Funding Conservation

Multilevel Governance and the LIFE Nature Programme

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Abstract

The twin crises of Climate Change and Mass Extinction are the defining geopolitical mega-trends of this century. There is an embedded multilevel tension in these crises, that they cannot be affected at the local level, but that it is the local, physical level which affects them, suggesting a multilevel governance framework is most appropriate for analyses of these crises. There is also high politicization around attempts to respond to these crises in meaningful ways. Together, this suggests an avenue for analyzing trends in crises related governance through multilevel governance and post-functionalism.

This paper sets out first to provide a rough background on the development of the biodiversity crisis, with particular attention to its social-historical connection to Europe. Natura 2000 and the LIFE Programme are then introduced as the chief EU level instruments in responding to this crisis, and as prime candidates for analysis through a post-functionalist, multilevel governance framework.

To add an additional level of predictability in the relationship between multilevel governance and post-functionalism, rational choice is discussed, and a sketch of a framework developed to incorporate it in the two. This is then applied to the somewhat general question of ‘How does the multilevel governance of the LIFE Programme affect conservation trends’ to generate a series of quantitatively testable hypotheses.

These hypotheses are then tested through a series of generalized linear models. The findings from these tests generally conform to the expectations of the hypotheses. They find that, in general, the LIFE programme does improve conservation trends and that the LIFE programme is responsive to changes in conservation trends, but that this is heavily dependent on the Member States and should not be seen as a function of the Commission. Furthermore, these tests suggest that public authorities tend to engage LIFE Programme project funding in locations which maximize project funds relative to multiple conservation interests, though this is not the case for regional authorities. Contrary to expectations, local authorities are seen to have no preference by the Commission in leading projects and are seen to have the least significant impact on conservation trends compared to other authority types. Lastly, the Commission was suggested to have little interest in the location of projects relative to their integration into other non-Natura 2000 protection regimes.

In all, these tests are limited by a combination of data and scope. The framework introduced, however, does seem to provide some predictability, and is potentially a good candidate for analysis beyond policy effectiveness, and into trends of EU integration.

Introduction

At no point in the history of life on earth, save for the asteroid impact marking the end of the Mesozoic Era, has life itself so suddenly and dramatically been confronted with existential challenges like those unleashed only moments ago in the Industrial Revolution (Keller et al., 2018). Confronting these challenges, and the systems which produce them, is increasingly recognized as a historical imperative (Keller et al., 2018; Victor, 2012; IPCC, 2023) and beginning to attract political action reflecting the work which must be done to do so (Gupta, 2010). However, these political actions, necessarily international and systemic in character (Gupta, 2010), remain insufficient in maintaining the stability of the Holocene epoch, from which we are within an ever-shrinking hair's breadth of departing (Waters et al, 2016; Ditlevensen and Johnsen, 2010).

These challenges can be broadly distilled into the Climate Crisis, and the Biodiversity Crisis (Pörtner et al., 2023), with the Climate Crisis reflecting the destabilization of earth's climate through (primarily) the dramatic increase in atmospheric CO2, and the Biodiversity Crisis reflecting the destabilization of global ecosystems. These crises are largely interlinked and feed into one another (Pörtner et al., 2023), and are yet fundamentally different. Echoed in this difference is the degree of progress made in overcoming these crises, wherein progress has more so been made in favor of addressing the Climate Crisis than the Biodiversity Crisis (Directorate-General for Climate Action, 2023; Dhakal et al., 2022).

While there are many reasons for this difference, including tradeoffs in how addressing one may come at the expense of the other (Pörtner et al., 2023), they fundamentally reflect the fact that a resolution to the Climate Crisis is easier to achieve than a resolution to the Biodiversity Crisis. In contrast to the Climate Crisis, where the resolution is relatively simple (stop emitting CO2), the five main drivers of the Biodiversity Crisis in need of addressing; Land/Sea Use, Natural Resource Exploitation, Climate Change, Invasive Species, and Pollution (UNEP, 2023; Otero et al., 2020) suggest not only the need for a substantial re-evaluation of the systemic relationship between human activity and the natural world on a global scale, but also meaningful progress towards resolving the Climate Crisis. In this way, meaningfully addressing the Climate Crisis (with all its struggles) is *necessary* for addressing the Biodiversity Crisis, but alone is not *sufficient.* Therefore, such a transition to resolve the Biodiversity Crisis will take at least as long as the resolution to the Climate Crisis and will be at least as challenging, with an additional series of challenges to every constellation of social power based in both their specific and interlinked relations to the natural world.

The complex anatomy of the Biodiversity Crisis does not necessarily translate to a complexity in the observed outcomes of biodiversity's status today. The relationships between species form the dynamic underlying basis of ecosystems (Ives and Carpenter, 2007). As species come and go, these ecosystems change, and in turn induce further change, albeit very slowly. In this way, extinction, or the disappearance of species, is a normal phenomenon endogenous to these ecosystems, with the generally agreed upon 'natural' rate of extinction between .1 and 1 species per million per year (Smithsonian, n.d.; De Vos et al., 2014). Despite this, the current planetary rate of extinction is estimated at anywhere between hundreds and thousands of times greater than the 'natural' rate (De Vos et al., 2014; Singh, 2002). Difficulties in pinning down a more exact extinction rate stem from difficulties in data collection (Daigle and Janicki, 2022) and an incomplete knowledge of all species on Earth (Mora et al, 2011), yet the trend is conclusive: species are disappearing at a catastrophically alarming rate everywhere.

The global and systemic character of the Biodiversity Crisis suggests that it cannot and should not be considered separately from the global power systems which have developed alongside the crisis' birth and acceleration. While the benchmark for the crises' acceleration seems to be accepted as taking place in the 1970s (WWF, 2021), the systems driving global ecosystem destabilization long predate this inflection point (Waters et al, 2016, Millhauser and Earle, 2022). The systems in question are those historically indivisible from the process of industrialization and are intimately bound up in the Central State, namely those systems of investment and ownership, and those which connect them globally (Otero et al., 2020). In this way, the Biodiversity Crisis concerns a political history which extends through the past several hundred years, which necessarily involves the fact of colonialism, and is deeply European in character.

While it is undeniable that the birth and acceleration of the Biodiversity Crisis has coincided with the twin mega-trends of colonialism and industrialization (Waters et al., 2016; Raja, 2022; Dorninger et al., 2021), thus making its historical fact inseparable from the two, it would be inappropriate to say that the relationship between people and the natural world prior to the emergence of these trends did not also destabilize this 'natural' rate of extinction. The paleontological and archeological record suggests that, over the past several hundred thousand years, the first arrival of humans to new areas of the world similarly coincided with a localized extinction of many of those areas' megafauna (Barnosky and Lindsey, 2010; Miller et al., 2005). Of course, the Biodiversity Crisis differs from these localized extinctions for reasons of scale and degree - whereas the radiating effects of megafauna extinction on local ecosystems cannot be denied, the Biodiversity Crisis threatens *all* species, thus generating an ever-increasing intensity of polycentric shockwaves in *all* global ecosystems. However, this record raises some nuance to the origins of the Biodiversity Crisis, in particular questions about blame: whether the European character of the Biodiversity Crisis was necessary for its ignition, or simply sufficient. Such discussion however, lies beyond the scope of this paper.

