (2003) lead us to suspect very small effects in terms of relative changes in supply and demand of wood products.

Table 2.4 shows indeed very small relative effects on wood products. Fuel-wood, which is in direct competition with the pulp industry for primary industrial wood, shows higher impacts (from $-0,14\%$ to $0,4\%$). On the other hand, sawnwood shows negligible impacts as it is a transformed product from roundwood and thus far from the pulp industry (see Figure 2.2).

We also need to assess the climate impact that derives from these economic impacts. It regroups material and energy substitution related to the use of wood products, as well

as emissions linked to forest exploitation (forest operations and transport). As emissions of forest operations are negligible compared to energy and material substitution, they are not displayed in Table 2.3[6]. While they remain low in terms of relative changes, these sequestration mechanisms can potentially offset additional emissions of pulp transformation. This is especially true for energy substitution, derived from the differences in fuel-wood usage. It does not compensate higher emissions from transformation detailed above, but when pulps are substitutes, they amount 10% of the emissions variation displayed for the pulp in industry (in Table 2.2). However, when pulps are complementary products, sequestration is lower, thus deepening the negative climate impact of pulp recycling.

Scenario	`low`	`med`	`high`
Energy substitution ($MtCO_2eq$)			
$\eta = 0,2$	-0,012 (-0,08%)	-0,019 (-0,14%)	-0,030 (-0,18%)
$\eta = -0,2$	0,010 (+0,07%)	0,017 (+0,12%)	0,023 (+0,16%)
$\eta = -1$	0,033 (+0,23%)	0,055 (+0,39%)	0,072 (+0,51%)
Material substitution ($MtCO_2eq$)			
$\eta = 0,2$	-0,004 (-0,01%)	-0,006 (-0,02%)	-0,008 (-0,03%)
$\eta = -0,2$	0,003 (+0,01%)	0,005 (+0,02%)	0,007 (+0,03%)
$\eta = -1$	0,009 (+0,04%)	0,016 (+0,06%)	0,022 (+0,09%)

Table 2.3: Changes in carbon substitution for other wood products (compared with scenario `baseline`) - yearly average for 2013-2080

2.3.3 Impacts on the forest resource

The specification of resource impacts relies first on inventoried and non-inventoried biomass (branches and roots), as well as wood products (however this last one is negligible in terms of absolute values). We see from Table 2.4 that in the case of complementary pulp products, effects on stocks exacerbate the negative climate effect of increased recycling. Substitution in contrast leads to higher sequestrated carbon with up to a $1,5\%$ increase in the most optimistic case (high substitution $\eta = -1$ and scenario `high`).

[6]Unlike Table 2.2 where we show GHG emissions amounts, we produce in Table 2.3, 2.4 and 2.6 GHG substitution and sequestration, where a negative sign means more emissions, and a positive sign more mitigation.

Scenario	`low`	`med`	`high`
Carbon sequestration pool ($MtCO_2eq$)			
$\eta = 0,2$	-0,143 (-0,2%)	-0,232 (-0,3%)	-0,310 (-0,4%)
$\eta = -0,2$	0,141 (+0,2%)	0,245 (+0,4%)	0,336 (+0,5%)
$\eta = -1$	0,453 (+0,7%)	0,781 (+1,1%)	1,027 (+1,5%)

Table 2.4: Changes in carbons stocks (compared with scenario `baseline`) - yearly average for 2013-2080

Changes for the forest resource are also reflected in landscapes evolutions, as increased recycling has impacts in terms of forest management after some decades. Results for the volume of resource in Table 2.5[7] are coherent with results on carbon stocks for the sector, with an increase when substitutes and a decrease when complements. We can notice that this increase (resp. decrease when $\eta = 0,2$) mainly concerns the broadleaved cover. Changes in forest areas are also noticed. While the overall forest cover remains stable in France with no evolution in total, the ratio of species evolves with more broadleaves (resp. less when $\eta > 0$) and less conifers (resp. more). These results are in relative values limited, reflecting the overall stability of the forest sector. However, these landscape impacts can lead to modifications in terms of ecosystem services from the forest sector, with an overall reduction in some cases (Gamfeldt et al., 2013).

	Broadleaves	Conifers	Total
Forest areas (Mha)			
$\eta = 0,2$	-1,980 (-0,02%)	1,730 (+0,04%)	0 (0%)
$\eta = -0,2$	1,749 (+0,02%)	-1,595 (-0,04%)	0 (0%)
$\eta = -1$	5,853 (+0,07%)	-5,420 (-0,14%)	0 (0%)
Forest volumes (Mm^3)			
$\eta = 0,2$	-6,318 (-0,20%)	-1,695 (-0,18%)	-9,106 (-0,19%)
$\eta = -0,2$	5,264 (+0,17%)	1,297 (+0,14%)	7,441 (+0,16%)
$\eta = -1$	17,637 (+0,57%)	5,121 (+0,55%)	26,012 (+0,56%)

Table 2.5: Changes in forest volumes and areas between `baseline` and `med` in 2070

[7]Note that we do not display results for lands covered with a mix of conifers and broadleaves.

2.3.4 Global carbon impact on the forest sector

Results on sequestration described above combined with changes in GHG emissions give the total CO_2 balance of the forest sector. Reduced sequestration with complementary pulp products lead to important diminutions in the global balance, as we find a reduction between $0,299$ and $0,660$ $MtCO_2eq$/year (Table 2.6). On the other hand, with low substitution ($\eta = 0,2$) we find an equilibrium between additional emissions and sequestration, with slightly positive values for the carbon balance, although negligible compared with the use of other values for elasticity η. This effect is indeed higher when we examine the optimistic case of a high substitution between pulps, up to $0,8\%$ of the total carbon balance of the French forest sector for scenario `high`.

While these global results are relatively small for the scale of the forest sector, it shows the central role of substitutability and complementarity to understand the environmental impact of paper recycling. With complementarity, reduced recycling costs and thus increased production of pulp overall (as shown in Table 2.3) leads to a higher usage of the virgin resource, coupled with increased forest utilization. On the other hand, substitution between pulps leads to lower forest exploitation, then higher sequestration compensating emissions of the pulp industry. It is interesting to notice that the forest sector only alleviates an increased carbon impact of the recycling sector when added recycled products are substitutes to wood-pulp. Besides, this is true for both substitution mechanisms (Table 2.3) and sequestration mechanisms (Table 2.4).

Scenario	`low`	`med`	`high`
	Global carbon balance ($MtCO_2eq$)		
$\eta = 0,2$	-0,299 (-0,3%)	-0,489 (-0,5%)	-0,660 (-0,6%)
$\eta = -0,2$	0,027 (+0,03%)	0,041 (+0,04%)	0,050 (+0,05%)
$\eta = -1$	0,390 (+0,4%)	0,660 (+0,6%)	0,857 (+0,8%)

Table 2.6: Changes in global GHG balance (compared with scenario `baseline`) - yearly average for 2013-2080

2.4 Conclusion

With this work, we contribute to the already existing literature focusing on recycling of wood products. The question of wether recycling has a positive or negative carbon impact is crucial when issues relative to climate change and circular economy are more and more prominent. Based on the present situation of the French forest sector, we examine what would be the impact of increased recycling, simulated through a cost reduction in the sector. The use of FFSM enables us to have a broader perspective for the climate impact. The market module and the recursive structure expands a standard LCA analysis with both economic impacts on other wood products sectors and variations in carbon substitution and sequestration.

First, we show the expected impact of recycling on wood-pulp industry. Expected qualitative results regarding the complementarity or substitution of both products are found, with respectively a positive or negative impact on demand. Beyond this economic outcome, GHG emissions of the pulp sector are always increasing, thus raising the question of their possible mitigation through the rest of the forest sector.

Second, we find small effects on other wood products. This is expected as the economic model combines small price-elasticities of products competing for timber resources and small cross-elasticities between virgin and recycled. The model produces small changes for transformed products from industrial wood (such as fuel-wood and pannels), with a higher demand when pulp recycling is a substitute (and vice versa when complements). However this effect is negligible for other wood products such as sawn wood and plywood. This involves small effects on carbon substitution mechanisms (material and energy).

Finally, our study of global carbon sequestration highlights small relative effects overall (between $0,4\%$ and $0,8\%$ in the additional carbon balance of the sector). However in terms of absolute evolution of the net sequestration potential, we observe a positive impact when considering substitution in the pulp industry, and a negative impact when considering complementarity. The already existing literature on the econometrics of paper and cardboard recycling shows that strongly substitute products would be a very optimistic scenario, while

low complementarity is a more realistic one. Our results show that the development of recycling (and more generally of a circular economy) relies on the substitutability of natural capitals to be environmentally efficient (here virgin and recycled capitals)(Ayres, 2007).

Our work has several implications in terms of public policies. First, in a global situation than leans toward weak complementarity between virgin and recycled paper, the idea suggesting that recycling and circular economy is a good solution against global warming should be taken cautiously: in this situation, an increase in the use of recycling does not come along with savings of wood resources. It then appears than sobriety in the use of paper and cardboard is an inevitable objective to reduce GHG emissions from the sector. Second, there is a strong heterogeneity within the paper and cardboard industry: some products imply complementarity between recycled and virgin material, while other ones allows stronger substitutability. It follows that the sobriety objectives should be oriented toward products where complementarity is stronger. In contrast, for products where substitution is possible, the objective of sobriety may be less stringent and the use of recycling produce some climate benefits.

This sectoral analysis on the economic and carbon impacts of paper and cardboard recycling should be pursued with further investigation, including sensitivity analysis of other parameters in the model, such as recovery rates and the introduction of policies promoting recycling or a lower carbon footprint.

CHAPTER 3

AN ASSESSMENT OF THE EUROPEAN REGULATION ON BATTERY RECYCLING FOR ELECTRIC VEHICLES

* * *

Electric mobility is deemed necessary to cut CO2 emissions in the transport sector but the industrial and environmental impacts of lithium-ion battery manufacturing are controversial. A recent regulatory proposal from the European Commission introduces the obligation to attain a series of minimum thresholds of recycled materials for the new batteries to be manufactured after 2030. This chapter discusses the conditions required for that obligation to be fulfilled. Battery lifetimes are the most crucial parameters for recycling potential. This result could jeopardize avenues for extending battery lifetimes such as second-life battery usage. We recommend lower thresholds to improve the regulation credibility and the integration of other objectives in the regulation in order to address the lifetime of batteries.

* * *

3.1 Introduction

In a recent proposal for a regulation, the European Commission (EC) has introduced recycling as a key element for the development of an industry for electric vehicle (EV) and especially the battery industry (European Union: European Commission, 2020). It is supposed to be a central part of the European Union (EU) industrial strategy covering economic, social and environmental goals. It establishes a legislative framework for a more sustainable life-cycle of lithium-ion (Li-ion) batteries. One of its main initiative is the introduction of minimum thresholds for recycled material sourcing in new batteries.

This chapter aims to assess the conditions under which these thresholds may physically be reached. Our analysis of these conditions follows two steps. First, we set up a framework to qualitatively discuss the main characteristic trends of the battery sector and how these should affect sourcing from recycled materials. It allows us to separate the different mechanisms driving the amount of recycled material that can be incorporated in new batteries. The share of demand that can be covered by recycled materials in new batteries will be reduced by higher demand growth of electromobility, lower recycling efficiencies, longer lifetimes of batteries and technological changes that increase material use. Second, we estimate the sensitivity of recycling potentials to these mechanisms through a quantitative model. We set up a dynamic material flow analysis, calibrated using data from different reports from the International Energy Agency (IEA) and from the academic literature. For all materials, we find that battery lifespan is the most sensitive parameter driving the thresholds proposed by the EC. On the contrary, the growth rate of demand in batteries appears to bring little sensitivity. Regarding battery technological change, we find that only the recycling potential of cobalt is likely to be significantly impacted. Furthermore, the improvement of recycling processes is crucial to guarantee the sourcing in recycled lithium. Overall, our results indicate that the thresholds of recycled materials proposed by the EC may be difficult to comply with. At the light of this result, we discuss the eventual conflict in environmental goals that these minimum thresholds could cause. For instance, we highlight a trade-off between this

instrument and the development of a second life for batteries.