In approaching the political solutions to this crisis, it is necessary to address the fundamental paradigm associating these systems to the living world, which have thus far at least accepted the crisis' birth and acceleration. At the core of this paradigm is the assumption of a separation between humans and nature, to which biodiversity is synonymous (Foster and Clark, 2020; Paterson, 2006; Caillon et al., 2017). Herein, nature is viewed as an object to be mastered for human safety and prosperity, reflecting the Christian belief in Nature as a *gift* from God to Humanity (Lea, 1994). Nature is in this way without value, but rather valuable for what can be made of it and what it can do for people. This anthropocentric valuation of nature is unavoidably linked to the Biodiversity Crisis as it has happened, yet this valuation has also produced many of the methods from which solutions to the crisis are being approached.

One such approach to valuing nature is through 'Ecosystem Services', which attempt to put in monetary terms an estimated valuation reflecting the hitherto unpaid contribution of nature to people (Wallace, 2007). These Ecosystem Services encompass a wide range of uncompensated relationships from which the global economy is dependent, including, but not limited to pollination, medicines, water purification, food and wildfire protection, carbon sequestration, and more (OECD, 2019). In total, the global ecosystem services to humanity are, as of 2019, estimated between USD 125 and 140 trillion per year. Despite this, it is estimated land cover change and degradation led to a combined loss of between USD 10 and 31 trillion per year between 1997 and 2011 (OECD, 2019). This monetary evaluation is often also broken down to reflect the contributions to specific industries and areas of the world at varying scales (Bommarco et al., 2018; Wolff et al, 2017). Unlike Carbon Trading or Carbon Pricing schemes which are market interventions that try to incorporate the previously unpaid price of CO2 emissions in production (Klenert et al., 2018), Ecosystem Service accounting is at this stage primarily a referential tool for policymakers to justify the act of protecting nature (OECD, 2019; Directorate General of the Environment, 2021). This Ecosystem Service accounting provides an economically marginalist tool for mediating compromise between the parties who find at least some anthropocentric value in nature.

Unsurprisingly, given the European character of this paradigm and crisis, Ecosystem Service accounting features predominantly in the rationale to the von der Leyen Commission's 2030 Biodiversity Strategy, the central document outlining the European Union's (EU) plan for addressing the Biodiversity Crisis (COM/2020/380). This document also reflects the international political character of the crisis and the collective responses required to meet it. The 2030 Biodiversity Strategy aligns itself to the objectives of UN 2030 Agenda for Sustainable Development and, reflecting the interrelation to the Climate Crisis, the Paris Climate Accords. Furthermore, the Biodiversity Strategy preemptively aligns the EU strategy to the (as of then unsigned) United Nations' 2022 Kunming-Montreal Global Biodiversity Framework (GBF) which, most notably, commits all signatories to protect 30% of the world's lands and oceans for nature by 2030 (CBD/COP/DEC/15/4).

At the heart of the EUs strategy is the Natura 2000 Network (N2K). N2K is the EU's flagship nature conservation program. Born out of the Birds and Habitats, or Nature Directives (Directorate-General of the Environment, n.d.), and covering 18% of the EU landmass and an additional 10% of EU waters (European Environmental Agency, 2018), N2K is the world's largest international network of its kind, and well positioned as the EU tool to meet the GBF target. N2K is generally agreed upon as a moderately successful program in terms of Conservation Outcomes, with the stability of species populations and habitats within designated sites faring better than those outside (Sundseth, 2021), and the protected areas in turn providing between EUR 200 and 300 billion per year in Ecosystem Services (European Commission, 2017).

The more than 27,000 sites making up this network vary considerably. While the Nature Directives guarantee a minimum level of protection for all N2K sites, often correlating to Category IV - Habitat/Species Management Area or Category VI - Managed Resource Protected Area in the IUCN Protected Areas Category System (Dudley, 2008), a site may have more strict protections from, for example, a Member State's site integration to or establishment of a national park in the N2K system (Romão et al., 2012). Reflecting this degree of difference, only .6% of N2K sites in Belgium fall under the strictest protection categories while a full 72.7% of those in Finland are categorized under the strictest protections (Gatti et al., 2023). Furthermore, given the scale of sites and their proximity to one another, multiple categories of protection may apply to a single site (Gatti et al., 2023).

The selection of these sites is done by the Member States (Evans, 2012), each with a different approach. Whereas some Member States, such as France, have been characterized as following a top-down, expert driven approach (McCauley, 2008), others, like Germany, the process has largely been led by the regions (Eben, 2006). In the case of France, this top-down approach led to delays in the implementation of the Habitats Directive following local resistance to the French designation process (Alphandéry and Fortier, 2001). This selection process is very important, as it is suggested to influence conservation outcomes by building local support (or generating resistance) among the people who are ultimately involved in the daily interactions with a site (Young et al., 2013), and is considered one of the more important determinants of conservation outcomes in practice (Kruk et al., 2010) Furthermore, Member States are responsible for the development of Management Plans for sites in relation to the target species or habitats for which a site is designated (Sundseth, 2021). However, these management plans are still incomplete, reflecting the difficulties in developing a unified conservation objective in the context of multilevel and multidirectional interests (Sundseth, 2021; Louette et al., 2011).

The Commission's role in the governance of N2K sites is mainly through overseeing the EU Directives which oblige Member States to include areas within the N2K network (Evans, 2012). In this context, the Commission has taken Member States to the European Court of Justice for failure to meet designation obligations (Paavola, 2004). Additionally, the sites which are selected by the Member States go through a process of evaluation, wherein the Commission has the ultimate role of site approval (Council Directive 92/43/EEC). This approval is largely nominal and is meant to prevent inappropriate site designation where target species or habitats are not listed, and there is no credible link to the site's designation and their protection (Council Directive 92/43/EEC). This process ensures a degree of compliance with the Nature Directives which is not merely performative.

Beyond the Directives, the Commission operates the LIFE Nature funding programme, which seeks to improve the status of nature in N2K sites to meet the Article 2 objectives of the

Habitats Directive (Council Directive 92/43/EEC, European Environmental Agency, n.d.a). This programme is the primary tool for the EU to interact with Natura 2000 sites after the designation process. As the LIFE Nature programme co-funds projects, the remaining funds are made up by either the beneficiary, or by some third party (European Commission, 2019). The projects are written and operated by a variety of actors ranging from varying authorities, local, regional, and national, to NGOs and other private or semi-private actors (European Climate, Infrastructure, and Environment Executive Agency, 2024a). While the Commission exercises the role as the ultimate selector of these projects for funding, they are afterwards relatively autonomous to the Commission. The LIFE Nature programme reflects the multilevel governance of the N2K programme, and therefore, the study of the governance of the LIFE Nature programme can communicate the ways in which different actors navigate N2K, and how that navigation influences conservation outcomes.

In this way, N2K (and by extension LIFE) represents a European navigation of the tensions embedded at the heart of the biodiversity crisis. Namely, that the crises cannot be affected at the local level, but that it is the local, physical level which affects it. Herein, Natura 2000 and the LIFE programme seek to overcome a massive coordination problem by balancing multiple necessary and interrelated scales in the governance of European nature.

However, while N2K remains the EU's premier nature conservation policy, the 2030 Biodiversity Strategy explores future options for the network in the face of the accelerating biodiversity crisis (COM/2020/380). Herein, the Commission speaks of 'completing' N2K, giving it more support, and expanding the range of protected areas to align with the Kunming-Montreal Biodiversity Framework objectives. The Commission envisions this as an evolution in N2K into a 'Trans-European Nature Network' (COM/2020/380; European Environmental Agency, 2020). The legislative backbone to this project has been the so-called EU Biodiversity Law (COM/2022/304).

The Biodiversity Law has seen, particularly for its impacts on farmers, intense politicized pushback (Casert, 2024), which has seen the somewhat remarkable rebellion by the European People's Party grouping against its own Commissioner (O'Carroll and Greenfield, 2023; Elissaiou et al, 2023). Despite a final, nail-biting passage in the European Parliament (Canas, 2024), which saw the spatial targets reduced by a third to protecting only 20% of the EU landmass, well below the Kunming-Montreal Biodiversity Framework target, the legislation still has yet to pass in the Council and is looking unlikely to do so (Krzysztoszek and Cagney, 2024). With this legislation ever more likely to fail in passage, it is more important to evaluate how these layers of governance interact to produce potentially unique effects on conservation trends. This leads naturally to the research question.

Research Question

How does the multilevel governance of the EU LIFE Nature programme affect conservation trends for protected habitats and species?

The focus of this research question is not necessarily on the conservation outcomes themselves, but on the features of the LIFE Nature programme projects which compliment those conservation outcomes. The LIFE Nature programme is inherently linked to the EU's flagship conservation program, Natura 2000. Natura 2000 is a fundamentally multilevel program, and rests at the heart of the EU's 2030 Biodiversity Strategy. The 2030 Biodiversity is tied to the EU's so-called 'Biodiversity Law' which has seen intense politicization across the EU. Therefore, the LIFE Nature programme exists in a political space which is at once multilevel, and highly politicized. The inherently multilevel and politicized space in which the EU's LIFE Nature programme operates makes post-functionalism the best theoretical framework for exploring the research question.

Theory

Introduction to Post-Functionalism and Multilevel Governance

Post-Functionalism is one of the three main theories concerned with explaining, through predictable pathways, outcomes in European Integration. Post-Functionalism (PF) has its intellectual origins in the 2008 paper (Hooghe and Marks, 2008) by Liesbet Hooghe and Gary Marks describing an alternative to functionalist theories of integration, Neofunctionalism and Liberal-Intergovernmentalism. Central to these theories is the assumption that a mismatch between scale and efficiency drives a unidirectional integration process of varying speeds

(Niemann and Ioannou, 2015; Schimmelfennig, 2018). It is then the *functional advantage* of integration which makes it likely to occur, and impossible to retrench. These theories then content themselves with asking who drives this integration, supranational actors (Sweet and Sandholtz, 1997) or Member States (Moravcisk, 1995); what drives this integration, bargaining (Moravcisk, 1995) or spillovers (Schmitter, 1969); and at what speed does this integration occur, at the lowest common denominator (Moravcsik, 1993), or gradually increasing (Sweet and Sandholtz, 1997).

Post-functionalism responds to both theories by at once seemingly incorporating them and asserting their historical placement. They are said to reflect European Integration best for a time characterized by a *permissive consensus* in European mass politics, which has since Maastricht, been steadily transformed into a *constraining dissensus* (Hooghe and Marks, 2008; Wilde, 2012). This inflection between periods has supposedly been marked by the degree of integration, and the politicization of the tensions linked to this degree of integration (Hooghe and Marks, 2008). The period since Maastricht has seen the repeated role of referendums on the future of Member States' integration with the Union as politicized instruments, with none more shocking, or vindicating to PF, than the 2016 Brexit Referendum (Czech and Krakowiak-Drzewiecka, 2019). In this way, PF asserts the importance of mass politics within the functionalist framework, arguing that politicization interacts with functional pressures to produce variable outcomes in (dis)integration (Hooghe and Marks, 2008).

The nature of this politicization then becomes the driving question for PF regarding how and why (dis)integration occurs. To answer this, PF introduces a framework by which politicization is assumed to interact with function in producing variable (dis)integration outcomes. Like Functionalists, Post-Functionalists agree that the impetus for reform is a functional mismatch between the form and function of the Union in given policy areas (Hooghe and Marks, 2008). Therefore, function still matters to PF, albeit in a modified way. Thus, the framework is as follows: This functional *reform impetus* leads to *issue creation* as a balancing of positions between political parties, interest groups, and public opinion. The interaction with these issues drives the selection by political parties of the *arena*, mass politics or among interest groups, in which this interaction will take place. The *rules* around the arena underpin the selection process by these parties. The rules then define the *conflict structure* along which these issues are confronted, whether it is biased towards 'identity' or 'distribution' (Hooghe and

Marks, 2008). Herein, distribution should be read as a reflection of the technocratic solution to the functional reform impetus.

The pillars to this framework each generate many questions for PF research on European Integration. These questions focus on, among others, identity, mobilization, parties, interest groups, group bargaining, and of course, function. PF then introduces the importance of structure, institutions, legitimacy, democracy, psychology, and power into what had previously been a domain of theory driven largely by economics. In doing so, PF modifies the roles of Member States and Interest Groups in their influence on European Integration, bridging a narrative gap between Liberal-Intergovernmentalsim and Neofunctionalism wherein integration occurs both daily, and in grand moments (Sweet and Sandholtz, 1997).

The nature of this bridging reflects the unique approach of PF to what European Integration *is*. Whereas both Liberal-Intergovernmentalism and Neofunctionalism see integration as a one-way process of centralizing authority at the European level, driven notably by different actors, PF explicitly argues that this variable process is essentially multilevel (Hooghe and Marks, 2008; Hooghe and Marks, 2020). In this context the centralization of authority is said to have low transaction costs and low flexibility, Multilevel Governance (MLG) systems are said to have the opposite (Hooghe and Marks, 2003), and for this reason, are said to be defined by *coordination problems*. In this way, integration for MLG means a process by which the operational relations in decentralized governance become increasingly interdependent, with the question driving integration focusing on how this interdependence is coordinated. Thus, multilevel governance is a description of how some types of governance *are* (Hooghe and Marks, 2003), and a structure driving the decentralization of authority *from* the State *throughout* the European Union.

On this latter point, there is debate. While it is agreed that MLG provides a compelling description to the appearance of multilevel systems, it is said to lack a causal mechanism linking that description to increased or decreased integration (Jordan, 2001). For this reason, the primary task in the application of MLG is in clarifying the mechanism by which it is suggested to produce an effect. For this reason, while PF was introduced alongside MLG (Hooghe and Marks, 2008), it does not tend to be considered critical to the theory and is instead treated more as an addendum. The remainder of this section will attempt to trace an outline for how this may be rethought.

As a framework unincorporated with PF, MLG describes a form of structural arrangements wherein governance occurs in a system defined by the decentralization of power from the Central State (Hooghe and Marks, 2003). Herein, MLG is the act of governing from multiple, interrelated points of authority which reflect the governed subject (Hooghe and Marks, 2003). In this way, MLG is considered to have two main types defined by their functional and jurisdictional mandate (Hooghe and Marks, 2003). Type I MLG describes a Russian doll of nested traditional authorities with non-functional mandates. The interlinked layers of Type I MLG are said to represent community insofar as their non-overlapping, jurisdictional territoriality is defined by people of specific commonalities, along lines of religion, culture, language, and more (Hooghe and Mark, 2021). These authorities are defined by a general competence to govern, and in the paradigm of 'voice' and 'exit', represent voice (Hooghe and Marks, 2003, Hirschman, 2011). In the EU, this Type I governance is typically illustrated by a Russian doll arrangement between Localities, Regions, Member States, and the EU.