More generally, the proposal of regulation from the EC is motivated by the ongoing transformation of the automobile industry. Indeed, the penetration of EVs has grown steadily for a few years, with EU market shares increased from 0.4% in 2014 up to 4.6% in 2020. EVs have been advocated as one of the main technological option to decarbonize the transport sector and to decrease urban air pollution. This led government to provide a large policy support for electromobility. In parallel, Li-ion batteries - the central components of EVs - have been subject to remarkable technological progress. As a result, their costs have fallen from USD 1 100 per kilowatt-hour (kWh) in 2010 to USD 156/kWh in 2020 (Nykvist et al., 2019; IEA, 2020). In this context, the growing importance of the Li-ion batteries also brings sustainability and strategic concerns. These include the carbon footprint of the life-cycle and waste management (Dai et al., 2019). These also include concerns regarding the availability of raw material, social impacts of the metal extraction as well as supply chain risk management (Hache et al., 2019; IEA, 2021b). Since 2008, the EU has taken into account this component of its industrial sectors. It mainly focuses on sustaining the access to those materials in the EU and worldwide, and developing recycling as well as resource efficiency. To address these issues, the EU has developed several initiatives, such as the European Battery Alliance[1] and the Raw Materials Initiative.[2] The proposal of the EC intends to bring a legislative framework to these initiatives.

Beyond the discussion of the regulation proposal of the European Commission, we contribute to two main strands of literature. First, we relate to the literature exploring the material usage for the development of electromobility. Ballinger et al. (2019) explore the risks for the EV sector related to material supply, in particular for graphite, lithium and cobalt used in batteries. Xu et al. (2020) and Sato and Nakata (2019) propose a long term analysis of the material demand for batteries. While the former show that the role of recy-

[1]The European Battery Alliance is a consortium of industrial actors that aims at developing a European battery industry (https://www.eba250.com/).

[2]https://ec.europa.eu/growth/sectors/raw-materials/policy-and-strategy-raw-materials_fr

cling remains minor while the market is in expansion, the latter find between 30 and 50% of mineral being supplied by recovery in 2035 in Japan. More generally, some articles address the broad topic of mineral criticity in the energy transition. A thorough literature study from Watari et al. (2020) covers the criticality of a wide range of material up to 2050. They highlight the lack of both socio-environmental analysis and inclusion of recycling strategies. While doing so, they link metal availability challenges to the key technologies of the energy transition. In the same spirit, Seck et al. (2020) examine the importance of copper use in the transition. They test different scenarios in a world model representing transport and energy sectors and find that the increase in copper demand could lead to an important stress on known mining resources. They use the same model for lithium and cobalt demand (Hache et al., 2019; Seck et al., 2022). While they do not find the same stress on mineral resources as for copper, they do highlight the risks caused by geographically concentrated resources, geopolitical strategies, lack of investments and environmental consequences of extraction. Helbig et al. (2018) also address risks in the supply chain of material used in Li-ion batteries. They compute a level of risk for a range of battery technologies, depending on their mineral composition. From the grey literature, reports from the IEA (IEA, 2020, 2021a) present the global trends of EV markets. They also tackle issues of metal scarcity in the energy transition, highlighting the important role that could be played by recycling in the following years in order to ensure the sustainability of material supply chains IEA (2021b). We contribute to this literature by disentangling the key-mechanisms of recycling potentials and by estimating the relative magnitude of those effects. We also focus on the European context.

Second, we relate to the literature on the aspects of the end of life of EV batteries. A review from Lv et al. (2018) lists the different recycling technologies for lithium, and discusses technological, environmental and economic perspectives for pyro-metallurgical and hydro-metallurgical processes. Other authors have investigated on the possibility of extending battery lifetimes through the re-utilization of used batteries. Such second life of batteries

could indeed provide stationary storage for the power grid and thus reducing their costs (Few et al., 2018; Gur et al., 2018; Martinez-Laserna et al., 2018; Wu et al., 2020). We contribute to this literature by showing potential incompatibilities when considering at the same time recycled sourcing of materials and other sustainability objectives such as the reuse of EV batteries.

The rest of this chapter is organized as follows. Section 3.2 presents the proposal of the EC and its context. Section 3.3 gives qualitative insights on the drivers of the level of recycled materials content from used batteries. Section 3.4 and 3.5 respectively present the methods and the results of the quantitative analysis. Sections 3.6 concludes.

3.2 Proposal of the European Commission

The current EU regulatory framework on batteries is established by a directive from the European Commission in 2006 (Council of European Union, 2006).[3] This directive aims at providing common rules across the EU market that would limit environmental impacts of battery wastes.[4] It distinguishes three kinds of battery types to be regulated: portable (electronic equipment), automotive (for starting, lighting and ignition) and industrial (traction for vehicles and other industrial applications). The directive applies from manufacturing to end of life. It enforces the extended producer responsibility principle. More precisely, it defines targets for the collection of portable battery wastes,[5] and maximum contents of cadmium and mercury in new batteries.

In the late 2020, the EC has proposed a new regulation for the regulation of batteries that would replace the 2006 directive (European Union: European Commission, 2020).[6] Indeed, the disruption of Li-ion batteries fueled by the development of electromobility has outdated

[3]The 2006 directive is not limited to Li-ion batteries, but the whole battery sector.

[4]Targeted sources of pollution involved cadmium, mercury and lead

[5]25% by 2012, 45% by 2016

[6]Unlike directives, regulations are automatically applied in EU members, without any transposition into national law. It results in a uniform effect for all the Union, in line with the objective of creating a coherent industrial sector at EU scale. See https://europa.eu/european-union/law/legal-acts_en for more details.

made the former directive. Within the context of EU's Green New Deal, this proposal joins other measures pursuing the development of the sector in Europe, such as the initiative of the European Battery Alliance. It aims at providing a legislative framework that would gather industrial, strategical and environmental objectives. The proposal of the EC defines several environmental objectives that spread on the period 2025-2035. These objectives comprise the development of recycling, the limitation of the carbon footprint and the transparency of the supply chain.

		Cobalt	Copper	Lead	Lithium	Nickel
Minimum recycled content in new batteries	2030	12%	NA	85%	4%	4%
	2035	20%	NA	85%	10%	12%
Recovery rate from scrapped batteries	2026	90%	90%	90%	35%	90%
	2030	95%	95%	95%	70%	95%

Table 3.1: Targets on battery recycling of the 2020 EC proposal

This chapter focuses on the objectives regarding recycling. The main elements from the proposal establish minimum recycled contents for several materials in new batteries. These constraints only apply to cobalt, lithium and nickel for the Li-ion technology, and lead for other battery technologies. Minimum thresholds are defined in 2030 and 2035. Table 3.1 displays them. The proposal also defines mandatory target levels of recovered materials from scrapped batteries for cobalt, copper, lead, lithium and nickel, as displayed in Table 3.1. It is expressed in shares of these minerals as well as on total weight of the batteries.[7]

To the best of our knowledge, most of the proposal and in particular the minimum thresholds on recycling come from a feasibility study ordered by the EC (European Union: European Commission, Oeko Institut., Ramboll. and Umweltbundesamt., 2021; European Union: European Commission, Trinomics. and Oeko Institut e.V., 2021). They present a number of measures to improve the 2006 directive. They are based on a modelling assessment complemented with interviews with stakeholders. To compute the thresholds, authors set up a material flow model to assess the availability of minerals to be recycled in the upcoming

[7]After 2025, 75% of lead batteries, 65% of Li-ion batteries and 50% of other battery waste.

years. By doing so, the proposed thresholds later used by the EC are set to match available materials for recycling. It appears that the proposed minimum thresholds of recycled contents match high estimates considering the availability of recycled material.[8] Furthermore, the impact of different parameters stemming from the technological and economical context given by IEA reports are not quantitatively discussed in the report (IEA, 2020, 2021a,b). In this chapter, we propose to fill this gap by investigating in particular the impact of demand growth, lifespan of batteries, recycling efficiency and technological changes.

3.3 Qualitative Insights

Having clarified the European context, we now provide the intuitions that will serve the reasoning presented in the quantitative sections which follow. Assessing the minimum thresholds of incorporated recycled materials given by the EC proposal can be done by computing the maximum ratio of recycled waste material over demand (RMD ratio hereafter).[9] As in the impact studies that inspired the EC proposal, such ratios can be estimated with complex dynamic material flow analysis, which requires a large number of hypothesis and data. We argue that a simplified version of such models is useful to draw the essential drivers of this dynamics.

We compute the RMD ratio in a two-period model ($t = 0, 1$). Each period, a quantity Q_t of batteries for electric vehicles are produced (in kWh), given a technology mix that requires an average intensity $w_{k,t}$ for each material k (in kg/kWh). At period 1, a quantity $P_{0,1}Q_0$ of battery is scrapped. These wastes are recovered and recycled with an overall efficiency $\epsilon_{k,1}$. For each material k, we define $RMD_{k,1}$ as the ratio of recycled material from battery wastes over demand at period 1. Such ratio can be decomposed as follows:

[8] Note a lower recovery rate for lithium. Indeed, the current state of lithium recycling technologies make it harder to recover it (a baseline of 10% in 2018 according to European Union: European Commission, Trinomics. and Oeko Institut e.V. (2021), matching other studies such as (Hache et al., 2019) mentioning low effective rates), thus involving lower recovery targets than nickel and cobalt.

[9] Such ratio corresponds to the maximum amount of material that can be recycled from yearly scrapped batteries and incorporated in new manufactured batteries.

$$RMD_{k,1} = \underbrace{P_{0,1}}_{\substack{\text{Scrapping} \\ \text{probability}}} \underbrace{\frac{Q_0}{Q_1}}_{\substack{\text{Demand} \\ \text{change}}} \underbrace{\frac{w_{k,0}}{w_{k,1}}}_{\substack{\text{Technological} \\ \text{change}}} \underbrace{\epsilon_{k,1}}_{\substack{\text{Recycling} \\ \text{efficiency}}} \tag{3.1}$$

Such decomposition, shown in equation (3.1) involves four terms, two of which being material-specific, while the other two are material-neutral.

- $P_{0,1}$ is the fraction of battery turned into waste and scrapped between the two periods. It is mainly linked to the average lifetime of batteries through the stochastic process of battery degradation. In particular, this fraction is reduced with higher average lifetimes, which can be caused by technological improvements or by the re-utilisation of used battery for other purpose.

- ratio Q_0/Q_1 of battery capacity demand at period 0 over period 1 relates to the evolution of demand for battery capacity. The larger is the development of electromobility,[10] the lower this ratio is. This ratio is supposed to be large during the uptake of electromobility, and converge to one as fossil fuel cars are phased out.

- ratio $w_{k,0}/w_{k,1}$ in material intensities k in period 0 over period 1 relates to the technological change of batteries. Such ratio decreases as battery technology requires lesser quantities of material k. Material footprint may decrease as a whole and there may be material substitutions, as displayed later when discussing our data (see Tables 3.3 and 3.4).

- $\epsilon_{k,1}$ is the fraction of recovered material from battery wastes. It encompasses both the collection of battery wastes and the efficiency of recycling processes. Contrary to the other components, this parameter may be the most crucial in economic terms as

[10] Such development includes both the number of vehicles and battery size. Battery size will increase with the autonomy of electric vehicles, and should converge once autonomy of electric and thermal vehicles become close.

collecting a high share of disposed batteries or recycling at higher rates can involve very large marginal costs on recycling. Also, it has been indicated that it can be difficult to reach high recycling efficiencies for some material, such as lithium, with the same recycling process. One recycling process can foster the recovery of a material at the expense of another (Xu et al., 2020).

This decomposition can be illustrated with a numerical example. Let's consider a 10 years time lapse. During this period, consider that demand quintuples ($Q_0/Q_1 \approx 0.2$), material intensities remain steady $w_{k,0}/w_{k,1} \approx 1$, recycling is almost perfect ($\epsilon_k \approx 1$), and 40% of batteries produced in year 0 are scrapped in year 10 ($P_{0,1} \approx 0.4$). In this case, the ratio $RMD_{k,1} \approx 0.08$. Such ratio is in the range of the minimum thresholds given by the EC proposal.