Type II governance on the other hand describes functionally specific authorities whose mandate generally stems from Type I authorities (Hooghe and Marks, 2003). These delegated authorities are typically tasked with the provision of a single public good, off-hand examples being waste disposal or affordable housing registries. Type II governance reflects a technical mandate to address specific policy problems, whose scale of authority is proportionate to the efficient level at which such policy problems are best addressed (Follesdal, 1998). This assumption, that certain public goods have appropriate scales for their provision, also known as the subsidiarity principle, is baked into the EU and reflected in Protocol One and Protocol Two of the Treaty of the Functioning of the European Union (Pavy, 2024). In this way, the jurisdictions of these Type II models are said to be fluid, responsive, and at times overlap (Hooghe and Marks, 2003), meeting the functional needs of their mandates. It is important to note here that while the mandate of these Type II nodes is often conferred from Type I counterparts, this is not conceptually necessary. The governance from such nodes may come from NGOs or otherwise unincorporated (to Type I nodes) organizations who nevertheless provide a functionally driven public good, such as (among others) independent media or human rights organizations. This observation comes from the distinction between government and governance, whereas the latter refers to collective decision making, or decision making on behalf of a collective (Hooghe and Marks, 2020). Therefore, Government governs, but not all governance is done by Government. On this point, Type II nodes do not necessarily need to be related to government, so long as they govern, and in the Type II typology, provide a public good or service (Hooghe and Marks, 2003).

The nested Russian doll nature of the Type I model represents a clear verticality between nodes, and offers an ascending expansion of horizontality, such that each ascending node in the model naturally encompasses a wider territorial extent, equal to the territorial sum of each nested descending node. The Type II model is more dynamic. Herein, the Type II nodes are not only often dependent on the relevant, vertical nodes in the Type I model as a source of mandate (and funding), but also on the interdependent relationships between nodes at alternative levels with intersecting policy mandates (Hooghe and Marks, 2003). This latter point reflects the enormity of the coordination problem in MLG, which is functionally managed by a minimal number of Type I nodes and a strict mandate for Type II nodes. However, this means that the degree of verticality or horizontality between these Type II nodes is variable and depends on the specific compatibility of relationships between nodes. Thus, the horizontality and verticality of decision-making authority in Type II systems are more porous and represent a specific mesh of interdependence typically related to relevant Type I nodes, and functional compatibility between other Type II nodes.

MLG systems are in this way an amalgamation of Type I and II governance, a dynamic and evolving interrelational arrangement between the two. This evolutionary dynamism is fuelled by its embedded tensions and is at the heart of MLG's utility in not just describing itself within a MLG system, but also in forming expectations as to how decentralized (dis)integration happens. It is at this point that the post-functional rejoinder with MLG is necessary at illustrating this dynamism.

The Mediation Principle: (dis)integration = $(\text{Function and Identity})^{\text{Form}}$

The dynamism of Post-Functionalist Multilevel Governance is fueled by a two-part tension concerning what governance *is*. For Hooghe and Marks, governance is both the act of delivering benefits to a community, and a representational expression of that community (Hooghe and Marks, 2021). In other words, the tension fueling PF, that between function and identity, is a tension bound to governance as such, and must necessarily reflect the form of governance taken by the object in question. Therefore, form must necessarily mediate the relationship between function and identity. The pathway in which form is associated, tying MLG to the underlying principles of PF, is through the 'politicisation premise', which views function and community as constructions of political conflict (Hooghe and Marks, 2020) The assumptions underlying this mediation principle rests on rational choice in party behavior.

The variable levels of authority introduced by MLG are necessarily accompanied by the actors which engage authority at those levels. In the EU, this means a mixed bag of actors, political parties, interest groups, bureaucrats, and the relevant public, at each level. As each level is characterized by a different type of (to varying degrees) democratic Type I node, this also means that the five-part model of PF has applicability beyond the EU level in determining how policy is done. This association between the levels of Type I governance alone is best, not just for model simplicity in what amounts to a multi-actor, multi-level game, but also because the nodes of authority in Type II governance are often tied to a either specific Type I node, or constellation of Type I nodes. This does not, however, mean that it is only the Type I actors which are relevant in discussing integration, only that the actors are involved at levels corresponding to a Type I node.

The anatomy of this game for the EU at the present involves the EU at one level, the 27 Member States at another, 242 NUTS 2 regions, and nearly 100,000 Local Administrative Units (Eurostat, n.d.). Regarding the regions and local administrative units, it is important to note that the actual regions and localities with authority to govern do not always comply with the EU's statistical nomenclature (Gouardères, 2024). Therefore, the numbers are illustrative only of the more important fact that the number of Type I nodes involved in the EU's governance make for an incredibly complex field of behavior and outcomes for the many, many more actors involved with each node and the far more numerous Type II nodes associated with them. By focusing on actor preference, the numerical challenge presented by PFMLG can be somewhat sidestepped, leading to useful conclusions on the multilevel process of integration in the EU, and for the research question, how function is affected by different actors pursuing relatively independent interests in that decentralized but interdependent system.

Actor Preference

The first assumption on actor preference is that parties in power want to stay in power, and parties out of power wish to come into power. This assumption draws from the main features of the three general schools of thought in rational choice party behavior, that they engage in office seeking, vote seeking, and/or policy seeking (Strom, 1990; Budge and Laver, 1986). Treating these assumptions together rather than separately, that parties and politicians can act in ways which express either or all these behaviors, leaves the assumptions much more compatible with reality, and applicable to additional modeling. This is because, between each assumption, the underlying current remains that parties seek power. Furthermore, if parties seek power, then it is also expected that they do not seek to lose it when they are in power.

In democratic systems, politicization matters because of its relationship to party competition (Hooghe and Marks, 2008). Party competition thus mobilizes mass politics because it can be used to strengthen a party's position, and/or undermine the positions of others. This presents a tension for parties, because politicization and mass politics is thus at once a tool for parties, and a threat to their power. The necessity of navigating mass politics suggests its importance even when issues stay in the interest group arena. This is to say, issue politicization tends to continue after acts on those issues have been made, as there is an element of delay inherent between an action being made and its effect being felt (Batini and Nelson, 2003; Auerbach and Gorodnichenko, 2012). In this case, the true effect of public opinion on whether an act can be politicized for or against parties in power can only be known after the fact of its execution. This echoes the assertion by Marks and Hooghe that the shift to the *constraining dissensus* comes only after the effects of integration have begun to be felt, and those effects have been politicized (Hooghe and Marks, 2008).

Furthermore, in representative systems like the EU, the moment of real engagement by the public with the potentially politicized issues is on election day. In the interim between elections, whether they are parliamentary or a referendum, there are many other issues which may or may not be politicized. Whereas referendums have the distinction of focusing an election on a single issue, they still do not happen in a vacuum. So, the party competition which surrounds these elections politicizes in an environment where a basket of decisions and a basket of consequences have already been made and felt. Therefore, parties must navigate the *potential* threat of mass politics when they make decisions on issues, even if it never enters the arena of politicization, and parties must weigh this navigation in a context which necessarily includes all other decisions which they have made. For this reason, parties likely engage in short-term assumptions of what they believe public opinion to be at the next election, not the date when a decision itself is being made.

Because parties both benefit from and are threatened by mass politics, they are assumed to use the authority in their node to insulate from threats and to mobilize advantages by engaging in Political Entrenchment (Saint-Paul et al., 2016; Levinson and Sachs, 2015). Political Entrenchment is the process by which parties in power use that power to increase the likelihood that they will stay in power. While Political Entrenchment (PE) through reform is often considered a threat to the rule of law, and therefore illegitimate, PE is also functional, involving marginal changes to enhance the likelihood of continued governing through many small changes meant to subdue opposition and expand support (Levinson and Sachs, 2015). This functional entrenchment can be reflected in the selection of specific bureaucrats, changes in debt rules, or the expansion of welfare programs. This functionality thus ties in with the functional pressures presented in the five-part PF model, and presents the expectation whereby the methods in which parties engage in overcoming functional misalignments are expected to converge around policy solutions which tend to entrench those instituting parties. In this way, entrenchment aligns with the underlying assumptions of party behavior in vote, policy, and office seeking (Strom, 1990).

With this said, political entrenchment, even without legal reform, is often viewed as undesirable and illegitimate (Levinson and Sachs, 2015, Helmke et al., 2022). This raises the possibility of politicization and mass politics as a response to political entrenchment. Therefore, parties are expected to engage with their entrenchment most where it ties in with those functional pressures for policy action. In this way, entrenchment follows the opportunity to entrench, whereby functional mismatches present such opportunities. Whereas the actions taken by parties to entrench vary according to their constituencies and their relationships to others which threaten their status, the core of this 'power to entrench' is the authority to do so. In this way, all political parties in power are suggested to have a vested interest which tends to maintain the authority that they have in their node.

This supposed tendency to maintain authority is not absolute, and there are cases when the relinquishing of authority amounts to further entrenchment. These would be cases where the relinquishing of authority constrains the ability of others to threaten the status of ruling parties in the future more than it constrains those same parties' abilities to entrench at the node from which they govern. The Tory support for the Single European Act (SEA) represents a good historical example of this dynamic in action. Despite historically being the pro-European party in the UK (Keedus et al., 2018), the Tories remained unique in the EU among major conservative parties for entertaining eurosceptic discourse (Glencross, 2014). Under Thatcher, the UK is famous for having secured an opt-out from EU Common Agricultural Policy contributions on the basis of its supposed disproportionality (Keedus et al., 2018). This then presents a problem to why the Thatcher government agreed to the SEA, when it reduced the power of the Member States relative to the European Union through the introduction of Qualified Majority Voting (QMV), increased EU competences, and added powers to the European Parliament (Maciejewski and Verdins, 2024). The approach presented here suggests that the deepening of the Single Market through the SEA subjected the power base of their political rivals to destabilizing competition with the labor movements and productive capacities in other Member States. Furthermore, the areas of competence which were to be subjected to QMV, the free movements of capital and services, and a common tariff and transport policy, represented areas of less concern to Tory power and its ability to entrench than to Labour (Maciejewski and Verdins, 2024). In its historical context, this relationship seems more evident, as it reflects historical euroscepticism in the UK as a Labour party phenomenon (Glencross, 2014), and most tellingly, this support for the Single European Act (1986) under Thatcher came directly on the heels of Thatcher's (in)famous heavy handedness in breaking strikes and labor power across the UK from 1984 to 1985 (Towers, 1989).

This conception mirrors much of the Liberal-Intergovernmental (LI) take on Tory support for the SEA (Moravcsik, 1991). This is a good thing, because it suggests a common framework for understanding these grand moments in integration. Unlike LI however, which sees this dynamic as one supporting the policy goals of the Tories, this framework ties SEA support more explicitly to goals which disproportionately disadvantage the power base of the Tory political rivals. Because party competition matters in this case, so too does politicization, or the potential thereof. The Tories demonstrated a pre-emption of politicization and mass politics by weakening the connection between a particular base, the most threatening alternative to Tory power, and by constraining that alternative's ability to entrench in the future with the transfer of selected competences to a more empowered EU.

The creation of the European Coal and Steel Community (ECSC) is another example of this authority relinquishing dynamic. Unlike the Tory support for the SEA, this political entrenchment was much more explicit. The Schuman Declaration plainly states the intention of the ECSC was to make impossible the act of war between its members by binding the necessary materials for warfare to a greater community (Schuman, 1950). The genetics of the EU make it plainly to be an instrument which establishes the mutual security of political institutions by restricting the authority of those same institutions to engage in zero-sum politics. This of course echoes Carl von Clausewitz's famous principle that war is just 'politics by other means' (Clausewitz, 1918).

While thus far the focus of entrenchment has been on a relationship between parties ruling in a node, and others which threaten that power, it deserves to be mentioned that, while the language will remain the same, parties may pursue types of entrenchment which insulate them from elements which challenge their traditional locus of power coming from within their own parties. The Brexit referendum can be seen as a good example of this, as an attempt to suppress the right-wing faction of the Tory party and their potential allies, then Prime Minister David Cameron pushed for a referendum on the future of the UK's relationship with the EU (Evans, 1998; Glencross, 2014). For Cameron, addressing this rising element was necessary because their presence created a growing tension which threatened the traditionally pro-business, pro-EU core of Tory power, and by extension, the ability of the Tories to maintain power (Glencross, 2014). By pushing for a referendum, Cameron gambled on overall voter support for EU membership, and hoped for an outcome whereby the Brexit element of the Tory party could be silenced by a "once in a generation decision" (Sparrow, 2016), and that the source of that silencing could come from outside the Tory party; at once preserving the wider coalition of Tory power which included Brexit oriented MP support, while denying them their opportunity to affect the main pillar of Tory power. This intra-party entrenching suggests that the Tory element led by Cameron felt keen on maintaining the continuous politicized mobilization of EU skeptical voters. In this way, the Cameron-led Tories tacitly supported a political opportunism whereby their anticipated referendum outcome would not necessarily lead to a demobilization of EU-skeptic Tory voters. What remains to be seen is whether the pro-Brexit elements of the Tory party themselves actually wanted Brexit, or whether they too felt that it was an easy way to ensure a continuous mobilization of voters. In all, this suggests that political entrenchment occurs in intra-party politics, and can be engaged by individual politicians, not just parties.

While the aftermath of this call for referendum is widely considered a primary example of identity mobilization (Czech and Krakowiak-Drzewiecka 2019; Wendler and Hurrelmann, 2022), it deserves additional mention that the lead-up to the referendum was itself characterized by growing MP success in using this politicization to win seats within the Tory party and in UKIP (Glencross, 2014; Evans, 1998). This reinforces the idea that political actions themselves do not exist in a vacuum, and the politicization which those actions navigate both precede and follow them. This does not, however, necessarily violate the PF idea that functional mismatches lead to pressure for actions. The politicization in and around Brexit focused specifically on the functional benefits of the relationship between the UK and the EU (Glencross, 2014). While the specific language differed according to intended audience regarding the primary consequences of this unequal relationship (identity vs efficiency) (Dallison, 2019; Stewart and Mason, 2016), it nevertheless remained functionally oriented around the idea that the relationship itself was not optimal, and therefore needed to be abandoned. Thus, political entrenchment as party behavior comes in both the anticipation of future threats and the response to past threats radiating from sources located upwards, downwards, inwards, and outwards.