3.4 Quantitative Analysis: Methods

3.4.1 Model

The previous section exhibited the main drivers for the ratios of recycled material over demand. This following section aims at quantitatively estimating these levels and confronting them to the EC targets. We model the material flow of Li-ion batteries in the EU at both extremities of the lifecycle with yearly periods corresponding to 2020-2035: the demand of batteries (by extension related to the market for EVs) and the waste aggregation following the end-of-life. Note that this model can be seen as a sophisticated extension of the decomposition previously discussed. The underlying variable for the flow of batteries is the aggregate flow $F_{k,t}$ (in kg) for a specific material k at year t:

$$F_{k,t} = \sum_i \sum_j Q_{i,t} m_{j,t} w_{j,k} \tag{3.2}$$

This flow is summed for all types of EVs i and all technologies of Li-ion batteries j. $Q_{i,t}$ is the aggregated capacity (in kWh) from EVs of type i sold each year, and is computed as the product of the number of vehicle of type i with their related average battery size. $m_{j,t}$ is the market share of a battery technology and $w_{j,k}$ is the material k intensity of a specific technology j (kg/kWh).[11] The definition of indexes $\{i, j, k, t\}$ is summed up in Table 3.2. Differences between Li-ion battery technologies mainly stem from the conception of the cathode.

Index	Name	Description
i	EV types	cars BEV, cars PHEV, vans BEV, vans PHEV[a]
j	Li-ion battery technology	ASSB, LMO, LFP, NMC333, NMC532, NMC622, NMC811, NCA+[b]
k	material	copper, lithium, nickel, manganese, cobalt, graphite
t	time	2020 to 2035

[a] BEV: battery electric vehicle; PHEV: plug-in hybrid electric vehicle
[b] ASSB: solid-state; LMO: Lithium-Manganese-Oxyde; LFP: Lithium-Iron-Phosphate; NMC: Nickel-Manganese-Cobalt; NCA: Nickel-Cobalt-Aluminium

Table 3.2: Indexes of the model

Each battery (i, j) in use has a probability $P_{i,j,\tau,t-\tau}$ to be scrapped, where τ is the year of production of the battery, hence $t-\tau$ is the age of the battery. This allows us to compute for each year t the flow of waste from production year τ for material k when no decay or recycling takes place:

$$B_{k,\tau,t} = \sum_i \sum_j V_{i,\tau} b_{i,\tau} m_{j,t} w_{j,k} P_{i,j,\tau,t-\tau} \tag{3.3}$$

The probability $P_{i,j,\tau,t-\tau}$ follows a Weibull distribution calibrated with minimum, maximum and most likely lifespans of Li-ion batteries. We use a Weibull distribution as the

[11] Contrary to the qualitative analysis, material intensity $w_{j,k}$ is not time related, as it relates to a specific technology whose market share will evolve through time.

literature shows that it fits well the lifecycle of durable goods (Xu et al., 2020; Elshkaki, 2005; Melo, 1999; Spatari et al., 2005).[12] Finally, we can compute for each year t the aggregated waste flow of material k:

$$W_{k,t} = \sum_{\tau=0}^{t-1} (B_{k,\tau,t} - B_{k,\tau,t-1}) \tag{3.4}$$

Finally, the ratio of recycled material over demand (RMD ratio) for each material k can be computed as follows:

$$RMD_{k,t} = \frac{\epsilon_{k,t} W_{k,t}}{F_{k,t}} \tag{3.5}$$

with $\epsilon_{k,t}$ the efficiency of the material recovery process. It includes physical collection of used batteries from used cars as well as the efficiency of recycling processes.[13]

Following the decomposition from the qualitative analysis, we elaborate several scenarii that aim at investigating the influence of demand evolution, battery lifespans, technological change and recycling efficiencies.

3.4.2 Data

The main source of data for the calibration of the model comes from recent IEA reports (IEA, 2020, 2021a). Projections of the EV market in 2025 and 2030 comes from (IEA, 2020). We use a quadratic interpolation to get the evolution of the total EV battery demand (in kWh) for the range 2020-2035, with the two different scenarii studied in IEA reports. Battery sizes are also calibrated using these reports. The Stated Policies Scenario (*STEPS*) relies on already existing and announced policies and their expected consequences on the development of electromobility. On the other hand, the Sustainable Development Scenario (*SDS*) is more

[12] We also computed our model with a gamma function, with very little difference in the results that do not alter the policy implications.

[13] Due to the lack of data, we neglect material losses from the use of batteries.

ambitious and built in order to meet the Paris Agreement objectives. The extension of the demand curve up to 2035 is made with the hypothesis that the EV fleet does not yet reach its final deployment. Figure 3.1 shows the projections for battery demand. Note that a growing divergence between scenarii occurs after 2025, resulting for the *SDS* scenario in a twice larger demand in 2030 than for *STEPS*, and even more for 2035. Comparing these two scenarii allows to assess the effect of demand dynamics, which was the first component of the decomposition of the qualitative analysis.

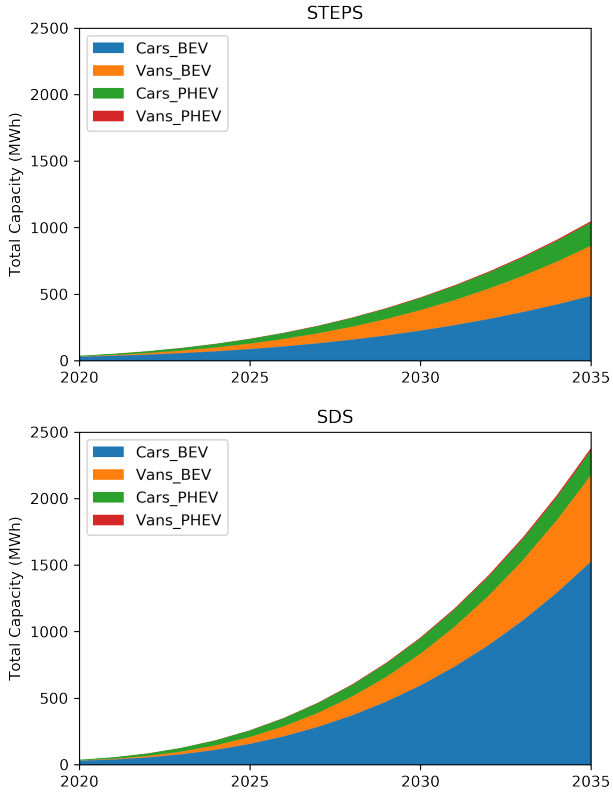


Figure 3.1: Projected EV batteries growth for two demand scenarii (extrapolated from IEA)

Material intensities of each battery cell technology $w_{j,k}$ are taken from IEA reports and (Xu et al., 2020) and shown on Table 3.3. We linearly interpolate projected market shares $m_{j,t}$ of battery technologies from the IEA (2021b) as shown in Table 3.4, given for up to 2040, consisting in the main technological scenario.

The projections of the IEA in Table 3.4 show a technological change that shifts from cobalt/manganese-based toward nickel-based batteries. This is caused by the phase-out of

NMC333 batteries in favor of NMC811 with higher nickel incorporation rates. Another key feature of this technological change is the rise of solid-state batteries (ASSB), with an important reduction in material use among which cobalt and nickel.

Battery Type	Lithium	Nickel	Cobalt	Manganese	Graphite	Copper
ASSB	0.174	0.000	0.000	0.000	0.000	0.293
LMO	0.107	0.000	0.000	1.400	0.867	1.040
LFP	0.093	0.000	0.000	0.000	1.027	1.067
NMC811	0.093	0.587	0.080	0.067	0.907	0.853
NMC622	0.107	0.533	0.187	0.160	0.907	0.880
NMC532	0.160	0.507	0.200	0.280	0.907	0.907
NMC333	0.160	0.347	0.360	0.307	0.907	0.933
NCA+	0.107	0.773	0.027	0.000	0.907	0.853

Table 3.3: Battery Composition for each battery type. Values are given in kg/kWh.

Scenario	Historic	*main*		*status quo*		*ASSB+*		*LFP+*	
Year	2020	2030	2040	2030	2040	2030	2040	2030	2040
ASSB	0	0.7	31	0	0	1.4	62	0.4	21.8
LMO	2	0	0	2	2	0	0	0	0
LFP	6	10	9	6	6	9.9	5	40	36
NMC811	15	60	53	15	15	59.6	29.2	40	37.3
NMC622	29	11	4	29	29	10.9	2.2	7.3	2.8
NMC532	22	7.6	0	22	22	7.5	0	5.1	0
NMC333	6	1	0	6	6	1	0	0.7	0
NCA+	20	9.7	3	20	20	9.7	1.6	6.5	2.1

Description of technological scenarii:
- *main*: Main scenario given used in IEA reports
- *status-quo*: market shares stay at the 2020 level
- *ASSB+*: Strong breakthrough of solid-state batteries (doubled market share)
- *LFP+*: Strong breakthrough of LFP batteries in 2030-2035 (four fold increase)

Table 3.4: Description technological scenarii with their respective market shares (in %).

In order to assess the impact of the evolution of material intensities, due to technological change, alternative scenarii are developed and summarized in Table 3.4, all considering a demand from the *STEPS* scenario. First, in scenario *status-quo*, market shares are maintained at their 2020 levels. Within it, battery manufacturing remains intensive in cobalt. Second, the scenario *ASSB+* simulates a large penetration of ASSB batteries. It leads to

an important decrease in material use, especially nickel, manganese and cobalt, due to the technological breakthrough. Third, the scenario *LFP+* is built to show an important market share of LFP batteries. It shows lower needs for nickel, manganese and cobalt. For the last two scenarii, proportions for other market shares are kept the same.

The Weibull distribution function is calibrated with the most likely lifetime for the battery, and minimum/maximum set at 99,7% of the distribution. There is a lack of knowledge regarding the lifespan of EV batteries, as the market is still at its early stage of development. For the specific case of electric vehicle, we find shorter lifespans in the literature, with 8 to 10 years (Casals et al., 2017, 2019). Lifespans of batteries are often high estimates in studies, with the example of Xu et al. (2020) who take an average of 15 years based on generic studies on automobiles. Our main scenario follows a most likely lifetime of 10 years. For sensitivity purposes, we also test scenarii with shorter lifespans, as described in Table 3.5, as well as longer as some emerging uses of batteries such as second life could postpone the year of recycled material availability (Martinez-Laserna et al., 2018). We use demand from the *STEPS* scenario for these ones.

Scenario	Minimum	Most likely	Maximum (99, 7%)
shorter lifetime	1	8	18
main	1	10	20
longer lifetime	1	12	22

Table 3.5: Description of scenarii on lifespans and other relevant parameter for the Weibull function of batteries (in years)

Recycling efficiencies $\epsilon_{k,t}$ are set with the targets of the proposal of the EC (Table 3.1), and linearly interpolated between 2020 and 2026, and between 2026 and 2030, and are constant after 2030. Note that the EC aims at a recovery rate for lithium going from 35% in 2026 to 70% in 2030, while it is around 10% in 2020 (European Union: European Commission, Trinomics. and Oeko Institut e.V., 2021), hence an optimistic seven-fold increase in a decade. We also test a scenario *low efficiency*, with demand from *STEPS*, where we maintain recycling efficiencies at their current 2020 levels, with 80% efficiency for both cobalt

and nickel, and 10% for lithium.[14]

3.5 Quantitative Analysis: Results

This section presents the results of the quantitative analysis. It focuses on the computation of the ratios of recycled materials over demand in new batteries (RMD ratios) up to 2035. First, it presents the evolution of these shares for each material for the two IEA scenarii *STEPS* and *SDS*. Then, it successively shows the impacts of battery lifespans, battery technological change and recycling efficiencies on RMD ratios.