If parties may relinquish authority as a means of entrenching themselves from threats, then parties (and/or politicians) should also be suggested to try and expand their authority for the same reasons. In liberal democratic systems like those found in the EU, authority is bound in the idea of consensual governance, where this sought after authority comes from must be at least nominally given by consent. For this reason, the authority-having nodes must be to some degree convinced that it is in their interest to relinquish that authority. Therefore, the process of authority relinquishing and authority acquisition are one and the same, with a few additions. First, because parties are assumed to try and entrench themselves to avoid politicization threats, which are themselves rooted in functional mismatches, the subject area in which authority is sought after is expected to be specific to that functional mismatch. Second, because parties are assumed to relinquish authority only when it is advantageous to their position of power, authority seeking is expected to reflect a politicized threat to party power. Third, authority seeking is expected to succeed when the necessary node(s) in which existing authority is located accept that not relinquishing authority undermines their ability to stay in power. Fourth, because authority seeking is framed in politicized threats to power, authority seeking is expected to legitimize politicization along those same lines, in turn generating real politicization threats to parties in power in nodes which do not relinquish authority. Lastly, because authority seeking generates politicization threats, threatened parties are expected to respond to them, directly or indirectly, through entrenchment.

Rational choice and political entrenchment therefore provide a framework for evaluating how and why authority is dispersed throughout a multilevel system. Furthermore, it centers politicization as such, rather than the politicization of the polity, as the driver of (dis)integration. This recognizes that the construction of politicized subjects in the context of authority dispersion is only important insofar as there is an outlet for such politicization to affect relevant compositions of authority. While much of the PF literature focuses on how politicization activates mass politics (identity) (Hooghe and Marks, 2008), the decision here is to privilege how authorities navigate environments where the threats and advantages of mass politics are ever-present.

Thus far, the focus on actor preference has centered Type I nodes. The shift to Type II actors justifies this, as the actor behaviors in Type I nodes are also assumed to influence the behaviors of those in Type II. More to the point, Type II actors, especially those involved in the macro decision making of their node, are assumed to necessarily have their behaviors bound to the interests of those in power in near Type I nodes, as it is the functionality of Type II nodes which directly interacts with the functional mismatches that threaten the power of those in Type I nodes. Thus, this interest results in modifications to the autonomy and capacity of Type II nodes to act within their competence. This assumption rests on the idea that the source of authority for all Type II nodes are ultimately Type I nodes.

While this connection might be most clear in cases where the Type II actors come from a specific governmental authority, like an Environmental Ministry, this is also true for those seemingly disconnected from those Type I nodes, like independent media outlets. Yet, the independence of these Type II actors are both guaranteed by Type I actors, with, for instance, passage of laws protecting Type II actors from private people and/or each other (Canan and Pring, 1988), and threatened by it, with potential regulatory changes (Irwin, 2024). In this way, the source of functional authority for all Type II actors ultimately rests in the actions (or inactions) of Type I actors. In a sense, this echoes István Mészáros' assertion that the state (Type I node) represents '*mediation* par excellence' of all internal social relations (necessarily including Type II nodes), in which case, no actor has their autonomy, or set of possible behaviors, fully independent of Type I nodes (Mészáros, 2022). When applied to Type II nodes, this set of possible behaviors necessarily includes the ability to engage in governance.

Such as it is, this means that Type II nodes have a contentious relationship with Type I nodes, at once both bound to their interests, and threatened by them. With that said, there are some important distinctions within Type II nodes. Of course, the degree to which these nodes are bound to Type I nodes differs, most notably where those Type II nodes are dependent on Type I nodes for resources necessary to govern, and where the direction of those nodes are determined by bureaucrats appointed by those Type I nodes. This scenario reflects Principal-Agent theory, whereby the 'Principle', in this case the Type I node, delegates the authority to act on its behalf in a specific area to an 'Agent', the Type II node, which has its own independent interests in the way in which such an act is to be done (Miller, 2005; Braun and Guston, 2003; Barbieri et al., 2013). In such relationships, the Principal is defined as having a preference to reduce the autonomy of the Agent acting on its behalf, and the Agent to increase its autonomy (Miller, 2005).

These areas defined by a Principal-Agent relationship are those whereby the Type I node has the authority to delegate, and the way in which this delegation appears reflects the specificity of that authority. Thus, Type II nodes are sensitive to changes in authority in Type I nodes, as such changes can rearrange the form of their relationship to those Type I nodes. This rearrangement is highly specific, and the degree to which the Type II node's autonomy is threatened can come from either enhanced authority in the Type I node, such as a change in the Type II node's mandate, or a loss of authority, such as a decrease in overall resources available to Type II nodes.

Importantly, the rearrangement of the relationship between Type I and II nodes is not determined solely by a change in authority of the Type I node. Again, the changing of bureaucrats appointed by the Type I node or the changing of resource allocation can still occur *ceteris paribus*. This Principal-Agent relationship predicts that Principal-led changes occur when the Principal believes that the Agent is not acting within its interest (Miller, 2005) The interest, in this case, of Type I nodes reflects the parties in power, and their preference for political entrenchment. Therefore, Type II nodes, and especially those characterized by a Principal-Agent relationship, should demonstrate a balancing between their functional mandate and the political landscape in which they operate, meaning that the outcomes related to the functional mandate of these Type II nodes may not always be optimal.

On the Agent side, the behaviors are assumed to reflect a desire for greater autonomy (Jensen and Meckling,1976). Furthermore, Agents are assumed to be functionally oriented, meaning that suboptimal outcomes in functionality are not endemic to the Agent's interest, but rather a result of the balancing Agents do in the context of limited autonomy and resources (Spremann, 1987). Because Type II actors are assumed to pursue functional optimums, it is expected that they will cooperate most with other, policy related Type II nodes at various scales where it increases their autonomy, resources, scale functionality or some combination of all three. Additionally, as these Type II nodes are characterized by limited resources, it is also assumed that they pursue cooperation where it is more efficient economically to do so. Despite this cooperation, Type II nodes are still bound to the interest of their nearest Type I nodes, especially those in Principal-Agent relationships. This suggests that realized functionality in the presence of cooperation is likely better than without, but the realized functionality itself can still be suboptimal for some or all Type II nodes involved.

This suboptimal functionality reflects backwards on the five part PF framework, generating politicization threats which induce further political responses, and so on. In this way, functional mismatches are both responded to and produced by the same system - echoing the Neofunctionalist assertion of Spillover Effects, whereby greater functional integration begets greater functional integration (Schmitter, 1969). Unlike Neofunctionalism however, these spillovers take the form of threats to entrenched political power, meaning that the outcomes of actions addressing these threats are dependent on the historical specificity of the parties in power in a series of nodes. This approach to spillovers is consistent with, and adds to, discussions on vertical and horizontal electoral spillovers within MLG systems (Schakel and Romanova, 2021).