3.5.1 Evolution of RMD Ratios and Effect of Demand Growth

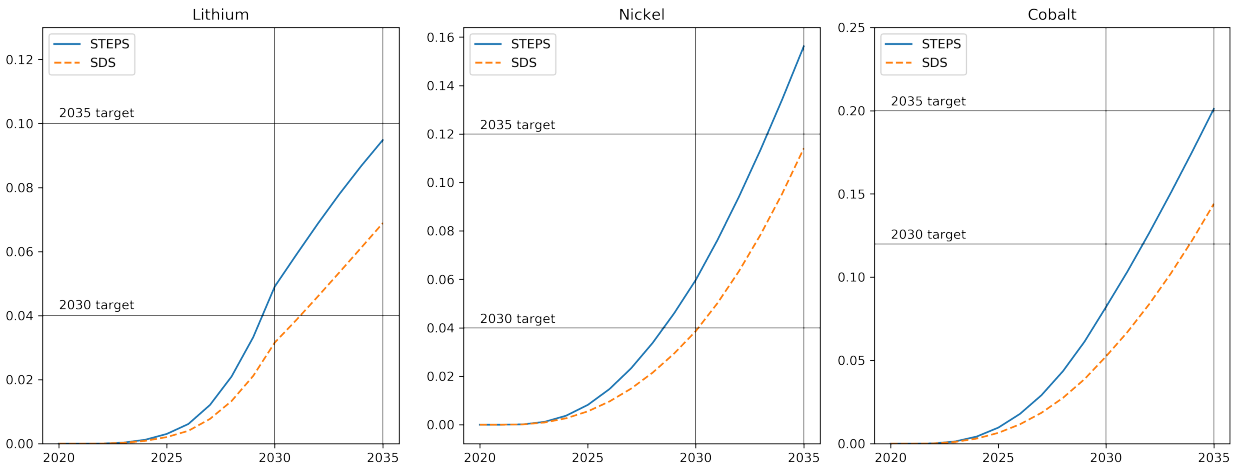


Figure 3.2: Ratios for Li, Ni and Co (policy objectives in dashed lines)

Evolutions of RMD ratios are displayed for lithium, nickel and cobalt on Figure 3.2. For both demand scenarii, all RMD ratios are increasing. It indicates that the amount of available waste that can get recycled grows faster than the demand in new batteries. Indeed, demand grows at a fairly constant rate, while available wastes grow faster due to the specificity of the degradation process, where the probability of becoming wastes grows up to the most likely lifetime, around ten years here, then decreases.

[14] As indicated as the 2020 baseline for material recovery rate in (European Union: European Commission, Trinomics. and Oeko Institut e.V., 2021).

Figure 3.2 also displays EC minimum thresholds for 2030 and 2035. These targets are only partially met. Regarding lithium, targets are only reached in 2030 under the *STEPS* scenario. It results from a combination of low recycling efficiency compared to other materials (70% after 2030) and the use of lithium intensive technologies. In the case of nickel, targets are met under *STEPS* scenario for both 2030 and 2035 targets, while under *SDS* they are barely met in 2030 and below the objective in 2035. Cobalt targets, which are much higher than for nickel and lithium, are in both cases not reached in 2030 and 2035, despite the early shift from cobalt/manganese to nickel. This transition makes relatively high amounts of cobalt available later in 2035 thanks to a decreasing demand, which explains that targets are almost reached for the *STEPS* scenario.

The comparison of *STEPS* and *SDS* scenarii highlights a conflict between a faster growth of electromobility and material sourcing from recycling. Faster demand growth in minerals makes it harder to achieve relatively high amounts of recycled inputs in new batteries. This indicates that while demand grows much faster in the SDS scenario than in STEPS, the difference between waste production growth is less significant.

3.5.2 Effect of the Battery Lifespan

	Lithium		Nickel		Cobalt	
Lifetime	2030	2035	2030	2035	2030	2035
8 years	**0,10**	**0,14**	**0,12**	**0.24**	**0.16**	**0.29**
10 years	**0.05**	0.09	**0.06**	**0.16**	0,08	**0,20**
12 years	0.02	0.06	0.03	0.09	0.04	0.13

Table 3.6: Ratios of recycled materials over demand (RMD ratios) for alternative battery lifetimes with the *STEPS* demand scenario. Values of RMD ratios above the respective EC minimum threshold are written in bold.

We perform a sensitivity analysis on the most likely lifespan of batteries, according to Table 3.5. Results are shown in Tables 3.6 for the *STEPS* demand scenario. RMD ratios are very sensitive to average lifetime changes. For a lifespan up to 12 years, available metals

for recycling barely reach 4% in 2030 for all metal inputs. Indeed, longer lifetimes impliy lower waste rates on the short run, as secondary materials are later available for recycling.

Oppositely, with shorter lifespans (8 years), meaning that batteries are more often replaced, EC targets can more easily be met.[15] Hence, the impact of battery lifespans seems much larger than the effect of demand growth, while differences between *STEPS* and *SDS* scenarii were up to a factor two impact on the demand. In the case of lifespans, a 20% increase or decrease (from 12 to 8 years) leads to much larger differences in RMD ratios.

This result has implications regarding the development of second life of batteries. In this scenario, EV batteries are being refurbished when they do not meet EV power standards and are used for stationary storage, which could increase the economic value of batteries (IEA, 2020; Martinez-Laserna et al., 2018). A well developed second-life sector significantly extends battery lifetimes, and therefore limits the availability of recycled materials for reaching their sustainability objectives. Furthermore, lifespans of batteries could be increased a lot more than 12 years according to Martinez-Laserna et al. (2018), with technical requirements and economic profitability being the main drivers of life durations. However, this demand for second-life should be nuanced as stationary energy storage is expected to be a minor share of the overall EU demand in capacity in the following years (Martinez-Laserna et al., 2018).

3.5.3 Effect of Technological Change

	Lithium		Nickel		Cobalt	
Scenario	2030	2035	2030	2035	2030	2035
status-quo	**0.04**	**0.10**	**0.06**	**0.13**	0.06	0.13
main	**0.05**	0.09	**0.06**	**0.16**	0.08	**0.20**
ASSB+	**0.05**	0.09	**0.06**	**0.19**	0.08	**0.24**
LFP+	**0.05**	**0.10**	**0.08**	**0.18**	0.11	**0.25**

Table 3.7: Ratios of recycled materials over demand (RMD ratios) for alternative technological scenarii with the *STEPS* demand scenario. Values of RMD ratios above the respective EC minimum threshold are written in bold.

[15] Here we examine this effect *ceteris paribus*, meaning we do not consider an impact of lifespans on demand growth of batteries.

After having discussed the influence of demand evolution and battery lifespans on the different RMD ratios, this section discusses alternative technological scenarii. Detailed in Table 3.4, each alternative scenario represents a characteristic trend, with demand following the *STEPS* scenario. Results show little differences in RMD ratios across scenarii as soon as batteries become less cobalt-intensive. This contrasts with the large sensitivity shown by battery lifespans.

As stated in the qualitative section, the effect of technological change is heterogeneous between materials. Indeed, all RMD ratios for lithium (resp. nickel) are in the range 0.04-0.05 (resp. 0.06-0.08) in 2030 and 0.09-0.10 (resp. 0.13-0.19) in 2035. In particular, despite the lithium intensity of ASSB batteries, similar RMD ratios are reached in 2035 for *ASSB+* and *main*, explainable by the early use of NMC333/532 with high lithium contents. The only deviation happens for the RMD ratio of cobalt under the *status-quo* scenario. In 2035, the RMD ratio is 0.13 compared to 0.20-0.25 for the other scenarii. Indeed, the *main* scenario as well as the *ASSB+* and *LFP+* scenarii show a significant decrease in cobalt intensity, especially after 2030. As this transition reduces its demand, RMD ratios for cobalt increase.

3.5.4 Effect of Recycling Efficiencies

	Lithium		Nickel		Cobalt	
Scenario	2030	2035	2030	2035	2030	2035
low efficiency	0.01	0.01	**0.05**	**0.13**	0.07	0.17
main	**0.05**	0.09	**0.06**	**0.16**	0.08	**0.20**

Table 3.8: Ratios of recycled materials over demand (RMD ratios) for an alternative recycling efficiency scenarii with the *STEPS* demand scenario. Values of RMD ratios above the respective EC minimum threshold are written in bold.

Finally, we also estimate the influence of recycling efficiencies in our results. As explained above, our *main* scenario is based on the EC targets for recovery rates in 2026 and 2030, with *STEPS* demand. They can include a very high increase, in particular for lithium (from 10% efficiency in 2020 to 70% in 2030). It implies key developments in terms of technology

and cost reductions. With scenario *low efficiency* (with *STEPS* demand too), we compute RMD ratios when recycling efficiencies do not change after 2020 (10% for lithium and 80% for both nickel and cobalt), thus remaining especially low for lithium . As expected, given the linearity of the recycling efficiency in the model (in equation (3.5)) important change occurs for lithium, with RMD ratios dropping to 0.01 (Table 3.8). To a lesser extent, RMD ratios for nickel and cobalt also drop, though recycling efficiencies were not expected to increase with the same magnitude.

3.6 Conclusion and Policy Implications

With its regulation proposal, the EC tries to cope with the fast-paced evolution of innovation and market development of Li-ion batteries. The ambition of the EC is to create conditions for the development of a local industry for electric vehicles while securing the different stages of its supply chain and limiting the life-cycle environmental impacts. This implies securing the sourcing of the necessary critical minerals to produce batteries. Developing a recycling industry would be a way to limit risks of geological scarcity and geopolitical tensions. This chapter discusses one of the main features of the proposal, which aims at enforcing the incorporation of minimum shares of recycled material in new batteries from 2030. Our approach is to disentangle the different key-assumptions for the calculations of such targets. Using a material flow model, we find that the targets given by the proposal are physically attainable under very specific assumptions on characteristics of battery technologies, recycling processes, and the evolution of demand. In particular, most of the targets given by the European Commission are not met for average battery lifetimes beyond 10 years. Keeping average battery lifetimes at 10 years might be unrealistic given the technological progress in the battery sector. To a lesser extent, we notice that meeting this target is even more difficult when the demand growth rate is high and recycling efficiencies are low. Heterogeneity between materials is also observed due to technological change and specific recycling

efficiencies.

This study sheds light on several consequences of EC regulation. First, the targets set by the EC are very likely to be too high to be realistic, given market growth, battery usage, and the expected technological framework. On the one hand, setting optimistic targets gives a powerful signal to industrial stakeholders, from battery and car manufacturers to the recycling industry. For instance, it guarantees producers of recycled material the existence of a demand for recycled input, thus encouraging a needed investment in the sector. On the other hand, setting unrealistic targets undermines the readability and stability of the regulation. We believe that too optimistic targets will severely increase the risk of needing downwards revisions, which would alter the credibility of the regulation. This study indicates that battery lifetime is the most sensitive parameter driving recycling. Hence, setting high targets could give incentives to decrease battery lifetimes. Indeed, shorter lifetimes would allow a faster circulation of material in the economy, thus increasing the potential use of recycled material. However, increasing lifetimes could be an objective in itself, as it would decrease material use.

Therefore, the policy recommendations of this study are twofold. First, we believe that the regulations should incorporate robust targets. Although it is stated that targets may be changed in the future, industrial actors of the sectors (battery and cars manufacturers, recycling industry) would need readable and credible regulations to make the right investments and technology choices. While this study indicates that the proposed targets may be difficult to reach, we believe it would be appropriate to propose lower targets for each material. Second, we believe the regulation could include other complementary indicators. Indeed, developing a sustainable battery sector should not rely on recycling and recycled content targets only. As our results indicate that battery lifetime is the main driver of recycled material incorporation in new batteries, we believe it would be adequate to set quantitative objectives on this matter. It could include precise targets on the lifetime of the battery, either regarding the lifetime in the electric vehicle only, or the total lifetime, which

could include eventual repurposing of used batteries.

Further research could complete our analysis in three directions. First, an analysis that incorporates strategies from the supply side to comply with the regulation can be made. For instance, battery wastes or recycled materials could be accumulated years before the start of the regulation and be incorporated in new batteries only once the regulation applies. There could also be strategies to meet recycling targets by extending the source of recovered material to other industries or geographical areas. However, this reasoning bears several limits: a significant part of total production of lithium and cobalt is already used for the battery industry, leaving relatively small amounts for recycling from other industries, especially with an exponentially growing sector; it is not in the spirit of the EC proposal which aims at regulating a specific sector, in this case the battery sector; it suggests coordination between different industries and the possibility of incompatible usage of materials; and, regarding environmental considerations, it would only result in a displacement of the material sourcing problem. Second, our sensitivity analysis could be enriched by accounting for the different links between the main parameters. For instance, larger battery lifetimes should also decrease battery renewal, hence new battery demand. Battery lifetimes should also differ between technologies. Third, the analysis could be improved by a resource economics approach, in order to assess how imposing recycling contents forces a demand for recycled material and how it affects prices on markets for primary and secondary materials.

CHAPTER 4

RECYCLING IN A SUSTAINABLE FRAMEWORK

* * *

The development of recycling, and more generally the concept of a circular economy, has the ambition to help developing more sustainable economic paths. However, sustainability economics also relates to equity issues in terms of consumption and environmental impacts. On this matter, literature on recycling does not rely on a sustainable criterion: in its vast majority, it relies on discounted utility, a criterion that does not favour equity. In this chapter, we study a recycling model with the maximin criterion. We characterize an optimal path and its shadow values, and we study a new Hartwick investment rule which now depends on waste accumulation and the use of recycling in production.

* * *

4.1 Context and motivations

The ambition of developing a circular economy is directly tied to the idea of developing a more sustainable economic paradigm. It is supposed to overcome the "extract-produce-consume-throw away" linear model. In that sense, the concept can be viewed as a mean for economic development to remain sustainable, as defined by the Brundtland Report (WCED, 1987). In this chapter, we aim to explain how recycling, in particular, is a tool to redefine sustainability objectives.

With recycling at its center, the development of a circular economy first appeared as a standalone goal in itself. Its formalization by Braungart and McDonough (2002), and the consequential "grey literature" that followed, drew on the idea of a circular economy as the objective of creating a closed loop system. This conception can be linked to closed systems studied by Ayres (1999) who defends the idea of a "total recycling" claiming that all material in the economy can be recovered, provided that a sufficient amount of energy is available. Ayres' work is influenced by Georgescu-Roegen (Georgescu-Roegen, 1970), and he refutes the limitation imposed by his "fourth law of thermodynamics" applied to resource use. This controversy relies on the boundness of resource productivity and thermodynamical limits, with both arguments being subject to criticism. An important criticism is that material entropy is not a consistent concept (Young, 1991; Couix, 2019). Georgescu-Roegen indeed extends entropy from the field of thermodynamics to mechanics: matter is subject to conservation but also degradation, creating irreversible waste and thus making perfect circularity unattainable. These observations raise questions regarding a normative approach of circular economy, stating: is it still relevant to address circular economy as an ideal economy, or as one solution along the road towards a sustainable economic model? With this in mind, one should ask what are the links between recycling (and circular economy on a wider scale) and sustainability. The literature review by Geissdoerfer et al. (2017) describes the relationship between those two fields. He highlights that most of the existing work adopts a conditional or beneficial relationship. This implies that a circular economy is

either a condition to develop a sustainable economic model or a part of the solution to get to such a state of the economy. This supports the idea of recycling not being a goal in itself but rather a way towards sustainability.

Examining the recycling industry (a central element for closing material loops), chapter one shows that while not being an objective per se, recycling can be a useful tool in reducing pressure on exhaustible resources and reducing the production of polluting effluents like waste and greenhouse gas (GHG). The underlying trade-off between consumption, exhaustion and externalities, all of those distributed through time, relates to the definition of sustainability in economics, as developed by Baumgärtner and Quaas (2010): the consideration of exploitation of resources while considering equality issues in terms of consumption and environmental degradation due to these activities.

In this chapter, we advocate for applying the objectives and tools of sustainability economics to the study of recycling, and, more broadly, the circular economy. We introduce a simple model based on the Dasgupta-Heal-Solow model with a maximin rule. We characterize the optimal path when using recycling, especially with its shadow values. We discuss the damage caused from waste in such a model, and we discuss the Hartwick investment rule. This now more complex resource management rule depends on waste accumulation and the use of recycling in production.

Current economic literature on sustainability proposes several criteria to define development trajectories where current consumption decisions impact future trajectories of the economy (Martinet, 2012). It addresses the issue of intergenerational equity by assigning preferences to different time periods. Most of the literature in resource economics and recycling in particular use discounted utility. However the use of discounted utility is criticized for being a "dictatorship of the present" according to Chichilnisky (1995), resulting in decreasing consumption paths. Among other criteria for developing consumption trajectories is the maximin which allows for maintaining a minimum sustainable consumption (Solow, 1974), provided that resource and capital are substitutable in production. It defines the

highest possible egalitarian path by maximizing the consumption of the poorest generation. An important underlying result that can be observed is the net investment needed in the economy for consumption to be sustained, with the notable Hartwick's rule at its center (Hartwick, 1977). This maximin criterion and the common analysis resulting from it will be the main focus in the modelling part of this chapter.

With regards to environmental aspects of sustainability, they are often taken into consideration through less equalitarian criteria like the previously mentioned discounted utility. In such an implementation, a price (direct or implicit) is placed on externalities. This is the method in our first chapter that also introduces a damage cost of waste accumulation as well as a cap on total GHG emissions. More ambitious criteria like maximin are less common, however. The notable examples of using this criteria have been in the context of environmental issues like amenities of natural capital (d'Autume and Schubert, 2008) and climate change (Stollery, 1998). Furthermore, in order to go beyond this sustainability paradigm called "weak sustainability", the concept of strong sustainability has been introduced (Neumayer, 2013). While weak sustainability relies on the objective of sustaining a certain level of utility, strong sustainability advocates that natural assets are essential to production, involving the need for their conservation. This central hypothesis insists that environmental quality should be maintained, thus not allowing pollution to go beyond certain thresholds. In this chapter, we discuss how recycling cannot be studied while ignoring this strong sustainability paradigm.

The rest of this chapter is organized as follows. Section 4.2 presents the literatures on recycling and on sustainability. Section 4.3 presents the maximin model of recycling and the results. Sections 4.4 concludes.

4.2 Recycling and sustainability

This chapter relates to several trends of literature: literature on sustainability and the maximin criterion; literature on the modeling of recycling and underlying sustainability issues; and, literature related to strong sustainability considerations in recycling.

Regarding weak sustainability considerations, we focus here on the maximin criterion. Primary work comes from Solow (1974), Hartwick (1977) and Dasgupta and Heal (1979). They describe optimal maximin paths in a modeling framework that considers capital and natural resources. Those standard-model paths have the property of being equalitarian, meaning that consumption remains the same for all generations. The discussion has been extended to the study of non-regular maximin paths, meaning that the consumption path is not constant, as opposed to the regular ones studied before. A taxonomy of possible regular and non-regular paths is discussed by Cairns and Martinet (2014) and examples of models describing non regular paths can be found in the work of Asako (1980) and Cairns and Tian (2010). Furthermore, existing literature examines maximin paths in different specific modeling setups to investigate sustainability: the inclusion of a climate constraint (Stollery, 1998); the addition of an amenity value for the natural resource (d'Autume and Schubert, 2008); the development of an economy with two reproductible assets (Cairns et al., 2019); and, an application to growth and fishery models with the goal of reconciling growth and sustainability (Cairns and Martinet, 2021). To the best of our knowledge, there is no work studying recycling with a maximin criterion. While some of the previously mentioned work shows that a positive value of consumption can be sustained forever, with assumed substitution between capital and the natural resource, we can view the addition of recycling as a way to improve this sustained well-being.

In the existing literature on resource economics with a recycling component, discounted utility has been used in the utility function in order to characterize the optimal extraction and recycling paths of a resource. Early works from Plourde (1972), Smith (1972) and Lusky (1976) determine the optimal allocation of labor between the extracting activity and the re-

cycled production. They include resource scarcity or environmental pollution, sometimes both. With an optimality criterion that favors the consumption of present generations, the underlying optimal path often leads to an exhaustion of the resource at the end of the program, sometimes asymptotically. In the model of Hoel (1978), the rate of resource extraction decreases until recycling happens and then reaches a stationary level in the long run. Hoel presents intergenerational inequalities in terms of consumption as early generations take advantage of the virgin resource extraction. Furthermore, welfare analysis do not always take into account how negative externalities impact differently across generations. In yet another approach, Huhtala (1999) introduces both damage from waste accumulation and negative effects of pollution on the renewal of the resource. This kind of literature often focuses on steady state analysis through either the absence of a resource constraint (Plourde, 1972; Lusky, 1976; Kuhn et al., 2003; De Beir et al., 2007) or the use of a renewable resource (Smith, 1972; Huhtala, 1999; André and Cerdá, 2006). Another explored topic is the link between growth and recycling, with models assessing wether or not recycling can facilitate reaching balanced growth paths by alleviating resource scarcity and/or externalities (Di Vita, 2001, 2007; Pittel et al., 2010; Sorensen, 2017; Lafforgue and Rouge, 2019). In addition to accounting for the effects of technical change as well as the standard substitution between resource and capital, they also includes the substitution between virgin and recycled resource as a means to sustain a growth path. The work from André and Cerdá (2005, 2006) shows that the elasticity of substitution between resources is central in the ability to respond to resource scarcity. They show that in the long run, relying on renewable resources is essential to sustain a non decreasing consumption path. Material limitations are also a takeaway from the work of Fagnart and Germain (2011). They demonstrate that even with recycling, growth based on a material output can only be transitional. Long term paths rely on what they call "qualitative growth", as material inputs either remain constant or decrease. Finally, recycling has recently been introduced in resource models in the context of the energy transition (IEA, 2021b). They explore specific scarcity issues for minerals used

in energy production (Rosendahl and Rubiano, 2019; Fabre et al., 2020; Chazel et al., 2020; Pommeret et al., 2021) or extraction paths with a climate change related constraint (chapter one). These models describe an optimal depletion path for the resource, with a discounted utility rule and finite or infinite horizons. The method leads to intergenerational inequities through decreasing extraction paths. Overall, recycling appears to be a sustainable process that is mainly used in an unsustainable framework by design. Having shown that studies on other criteria like maximin benefit from the inclusion of recycling, economic literature on recycling could also benefit from the addition of sustainability concerns when determining optimal paths of resource extraction and consumption.

Finally, the above mentioned literature on sustainability relies on what is called "weak sustainability" (Neumayer, 2013). Conclusions based on weak sustainability remain valid while it is assumed that natural resources can be substituted with man-made capital (Dasgupta and Heal, 1979). Oppositely, "strong sustainability" does not support this hypothesis. It argues that a minimum physical input is needed to sustain production. To this point, the work of Ayres (2007) - who has defended an optimistic view of total recycling (Ayres, 1999) - argues that substitution is limited in the short term for many technologies; thus, a strong sustainability paradigm should be adopted. Beyond the discussion on capital and natural resources, substitution also relates to a discussion on natural resources and recycled ones. It remains true for metals as recycling processes allow for the production of physically identical or similar material inputs (Villalba et al., 2002, 2004), while bearing significant costs (Ayres, 1997). However, while substitution can be enhanced through technological change (Lafforgue and Rouge, 2019), studies show that materials like paper and cardboard do not show high substitutability between virgin and recycled resources, and can even be complementary inputs in production (Figge and Thorpe, 2019; Mansikkasalo et al., 2014). For these reasons, recycling should be included in the strong sustainability framework: first, to assess recycling's effect on sustainable consumption paths and determine under what conditions it can allow for a minimum level of consumption in the economy ; and, second, to

examine what strong sustainability implies when considering the substitution of virgin and recycled resources.

4.3 A maximin model of recycling

4.3.1 Setup of the model

We develop a resource and waste model based on the study of Pittel et al. (2010). This model has the advantage of being based on a standard Dasgupta-Heal-Solow (DHS) model, which is central in the study of maximin problems (Dasgupta and Heal, 1974, 1979; Solow, 1974; d'Autume and Schubert, 2008; Cairns and Martinet, 2021). The economy has a stock of capital K, that follows the investment rule below:

$$\dot{K} = F(K(t), z(t), r(t)) - c(t) \tag{4.1}$$

Production $F(.)$ is either invested ($\dot{K}$) or consumed (c).[1] It relies on capital K and two material resources: the extracted one z and the recycled one r. Production follows the standard neoclassical hypothesis:[2] F is continuous, non decreasing and strictly concave on $\mathbb{R}^3_+$ ($F_x \geq 0$ and $F_x < 0$, $x \in \{K, z, r\}$); marginal productivity is bounded ($F_x(0) < +\infty$, $x \in \{K, z, r\}$); and, inputs are essential to production ($F(0, z, r) = F(K, 0, r) = F(K, z, 0) = 0$). We suppose that virgin and recycled resources are essential to production, in line with the previously mentioned literature on the substitutability of virgin resources by extracted resources (Ayres, 2007; Mansikkasalo et al., 2014; Figge and Thorpe, 2019).

The flow of extracted input comes from a stock S of exhaustible resource with the

[1] In order to simplify notations, we will hide the time subscript whenever it is convenient and clear from the context.