Theory Summation

In sum, the suggestion here is that ruling parties tend to follow the threatening actions of others within and beyond their node. These threats are functional, insofar as functional misalignment generates politicization threats, and those functionally inclined responses generate spillover threats. Herein, parties behave in ways which generate threats to others, and respond to threats which come from the anticipation of actions and reactions both realized and not. These actions are made to politically entrench those that make them. While this entrenchment tends to preserve existing patterns of authority, parties may often find it advantageous to relinquish authority, or to seek authority. In this way, normal political behavior generates ripple effects which drive continuous, system-wide adjustments to the Type I nodes in which the authority to govern is located.

The location in which authority is dispersed follows where it is best expected by the parties which relinquished it to entrench their political power. Likewise, the degree of that authority relinquished reflects best expectations by the relinquishing parties of the appropriate capacities sufficient to preserve their power. The newfound authority and capacity to govern thus comes with the expectation of its use in relation to the functionally related pressures which drove that node's authority acquisition. In this way, the changes in the location of authority generate functional demands on those gaining nodes, and new threats for politicization which the parties of those nodes must navigate as they respond to those new demands. In turn these responses radiate potential threats, and necessarily incorporate new Type II nodes into the governing process. These new and/or modified Type II nodes then generate their own functional demands, which radiate new potential threats to Type I nodes.

These Type II nodes seek to cooperate to preserve autonomy from the nearest Type I nodes, but balance that cooperation with near Type I node preference. The balancing of Type I node preference with the functional cooperation between Type II nodes leads to enhanced functionality only where it is mutually advantageous to the nearest Type I nodes. This generates system wide functional mismatches in policy areas where Type I nodes perceive full functional cooperation as a threat to their power. In this way, attempts by Type I nodes to increase overall functionality at the optimal scale in these policy areas drives politicization threats in the territorial extent of that scale. This, in turn, forces political parties to navigate those threats as they appear both within and beyond their node.

Thus, PFMLG views (dis)integration as a historical and evolving continuous *process* rather than an *outcome*. This process is understood to be driven by a decentralized dynamic between function and party politics, meaning that it is insufficient to reduce the engine of (dis)integration to either political compromise or functional cooperation, or a singular moment between the two. These two components drive one another, resulting in (dis)integration outcomes which are in no way guaranteed to be functionally optimal, or politically stable. The degree of functional suboptimality and political instability is mediated by the multilevel structure of the system, meaning that it is necessary to include the multilevel element of *form* to understand outcomes related to either element within the current arrangement on a given policy area.

Application of PFMLG to the LIFE Programme and Conservation

To understand how PFMLG may predict functional outcomes in the LIFE Programme, it is necessary first to provide a diagnostic of the program, the nodes involved, and their roles. The Programme for the Environment and Climate Action (LIFE Programme) was established at the EU level by Council Regulation (EEC) No 1973/92 in 1992 in relation to Article 130r of the Treaty establishing the European Economic Community, and has been maintained and updated since (European Climate, Infrastructure, and Environment Executive Agency, n.d.a). Since its creation, a portion of the LIFE programme has been dedicated to funding projects which protect and conserve nature in reference to the Habitats and Birds Directives (Council Regulation (EEC) No 1973/92), making the LIFE Programme the primary EU level funding instrument for enhancing the integrity of the Natura 2000 network. The LIFE Nature sub programme reflects this category, and for the purposes of this paper, is the sole sub programme of focus. While the LIFE Programme has undergone revision with each funding period since 1992, the Nature element has always been tied to the Natura 2000 network, its enhanced integrity, and ensuring designated species and habitats meet a favorable conservation status (Council Regulation (EEC) No 1973/92). Thus, the LIFE Nature Programme will be considered in this respect as unchanged between periods.

As a project funding instrument, the LIFE Programme has open calls for project proposals which fit in with its programming objectives. The European Climate, Infrastructure, and Environment Executive Agency (CINEA) acts on behalf of the Commission to select submitted projects which fit with the regulation for the funding period, and oversees the allocation of funds (European Climate, Infrastructure, and Environment Executive Agency, n.d.b). As a co-funding instrument, the LIFE Nature programme finances a portion of the overall selected project costs. This portion ranges up to 75 percent of the total project costs, depending on the type of project and whether they target species at risk of extinction, or habitats at risk of disappearance (European Climate, Infrastructure, and Environment Executive Agency, n.d.b;

Regulation (EU) 2021/783). In this way, the Commission (Type I), through CINEA (Type II), exerts control on the types of projects selected, and defines the overall framework objectives within which all submitted projects must comply. Thus, the Commission acts indirectly as a Principal in the delegation of projects, which are themselves Agents in implementing projects on behalf of the Commission in specific locations.

In the context of the research question, which is most focused on who does what and why in relation to what it means for conservation, the first task is to understand the effect of the LIFE Nature programme, that is, does it enhance conservation as it is intended? Because Type II nodes are assumed to be functionally oriented, this leads to hypothesis one:

EU LIFE Nature Projects improve conservation outcomes.

As these projects have applied to the Commission for funding, it is also assumed that the goals of the project leaders align with or reflect interest in improving conservation outcomes. For this reason, it is expected that the Commission and the project teams have shared interest in the project outcomes. In this case, sub-optimal project results are likely to be found on a patchwork basis, reflecting varied external social and political relationships to these projects. The countries in which sub-optimal outcomes are expected are those which have been brought to the European Court of Justice for failures in meeting the designation obligations of the Habitats Directive, those being Denmark, the Netherlands, Germany, France, Finland, Greece, and Ireland (Paavola, 2004).

The assumption of responsiveness on the part of Type I nodes, and the assumption of functionality among Type II nodes together would suggest that the Commission (or specifically CINEA) has an interest in actively engaging with changes in conservation trends by selecting specific projects. This leads to hypothesis two:

EU LIFE Nature projects are selected to target areas with worse conservation status.

Here, with the power of project selector, the Commission is assumed to play the role of allocating funding where it expects funding to matter most. As the LIFE Nature programme is the Commission's largest instrument for developing the integrity of N2K, failure to use it would amount to a functional mismatch where the Commission already has the authority to act. Because pursuing optimal outcomes for the *species and habitats*, not projects themselves, is the purpose of the LIFE Nature programme, it is expected that the Commission will demonstrate responsiveness to changes in the conservation status of those same species and habitats, thereby using its power as project selector to target them.

Species and habitats do not, however, exist in a vacuum. Their conservation status is directly dependent on the conservation status of other species and habitats (Harvey et al., 2017). In context, this means that the targeting of the species and habitats of interest is likely to focus on the areas in which they live, which are known to be of poor conservation status, and the project types themselves are likely to focus on those species and habitats on which target species or habitats depend. For this reason, it is likely that the focus of a functionally responsive Commission will be on the areas, more than the individual species, with the worst overall conservation status.

These N2K sites demand behaviors from those interacting with them to best reflect actions compatible with the conservation of intended species and habitats (European Environmental Agency, n.d. b). Therefore, the relationship local people have with the sites, and their buy-in to the sites objectives, are considered critical elements to the overall success of sites in improving conservation status (Kruk et al, 2010; Young et al., 2013). Reflecting this necessity is the fact that the designation of sites is not always popular (Maček, 2023). This leads to hypothesis three:

The Commission is likely to demonstrate preference for projects led by more local authorities.