[2] We denote by G_x the partial derivative of a function G according to variable x, $\frac{\partial G}{\partial x}$.

following dynamics:

$$\dot{S} = -z(t) \tag{4.2}$$

The flow of recycled input comes from a stock W of waste with the following dynamics:

$$\dot{W} = -r + \pi c - \rho W \tag{4.3}$$

Waste is accumulated through the share π of consumption, with π representing the physical share of consumption. We keep this share constant and exogenous ($\pi \in [0; 1]$). This waste stock is either recycled into an input r, or follows a natural resorption mechanism at a constant and exogenous rate ρ, $\rho \in [0; 1]$.

The economy is initially endowed with amounts K_0, S_0 and W_0 of stocks K, S and W ($\{K_0, S_0, W_0\} \in \mathbb{R}_+ \times \mathbb{R}_+^* \times \mathbb{R}_+$).

The economy includes identical consumers distributed along infinite generations, living instantaneously in continuous time. They derive utility $U(c, W)$ from consumption c and waste accumulation W. Utility $U(.)$ is twice differentiable, strictly increasing and strictly concave with respect to consumption ($U_c > 0$, $U_{cc} < 0$). It is twice differentiable and strictly decreasing with respect to waste accumulation ($U_W < 0$).[3]

The maximin value: definition – The maximin value m of a state (K, S, W) of the economy is defined as the highest level of utility U that can be sustained forever, starting from that state:

$$m(K, S, W) = \max_{u,c,z,r} u \tag{4.4}$$

$$\text{s.c. } (4.1), (4.2), (4.3) \text{ and } U(c, W) \geq u. \tag{4.5}$$

[3]This theoretical framework has been first studied for a maximin path by Stollery (1998), with the specific case of increasing temperature having an impact on production and utility. It brought further investigations on the topic to understand the role of negative externalities and find closed form solutions (d'Autume et al., 2010; Bazhanov, 2012).

In other words, the maximin criterion maximizes the utility of the poorest generation, with an equivalent mathematical expression being:

$$\max_{c,z,r} \left(\min_{t} U(c(t), W(t)) \right).$$

Optimal control problem – As demonstrated by Cairns and Long (2006) and Cairns et al. (2019), we can express a maximin problem through a standard optimal control problem. This approach consists in introducing the minimum level of utility u as an additional state variable $u(t)$, with $\dot{u} = 0$ and a free initial value $u(0)$. The problem is expressed as follows:

$$\begin{aligned}
&\max_{c(t),z(t),r(t)} u(0) \\
&\quad \text{s.c. } \dot{K} = F(K(t), z(t), r(t)) - c(t) \\
&\qquad \dot{S} = -z(t) \\
&\qquad \dot{W} = -r + \pi c - \rho W \\
&\qquad \dot{u} = 0 \\
&\qquad U(c(t), W(t)) - u(t) \geq 0 \\
&\qquad K(0) = K_0;\ S(0) = S_0;\ W(0) = W_0 \\
&\qquad \{c(t), z(t), r(t)\} \in \mathbb{R}^3_+ \\
&\qquad \{K(t), S(t), W(t), u(t)\} \in \mathbb{R}^4_+
\end{aligned} \tag{4.6}$$

From Proposition 2 in (Cairns et al., 2019), we know that an optimal solution complies with the following conditions.[4] There exist a number p_0, $p_0 \in \{0; 1\}$, costate variables $(\lambda(t), \mu(t), \eta(t), \gamma(t))$ associated respectively to state variables (K, S, W, u), and multiplier $\omega(t)$ associated to constraint (4.5) such that the following Hamiltonian and Lagrangian are

[4]For a detailed methodology and proof of the approach, see the work of Cairns et al. (2019). In particular, they use a the restrictive case of a upper-bounded production function that leads to a bounded maximin value. However, previous work on DHS models proves the existence for convex production functions (Mitra, 2013), given some specifications on the production function.

given by:

$$\mathcal{H} = \lambda(F(K, z, r) - c) + \mu(-z) + \eta(-r + \pi c - \rho W) + \gamma \times 0 \tag{4.7}$$

$$\mathcal{L} = \mathcal{H} + \omega(U(c, W) - u). \tag{4.8}$$

The optimal path follows the first order conditions:

$$\frac{\partial \mathcal{L}}{\partial c} = 0 \qquad \Leftrightarrow \lambda = \pi\eta + \omega U_c \tag{4.9}$$

$$\frac{\partial \mathcal{L}}{\partial z} = 0 \qquad \Leftrightarrow \lambda F_z = \mu \tag{4.10}$$

$$\frac{\partial \mathcal{L}}{\partial r} = 0 \qquad \Leftrightarrow \lambda F_r = \eta \tag{4.11}$$

$$\frac{\partial \mathcal{L}}{\partial K} = -\dot{\lambda} \qquad \Leftrightarrow \dot{\lambda} = -\lambda F_K \tag{4.12}$$

$$\frac{\partial \mathcal{L}}{\partial S} = -\dot{\mu} \qquad \Leftrightarrow \dot{\mu} = 0 \tag{4.13}$$

$$\frac{\partial \mathcal{L}}{\partial W} = -\dot{\eta} \qquad \Leftrightarrow \dot{\eta} = \rho\eta - \omega U_W \tag{4.14}$$

$$\frac{\partial \mathcal{L}}{\partial u} = -\dot{\gamma} \qquad \Leftrightarrow \dot{\gamma} = \omega \tag{4.15}$$

the slackness condition:

$$U(c, W) - u \geq 0 \; ; \; \omega \geq 0 \; ; \; \omega(U(c, W) - u) = 0 \tag{4.16}$$

transversality conditions:[5]

$$\lim_{\infty} \eta \geq 0 \text{ and } \lim_{\infty} \lambda = \lim_{\infty} \mu = \lim_{\infty} \eta W = \lim_{\infty} \gamma = 0 \tag{4.17}$$

[5]Unlike Cairns et al. (2019), we need a pure state constraint on the shadow values of W, as the stock could be exhausted by recycling and resorption while being refurnished by consumption. This leads to the possibility of a non zero shadow value at infinity, following Eq. (215b) Seierstad and Sydsaeter (1987), Theorem 16, p. 244-245.

and optimality condition on the free initial state $u(0)$:

$$\gamma(0) = -p_0 \tag{4.18}$$

As shown by Cairns and Long (2006) and Cairns et al. (2019), the Hamiltonian of the program is the net investment and is equal to zero along an optimal path. Furthermore, from Cairns and Martinet (2014), this net investment is the time derivative of the maximin value[6]

$$\mathcal{H}(K, S, W, u, c, z, r, \lambda, \mu, \eta, \gamma) = 0 \tag{4.19}$$

Regularity of the path – Let us examine the behavior of shadow value γ associated with the constraint of utility parameter u. By integrating equation (4.15) and using transversality conditions (4.17) and initial state condition (4.18), we get:

$$\int_0^\infty \omega(t)dt = p_0 \tag{4.20}$$

Variable ω relates to the constraint on a minimal level of utility to sustain, thus equal zero when the constraint holds. Equation (4.20) expresses wether equity holds during the program, *i.e.* wether utility varies from the constant path given by parameter u. Cairns and Long (2006) and Cairns et al. (2019) relate to this cost ω as the shadow value of equity, the opportunity cost maintaining a constant level of utility during the program. Constant utility paths are called regular paths and are conditioned by $\omega > 0$ thus $p_0 = 1$. Conditions for the emergence of non-regularity are given and interpreted by Cairns and Martinet (2014). Following their analysis, the path of our model is regular as utility is not locally bounded and the investment can be controled in the model (see equation (4.1)).

[6] Cairns and Long (2006) show the correspondance between costate variables and shadow values of the stocks: $\lambda = \frac{\partial m}{\partial K}$, $\mu = \frac{\partial m}{\partial S}$, $\eta = \frac{\partial m}{\partial W}$ and $\gamma = \frac{\partial m}{\partial u}$.

4.3.2 Characterization of the maximin path

We suppose that the set of maximin path is not empty.[7] From first order condition (4.9), the shadow value of equity is linked to shadow values of the capital and the waste stock. Indeed, the shadow value of the resource λ is equal to the the marginal utility of consumption weighed by shadow value of equity ωU_c, plus the shadow value of waste weighed by the material share of consumption $\pi\eta$. In other words, the value of the resource is the sum of a term relative to the opportunity cost of maintaining a minimum level of utility at a certain marginal utility of consumption, and a term relative to the value of waste at the level of the material share of production.

Proposition 13. *Shadow values of stocks have the same sign and a necessary and sufficient condition to be negative is that the marginal productivity of the recycled resource must be higher than the inverse of the material share in consumption:*

$$\frac{1}{\pi} < F_r \tag{4.21}$$

Proof. From (4.10) and (4.11), we know that shadow values λ, μ and η have the same sign. Rearranging (4.9) with (4.11) gives $\eta(\frac{1}{F_r} - \pi) = \omega U_c$. As $\omega > 0$ according to the slackness conditions and marginal utility of consumption is positive, it gives the necessary and sufficient condition $1 < \pi F_r$ for negative shadow values. □

If the shadow value of waste weighs negatively on the resource, with for instance an important disutility of waste, the resource and capital shadow values become negative. With a production function that does not rely a lot on the use of the recycled resource (meaning high marginal productivity due to concavity), or having a consumption that is highly material: the economy produces a lot of waste and that waste is not very useful for production. Waste weighs negatively on the economy through disutility. In this case, all stocks

[7] From Mitra (2013) and Cairns et al. (2019), we know that it is the case when waste has no effect on utility drawn from consumption.

contribute negatively to sustainability, as waste is too much of a burden in the economy: a marginal increase in capital or in the resource stock leads to a reduction of the maximin value as the subsequent production will lead to more harmful waste ($\frac{\partial m}{\partial X} > 0$, $X \in \{K, S\}$).

The combination of (4.10) and (4.11) give us the relative value of the resource and waste, equal to the marginal rate of substitution in production between waste and extracted resource:

$$\frac{\mu}{\eta} = \frac{F_z}{F_r} \tag{4.22}$$

According to equation (4.12), when positive the shadow value of capital decreases at the level of its productivity. The resource stock shadow value is constant according to equation (4.13). Rearranging (4.14) with first order conditions (4.9) and (4.11) give us the growth rate of the shadow value of waste:

$$\frac{\dot{\eta}}{\eta} = \rho - \left(\frac{1}{F_r} - \pi\right)\frac{U_W}{U_c} \tag{4.23}$$

Proposition 14. *There is an instant T_W such as $W(t) > 0$ while $t < T_W$ and $W(t) = 0$ when $t > T_W$ (with possibly $T_W = \infty$).*

- *When $t < T_W$, shadow values of stocks are always negative or always positive.*
- *When $t > T_W$, shadow values of stocks are positive, η increases and λ decreases.*

Proof. Transversality conditions on W give us that the waste stock is either zero at infinity, or its shadow value is zero. Suppose that the waste stock is not depleted. Cairns and Long (2006) precise that continuity of the shadow values is guaranteed only while the problem does not have pure state constraints that are reached (following the work of Seierstad and Sydsaeter (1987)). μ would then be constant and equal to zero (according to its transversality condition), thus all three shadow values of stocks are (according to equations (4.10) and (4.11)). This contradicts equation (4.9) and (4.20) as the path is regular ($\omega > 0$). By

contraposition, the waste stock must reach a phase of depletion. Transversality condition on η give us that shadow values are positive.

While $t < T_W$, equation (4.23) and first order conditions (4.10) and (4.11) give us the optimal behavior. At T_W, a discontinuity can occur (especially if shadow values are negative)(Cairns and Long, 2006). When $t > T_W$, as shadow values are positive, equations ruling their dynamics give us that η increases and λ decreases. □

Proposition 14 tells us that the previously described phase where shadow costs are negative can only happen during a transitory phase. From other first order conditions (4.12) and (4.13), we know that after T_W, the shadow value of the extracted resource is constant, and the shadow value of capital decreases at a rate equal to its marginal productivity.

Corollary 1. *After T_W, the marginal productivity of the recycled resource is bounded: $F_r < \frac{1}{\pi}$.*

The previous corollary involves that at given capital level and resource extraction, the use of the recycled resource has a lower bound.

By log-derivation of (4.10), we get that at any time on the optimal path, the marginal productivity of the resource increases at the level of the marginal productivity of capital:

$$F_K = \frac{\dot{F}_z}{F_z} \tag{4.24}$$

As the resource is depleted, meaning the use of extractive inputs in production must decrease, the marginal productivity grows due to concavity. As capital accumulates, its marginal productivity decreases. Equality (4.24) reflects the compensation by capital to sustain production. We can note that this relation does not depend on the the productivity of the recycled resource. Recycling has an effect hidden in the marginal productivity of the resource according to the following proposition.

Proposition 15. *Marginal productivities of material resources are related through the following equation:*

$$\frac{\dot{F}_r}{F_r} - \frac{\dot{F}_z}{F_z} = \rho + \left(\pi - \frac{1}{F_r}\right)\frac{U_W}{U_c} \tag{4.25}$$

Proof. The relation comes from the combination of equation (4.23) and the log-derivation of (4.11). □

After T_W, the right-hand side of equation (4.25) is positive: marginal productivity of the recycled resource grows quicker that the marginal productivity one of the virgin one. This difference, explained by the right hand side of the equation relies on the self resorption of waste ρ and a term relative to the reduction of the waste stock and its associated negative externality $\left(\pi - \frac{1}{F_r}\right)\frac{U_W}{U_c}$.

As explained by Cairns and Long (2006) and Cairns et al. (2019), the shadow value of equity ω can be interpreted as an endogenous discount factor, with the intertemporal maximization of $\omega(t)U(c(t), W(t))$ (Cairns and Long (2006), Proposition 3 p. 281). However ω is here the opportunity cost of maintaining the sustained level of utility, and not the preference between two time periods. Therefore we can isolate a discount rate that follows a pseudo Keynes-Ramsey rule by log-derivating equation (4.9) and replacing shadow values: [8]

$$-\frac{\dot{\omega}}{\omega} = F_K + \frac{\dot{U}_c}{U_c} + \frac{\pi\dot{F}_r}{1-\pi F_r} \tag{4.26}$$

Equation (4.26) shows that this virtual discount rate decreases at rate equal to two the sum of two standard terms already found by Cairns et al. (2019), as well as a third term relative to recycling. The first two terms are the marginal productivity of capital F_K and the rate of change in marginal utility in consumption $\dot{U}_c/U_c$. A third term $\pi\dot{F}_r/(1-\pi F_r)$

[8] Replacing η by (4.11) in (4.9) gives $\lambda(1-\pi F_r) = \omega U_c$. The we log-derivate the expression and we use (4.12) to replace λ.

relates to the rate of change in marginal productivity of the recycled resource weighed by the material part of consumption π. In a sense, one could consider that

4.3.3 Hartwick rule and sustainability

Hartwick rule

If an optimal sustainable path exists, it follows the Hartwick investment rule:

Proposition 16. *The optimal maximin path has an optimal Hartwick investment rule such as:*

$$\dot{K} = zF_z - \dot{W}F_r \tag{4.27}$$

Proof. We use the nil net investment condition (4.19) and combine it with first order conditions (4.10) and (4.11) to obtain equation (4.27). □

With the addition of the waste stock W, we can see that the Hartwick rule makes you invest more or less. The first term zF_z is common to standard models: we invest at the level of the productivity of the natural resource. However we must add here a second part, modulating the investment. The idea here is that recycling allows to invest less/more in order to sustain utility, at the level F_r, the marginal productivity of the recycled resource, times the apparent waste accumulation $\dot{W}$. When accumulated waste increases ($\dot{W} > 0$), more resource is available and we can invest less ceteris paribus. However, to this availability of the "waste resource" that allows less investment in capital must also be confronted the weigh of waste through disutility. This effect can be expected to intervene in the marginal productivity of recycling F_r, as shown in equation (4.25).

Starting from the Hartwick rule, we now want to assess the effect of recycling on strong sustainability criteria. Is a maximin path exists, it follows the Hartwick rule. However in the DHS model, this rule imposes an upper bound on the share of production to invest. If

this share given by the Hartwick rule is too high, a minimum level of consumption cannot be sustained. By giving Cobb Douglas forms to production functions ($F(K,z) = K^{\alpha}z^{\beta}$), work from Dasgupta and Heal (1979) give us that a maximin solution is attainable while $\alpha > \beta$.[9] It means that production should rely more on capital than resource for a sustainable maximin path to exist. Such a production function (where $\alpha < \beta$) without a maximin solution is one way of expressing that resource is essential to production, thus being in line with the strong sustainability paradigm (Neumayer, 2013).[10] With this kind of hypothesis in our model, we do not know if a maximin path exists, as it does not in the case without recycling.

We introduce a Cobb-Douglas function for production:

$$F(K,z,r) = K^{\alpha}z^{\beta}r^{\delta} \tag{4.28}$$

with $\alpha, \beta, \delta > 0$ and $\alpha + \beta + \delta < 1$.

We use the following utility function:

$$U(c(t), W(t)) = \frac{v(t)^{1-1/\sigma}}{1-1/\sigma} \text{ and } v(t) = c(t)W(t)^{-\epsilon} \tag{4.29}$$

with $\sigma > 0$, $\sigma \neq 1$ and $\epsilon \geq 0$.

The investment rule is now:

$$\dot{K} = \left(\beta + \frac{-\delta}{r/\dot{W}}\right) F \equiv \theta F \tag{4.30}$$

Similarly to the standard case, equation (4.30) shows that a share θ of consumption should be invested in capital. In the recycling case, this share is not constant and an additional term $\frac{-\delta}{r/\dot{W}}$ is added to the elasticity β of production with respect to the resource.

[9] In DHS models, the Hartwick rule can be reduced to a fixed share of production, equal to the elasticity of production with respect to the resource: $\dot{K} = \beta F$. It follows that for the maximin to exist, we shall have to invest too much.

[10] As highlighted by Neumayer (2013), other possibilities would be to allow for time dependent production elasticities or using a constant elasticity of substitution (CES) function with an elasticity of substitution below one, meaning that there is complementarity between the resource and capital.

	$\dot{W} > 0$	$\dot{W} < 0$	$\dot{W} = 0$
$r > \lvert\dot{W}\rvert$	$\beta - \delta < \theta < \beta$ Not optimal	$\beta < \theta < \beta + \delta$	$\theta = \beta$
$r < \lvert\dot{W}\rvert$	$\theta < \beta - \delta$ Not optimal	$\beta + \delta < \theta$	$\theta = \beta$ Impossible
$r = \lvert\dot{W}\rvert$	$\beta - \delta$ Not optimal	$\theta = \beta + \delta$ Standard DHS model with two resources	$\theta = \beta$ Standard DHS model with one resource (not optimal)

Table 4.1: Share of reinvested production $\theta(t)$

$r/\dot{W}$ is the share of recycled input that contributes to the net variation of the waste stock. When it is equal to -1, $\pi c = \rho W$: self resorption exactly compensates the flow of waste and production can only rely on the existing stock. The waste stock behaves like a standard resource stock and we find the usual investment rule with two natural stocks: $\theta = \beta + \delta$ (see Table 4.1).

As previously mentioned, in the standard maximin problem for DHS models with a Cobb Douglas production function, the share of production that is reallocated to the capital is the elasticity of production with respect to the resource. For a sustainable path to exist, it must be lower than the elasticity of capital in production. This gives us insights to discuss sustainability of a maximin path with recycling: does the Hartwick rule impose a lower share of production to have a sustainable trajectory, in which case making the problem more likely to have a solution. Table 4.1 gathers the possible values this share θ can take during an sustainable path. θ is not fixed and can vary through time. However as shown previously, the optimal path leads to an exhaustion of the waste stock, in finite or infinite time. For this reason, first column where the waste stock increases cannot be optimal in the long run, while the underlying consumption path is trivially sustainable. If the stock decreases, the question of the feasibility is raised. When the rate is constant and matches a standard DHS model with two resources, the standard condition $\alpha > \theta$ applies in the long run for a sustainable path to exist. In the two other cases, sustainability is not trivial.

In the case of the waste stock reaching a steady state ($\dot{W} = 0$), θ is always equal to β. In this phase, the use of recycled material cannot be lower that the variation of waste, for obvious positivity reasons. When both are equal ($r = \dot{W} = 0$), self-resorption exactly compensates the waste flow πc. The recycled resource is not used and the model becomes a standard DHS one with a single depletable resource. However in the long run, it is not optimal as the transversality condition involves that W reaches zero, and consumption with it. Finally, the case where recycling is used in production involves that, in the long run with a depleted waste stock, we instantaneously use in production what comes from production ($r = \pi c$).

Closed form solution without externalities

In order to expand this analysis, we know suppose that time T_W where the waste stock is depleted is reached in finite time ($T_W < \infty$). It follows that during this phase, the utility function does not depend on waste: $U(c)$. The dynamics of the system are given by:

$$\dot{K} = \beta F \tag{4.31}$$

$$\dot{S} = -F^{1/\beta} K^{-\alpha/\beta} r^{-\delta/\beta} \tag{4.32}$$

Proposition 17. *When the waste stock is depleted in a finite time T_W, the optimal level of consumption is given by:*

$$c^* = \left(\pi^{\delta}(1-\beta)^{1-\beta}\right)^{1/(1-(\beta+\delta))} \left((\alpha-\beta) S_{T_W} K_{T_W}^{\frac{\alpha-\beta}{\beta}}\right)^{\beta/(1-(\beta+\delta))} \tag{4.33}$$

where S_{T_W} and K_{T_W} are the levels of stocks at T_W. In this case, a necessary condition for the production function is $\alpha > \beta$.

Proof. In order to find a closed-form solution, we follow the methodology of d'Autume and Schubert (2008). We introduce $c(U)$ consumption as a function of the utility U only (as $W = 0$). For a regular maximin path, it is constant. The investment rule and the dynamics

of capital give us that $F = c(U)/(1-\beta)$. As the waste stock is depleted, we have $r = \pi c(U)$. Replacing in (4.31) and (4.32), and combining both it gives us:

$$-\pi^{\delta/\beta}(1-\beta)^{\frac{1-\beta}{\beta}}c(U)^{-\frac{1-(\beta+\delta)}{\beta}}dS = \frac{1}{\beta}K^{-\alpha/\beta}dK$$

Integration between S_{T_W} and S on the left, and K_{T_W} and K on the right gives:

$$\pi^{\delta/\beta}(1-\beta)^{\frac{1-\beta}{\beta}}c(U)^{-\frac{1-(\beta+\delta)}{\beta}}(S_{T_W}-S) = \frac{1}{\alpha-\beta}\left(K_{T_W}^{1-\alpha/\beta} - K^{1-\alpha/\beta}\right)$$

At this stage, for a solution to exist, there must be $\alpha > \beta$, otherwise the resource difference diverges towards infinity. Isolating $c(U)$ we get the optimal level c^* given by (4.33). □

An important take of proposition 17 is that despite the existence of recycling, the weak sustainability constraint cannot be relaxed: we must have $\alpha > \beta$. However, it changes the sustainable level of consumption given by (4.33). When $\delta = 0$, we find the known value of sustainable consumption from Dasgupta and Heal (1979). Two effects of recycling are in play here.[11] First, consumption itself allows a higher level of utility as it can be reintegrated in production through recycling. Mathematically, it appears with exposants $\beta/(1-(\beta+\delta)) > \beta/(1-\beta)$. However, as consumption is partially material, the material share π also intervenes through the factor $\pi^{\delta/(1-(\beta+\delta))} < 1$. It attenuates the increase in sustainable consumption (compared with the level in the standard case without recycling). It should also be kept in mind that this level of consumption intervenes after a transitory phase with the depletion of the waste stock. However, one could assume that the optimality problem could start with an already depleted waste stock. In that case, equation (4.33) only includes exogenous parameters as stock levels are initial ones S_0 and K_0. Note that the effect of δ is ambiguous: one can assume that it alleviates part of the scarcity as recycling can be considered as a partially renewable resource. However it also leads to a dependency on a

[11] Note that the addition of recycling can be compared with elasticities α and β remaining the same. A proper comparison could also consider that initial elasticity $\tilde{\beta}$ in a model without recycling is distributed between elasticities of recycling and virgin resource: $\tilde{\beta} = \beta + \delta$.

resource depending on the level of consumption and its material share. Taking for instance a highly dematerialized consumption (π is low), the sustained level of consumption could be lower with recycling, as the economy relies on a poorly available resource. Note that as the previous reasoning is done when it has become optimal to keep waste at zero, so the rate of self resorption does not intervene. Analyzing the effect of recycling for an optimal path with a non zero waste stock involves less tractable equations as seen in equation (4.30).

Substitutable material resources

Another way of modeling resource input is to consider waste and virgin resource as complementary inputs (Di Vita, 2001, 2007). β is now the elasticity of all material inputs in production, and we introduce ζ ($0 < \zeta \leq 1$), the technological substitutability between both material inputs. Perfect substitutability occurs when $\zeta = 1$. Note that now, the strict combination of both material inputs are not essential in production. The production function becomes:

$$F(K, z, r) = K^{\alpha} (z + \zeta r)^{\beta} \tag{4.34}$$

In this case, the Hartwick rule is now the following:

$$\dot{K} = \left(1 - \frac{\zeta(\pi c - \rho W)}{z + \zeta r}\right) \beta F \tag{4.35}$$

The share $\theta(t)$ of consumption invested in capital relies on the ratio of input that comes from net production at the level of its technological efficiency $\zeta(\pi c - \rho W)$ in the global use of material input $z + \zeta r$. This ratio is positive, as we have seen that it is not optimal to increase the waste stock. Comparing with the standard Hartwick rule, it is lowered as net residuals of consumption $\zeta(\pi c - \rho W)$ are included in production. We find a similar result as before: recycled input allows for less investment to maintain a sustainable level of utility while the waste stock is not depleted.

4.4 Conclusion

While recycling and sustainability economics seem to be topics with strong connections, a framework of analysis that includes both is slow to emerge. In particular, despite many economic models representing recycling as a way to ensure optimal consumption paths (mainly through waste reduction and a less stringent resource scarcity), modeling, in its vast majority, relies on intertemporal criteria that do not favour equity and are thus unsustainable. This goes indeed in contradiction with the idea of sustainability as defined in diverse reports and in the economic literature (WCED, 1987; Baumgärtner and Quaas, 2010). These observations advocate for a more thorough study of recycling with the use of the concepts and mathematical tools of sustainability economics: this goes from a diversity of sustainability criteria to weak and strong sustainability frameworks (Martinet, 2012; Neumayer, 2013).

Studying a maximin problem with a growth model that includes waste and recycling is one first step in this direction. This model, while being simple, adds complexity to the analysis of a minimum sustainable consumption level. We are able to characterize an optimal path and its shadow values, and discuss some long-run analysis. More complex resource management rules are reflected by a new Hartwick investment rule. The rule now depends on waste accumulation and the use of recycling in production. In particular, the share of production that has to be dedicated to capital accumulation can be studied in light of the already existing results for models with maximin objectives (Dasgupta and Heal, 1979). In the specific case of a waste stock emptied in finite time, it is possible to discuss explicit consumption paths when specifying production functions. In the case of a Cobb-Douglas production function, we show that the assumption of having an elasticity of capital higher than the elasticity of resource in production is crucial for a sustainable path to exist. This weakens the idea that recycling could be, in this theoretical framework, a way to get a sustainable path when assuming the strong sustainability hypothesis.

Further research could go in two different directions. First, the study of transitory dynamics should be added, especially when it is optimal to deplete the waste stock asymptoti-

cally. It seems mathematically challenging to get explicit solutions, however it would greatly improve the comparative analysis between maximin with recycling and maximin without recycling. Second, the topic of recycling and strong sustainability is also central. Results here are not very optimistic regarding the impact of recycling on the long run. However, it could be expanded to other functional forms such as CES functions with elasticities lower than one, meaning that inputs are complements. This hypothesis is central for the resource/capital combination in production and allows to determine the sustainability framework considered in the model. Moreover, it is also the case for the extraction/recycling combination in production, as for many materials, empirical studies do not find perfect substitution between both inputs. This raises the question wether substitution between recycled and extracted material should also be a central hypothesis of the strong sustainability paradigm.

GENERAL CONCLUSION

This dissertation explores the economics of recycling when taking into account the different externalities linked to resource extraction, transformation and consumption. It includes the depletion of natural stocks, the environmental impacts of waste accumulation and GHG emissions. We examine the trade-offs between recycling and extraction in different cases, with different methodologies. For this purpose, we combine theoretical approaches of resource economics as well as simulation and prospective approaches using calibrated data.

In the first chapter, we consider a model in resource economics allowing us to specify under which conditions it is optimal to recycle and use a recycled resource in production. This analysis takes into account the social costs of extraction, consumption and recycling. In order to set optimal consumption paths, private costs and GHG emissions from recycling are also taken into account.

In the second chapter, this take on recycling is enhanced using an integrated bio-economic assessment model showing how increasing recycling for the paper and cardboard industry can lead to negative climate impacts. This counterintuitive result prevails when both resources are complementary for paper and cardboard production.

Focusing on the specific sector of Li-ion batteries, chapter three examines the feasibility of a policy imposing recycled contents thresholds with a resource flow model. Once again, we show that the optimal level of recycled material used in production is the function of

a trade-off between several parameters across time. In particular, another sustainability parameter - the lifetime of batteries - can hinder the objective of a high recycled content.

As recycling is strongly linked to sustainability objectives, chapter four assesses the lack of sustainability considerations in the macroeconomic literature of recycling. We introduce the first model that examines recycling through the use of the maximin criterion.

The rest of this conclusive section is divided in two parts. The first part examines policy implications of the different chapters and tries to discuss them in a transversal analysis. The second part proposes leads for future research.

Transversal and policy considerations

An holistic approach of recycling

The introduction to this dissertation describes how material use in the economy is informed by many different parameters related to resource stocks, the environment, climate change and social considerations. For this reason, a thorough study of recycling cannot be done without a holistic approach. This is especially true when addressing the climate impact of recycling. As shown in chapter one, trade-offs between inputs in production are influenced by the externalities we take into account. The use of recycled material can be encouraged in order to reduce GHG emissions (chapter one and three), but can also be at the origin of negative impacts on climate, given the specificities of some production sectors (chapter two). By considering recycling as a means for achieving sustainability, policies aiming at encouraging recycling have to take into consideration the multidimensional aspects behind it. This means accounting for trade-offs between different private and social costs, defining sectorial and spatial boundaries, and including the temporal dimension of the system (chapter four).

Can recycling be bad?

Some results from this dissertation can lead the reader to consider that recycling is not such a good thing, or can even be a bad thing in terms of climate change or other sustainability objectives (chapter two and three). In fact, as with any economic decision, the net social outcome of choosing between recycling or not recycling depends on the assumptions made. In the case of perfect substitution, higher GHG emissions, and not so high private costs, recycling is the favored option (chapter one). However, when recycling can in many cases be the socially optimal choice, properly defining spatial and temporal boundaries of the system, industries and markets can lead to negative impacts (chapter two and three). In the specific case of paper and cardboard recycling with a complementarity between recycled paper pulp and wood pulp, the marginal effect of an increase in recycling that has a negative climate impact. One cannot conclude that all paper recycling is bad, but given the evolution of the industry through the years, we are in a situation where more recycling can be harmful. These results advocate for more broadly defined sustainability objectives to be used, thus not only recycling objectives, when putting in place new policies.

Substitution and complementarity

From this dissertation, a central parameter when evaluating the impact of recycling and whether or not it should be promoted appears to be the substitutability between recycled and extracted materials. Perfect substitution, a feasible assumption for metals, allows for the switch from one input to the other without productivity considerations (chapter one and three). However, complementarity forces the industrial apparatus to produce more output when recycling is encouraged (chapter two), a result some authors called the circular economy rebound. In a way, the substitutability issue here is comparable to the debate addressed by sustainability economics: can accumulated capital be a substitute for natural capital. The economic literature shows that this question is crucial for the feasibility of economic paths that preserve natural resources. In the case of complementarity, part of what

is called the strong sustainability paradigm, long term consumption must be guaranteed by renewable resources. For this reason, policies regarding recycling should address the different sustainability paradigms (chapter four) and thus take into account, for each material considered, how it can be substituted.

Recycling and circular economy as a goal?

In parts of this dissertation, recycling is not taken as a goal, but as a production choice that comes from the trade-off between different private costs and externalities (chapter one and four). When recycling is a goal in itself (chapter three) or a consequence of a market equilibrium (chapter two), we show that it can lead to adverse results. Thus, we advocate for policy makers not to consider recycling as an objective per se, but one option, admittedly often favorable, for industries. Indeed, when considering the concept of a circular economy, recycling only appears as one of its principles (although a central one as it allows to close the material loop in the economy). It can be completed by other practices that are part of the circular economy such as reuse, eco-design or product-service systems. Recycling, and the circular economy on a larger scale, should be studied as means for a pathways towards sustainability. When not considering recycling as and end in itself, it involves reflecting on its meaning, relevance and impact. This also means coming back to the definition of sustainability as given by the Brundtland report, "Our Common Future" (WCED 1987): ensuring that humanity "meets the needs of the present without compromising the ability of future generations to meet their own needs", and designing policies that support this goal.

Future work

Recycling and sustainability accounting

Chapter three is a first attempt at reconciling the macroeconomics of recycling and sustainability accounting. The costs and benefits of recycling are indeed distributed across time.

An optimal recycling rate should take into account inequities across generations when allowing for more available resources as well as a reduction in waste accumulation. Furthermore, sustainability concerns cannot be separated from the topic of climate change: future generations can have welfare gains when current ones use less carbon intensive resources like, in most cases, recycled ones.

Sustainable accounting for an optimal depletion of resources is also a crucial topic for critical raw materials in the energy transition. It is necessary to assess the impact of recycling when those minerals will be crucial to develop low carbon infrastructures in the future decades.

In this dissertation we limit ourselves to the maximin criterion. Other frameworks of analysis could be used to study optimal recycling programs, such as a golden rule (maximizing long term utility) or the Chichilnisky criterion (a trade off between present and future generations).

Oligopolistic market structures for critical raw materials

This dissertation deals with several aspects of recycling with the scope of an arbitration between costs, resource availability and externalities. However it does not examine the market structure of recycling and mining industries. In particular, critical raw materials for the energy transition are managed by oligopolistic markets that operate on a worldwide scale. The emergence of recycling was an important topic of research in the second half of the 20^{th} century, with the example of the Alcoa antitrust case for aluminum. This branch of research is now subject to renewed interest as recycling industries are expected to gain in size. In particular, this poses interesting questions in the context of the increasing environmental and social pressures on these industries.

Substituting virgin and recycled resources

Being able to substitute virgin resources with recycled ones appears to be a central question in assessing the environmental and economical impact of recycling. Most of the theoretical work relies on arbitrary hypothesis such as perfect substitution, which is really only valid for non ferrous metals. In reality, combining both resources in industrial processes relies on complex physical and economical constraints. Having complementarity between inputs could be dramatic for the environmental impact of recycling as it could lead to a rebound effect of material use. Moreover, substitutability is interesting for policy makers as it can have an impact on policy instruments. It questions practices of industrials and waste managers through the choice of waste treatments and the quality of the recycled output.

Econometric work already exists for the paper and cardboard industry, as the sector is already mature in developed countries (in France, 66% of the production comes from recycled waste). However, the substitutability of recycled plastic resins is under-researched, while plastic recycling remains to be a critical challenge. In both economic and technical literature, it appears that categories of plastics encompass a wide range of resins with different properties, uses and the degree of recyclability. The prices of resins follow distinct trajectories as they are not linked to the same industrial sectors. As such, resins must be approached separately as heterogeneous products, since they generally are not interchangeable. Thus, I propose econometric research on plastic products as the next step towards understanding the potential of recycling for addressing sustainability in the plastics industry.

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* * *

Si notre or est fumier, en revanche, notre fumier est or.

Que fait-on de cet or fumier? On le balaye à l'abîme.

[...]

Ces tas d'ordures du coin des bornes, ces tombereaux de boue cahotés la nuit dans les rues, ces affreux tonneaux de la voirie, ces fétides écoulements de fange souterraine que le pavé vous cache, savez-vous ce que c'est?

C'est de la prairie en fleur, c'est de l'herbe verte, c'est du serpolet et du thym et de la sauge, c'est du gibier, c'est du bétail, c'est le mugissement satisfait des grands bœufs le soir, c'est du foin parfumé, c'est du blé doré, c'est du pain sur votre table, c'est du sang chaud dans vos veines, c'est de la santé, c'est de la joie, c'est de la vie.

Victor Hugo - *Les Misérables* - Livre V

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