Regarding the Political Entrenchment aspect of PFMLG, the relationship people have to political actions, namely N2K site designation, represents an area in which politicization threats may come. For this reason, political actors involved in these actions have an interest in mediating this relationship. For the Commission, LIFE Nature programme projects offer an opportunity to mediate that relationship positively, if (importantly) the actors implementing the projects share the same interest. Likewise, Political Entrenchment would also assume that more-local authorities have a greater interest in mediating individual relationships between people and programs, as the power of their vote relative to the sum of voters in that node is greater in more-local nodes. For this reason, more-local authorities which *apply* to lead projects within the framework of the LIFE Nature programme are likely also to share interest in accomplishing N2K network goals, and therefore, most likely to engage with local stakeholders in ways which try to positively mediate their relationship with the N2K network. This leads to hypothesis four:

LIFE Nature programme projects led by more-local authorities have the greatest positive impact on conservation status.

Hypothesis four reflects largely the same assumptions of hypothesis three. As the role of local buy-in to N2K site outcomes is considered an important feature in the successful impact of site outcomes (Kruk et al., 2010), and because the applying local authorities have a greater interest in maintaining a positive relationship between local stakeholders, the outcomes of more-local authority led projects are likely to be the most positive in terms of conservation status objectives.

Within the PFMLG framework, Type II actors are assumed to have their interest reflect that of the nearest Type I node. At the EU level, this assumes that CINEA's interest reflects the Commission. At descending scales within the Type I structure of the EU, this means that the associated authorities share their interest most with more-local Type I nodes instead. Therefore, those authorities are likely to engage with the LIFE Programme where it most likely reflects a balancing between those interests and the advantages that are conferred by LIFE Programme participation. In the context of more-local authorities, this is expected to correspond with a balancing of conservation interests, and the reduction in resource constraint that comes with project co-funding.

In this way, more-local authorities are expected to engage in projects where it maximizes the general conservation goals of those authorities. This takes into consideration the idea that N2K is not the only conservation system in the EU. Member States, Regions, and Localities have their own protected areas (Abellán and Sánchez-Fernández, 2015; Evans, 2012) and engage in other forms of international conservation efforts such as the Ramsar Convention for protected wetlands (Evans, 2012). Such as it is, EU Natura 2000 funding from the LIFE programme can be tangentially applied to these other protected sites when and where those sites intersect with N2K. This leads to Hypothesis Five:

Authorities are more likely to engage with the LIFE Programme in sites intersecting with multiple designated areas.

The PFMLG framework suggests two pathways through which authorities would engage LIFE programme projects in sites with multiple designations. The first is in the reduction of potential politicization risk through conservation projects. These may include stated goals of biodiversity protection in protected areas, such that certain areas or species carry greater significance to a mobilized community. More compellingly, it may include related goals, such as those tied to ‘nature based solutions’. An example of this latter point may be in restoring wetlands as a means of reducing the risk of severe and damaging flooding (Ferreira et al., 2021; Thorslund et al., 2017). Relatedly, it may also be expected that such site selection represents interest balancing, insofar as areas with a high number of protected area intersections are perhaps those in which conservation priorities have definitively ‘won out’. Therefore, such areas represent a low-risk project site for authorities, especially those most constrained by local interest.

The second pathway would come through enhanced autonomy seeking. In this instance, the functionally oriented, yet constrained, actors apply for EU co-funding to increase their capacity to accomplish goals which would be unlikely to occur from within the node. With a program as politicized as N2K, even sympathetic parties may be hesitant to fully support N2K site development, especially where budgetary resources are the most constrained. Through this pathway, it would be expected that least endowed authorities are most likely to apply for projects as a way of bypassing explicit near Type-1 node consent. Thus, site selection would be expected to fall in areas which maximize conservation goals relative to the capacity to achieve them. Multi-intersection sites are likely one such spot, making their selection ideal in applying EU funding to areas less likely to receive project funding on their own.

Furthermore, the PFMLG framework suggests that the Commission does not necessarily have a problem with this. For one, it means that the authority with the most knowledge and experience with a site and all the legal frameworks associated with it is the one implementing the project, thus increasing the likelihood of positive conservation outcomes (Joa et al., 2018). Furthermore, the availability of funding for sites intersecting the N2K network makes more likely the participation of those nodes in the LIFE Programme and the N2K network. This in turn could reduce politicization threats to the Commission by descending Type I nodes who might otherwise not see the benefit for participation.

With that said, it is important to reiterate that N2K sites guarantee a *minimum* level of legal protection to the species and habitats that they are assigned to protect, but not a legal *maximum*. In this way, N2K sites are compatible with and often exist in and on top of sites designated by national authorities for reasons and programs which pre-date and/or precede Natura 2000. Thus, National Parks, Strict Conservation areas, fish hatcheries, and more may all be included within the Natura 2000 network but are themselves primarily not Natura 2000.

Because the funding instrument of the LIFE Nature programme does not distinguish between sites within more strict conservation frameworks so long as the sites themselves are within the Natura 2000 network, and so long as the projects are meant to enhance conservation status for Nature Directives species and habitats, there is no reason why LIFE Nature funding cannot be applied to those nationally (or subnationally) designated sites. In this way, sites run the risk of not only attracting EU funding to those which intersect other priority protected areas, but also of diverting that funding to sites which have a higher status of protection and are thus less likely to need conservation status enhancing projects.

Therefore, the enhanced conservation status of species and habitats within these nationally designated sites is likely to correspond with additional benefits to the relevant Type I nodes. These benefits are expected to be conferred according to the particularities of the site in question, but in sum, the overriding assumption is that there are additional benefits conferred. Within the PFMLG framework, these benefits are suggested to insulate from politicization threats. Furthermore, the enacting of projects within areas fully designated nationally (or subnationally) but still within the Natura 2000 network means the enacting authority can interact with stakeholders on behalf of the designating authority, not necessarily on behalf or interest of the Commission. This means that the implementation of projects in these areas primarily represents opportunities for Type I nodes to mediate non-Natura 2000 politicization threats with LIFE Nature funding. The co-funding mechanism of the LIFE Nature projects likely means increased funding for projects that already would have been, suggesting more resources to ensure a higher quality outcome.

While the Commission might not be said to have a problem with this dynamic from a PFMLG perspective, this is not the same as saying the Commission prefers it. While this expected behavior nominally fits within the Commission's overall objectives with the LIFE Programme, and is thus permissible, it is not necessarily optimal. A key aspect of the EU's overall strategy with the Natura 2000 network is site connectivity, whereby sites act as an interconnected series of corridors for species and habitats to live and maintain their integrity. This fits with the recognition that even highly mobile species, like birds, may not on their own re-colonize a site even if it is very close, so long as those sites are not connected (Stiling, 2014). Therefore, site connectivity is the critical element for the overall network's success, and the widespread development of sites as such is said to expected deliver the greatest outcomes in terms of conservation status. This then leads to Hypothesis Six:

The Commission likely has preferences for projects in areas which have the minimum legal protection as only a Natura 2000 site.

This returns to the Commission's role as project selector and follows the assumption that the Commission has an interest in the overall integrity of the Natura 2000 network. Furthermore, this rides on the PFMLG assumption that the Commission has an interest in meeting a more optimal functional outcome in Natura 2000 sites as a means of insulating itself from politicization risks which may come from either continued biodiversity decline, or the need to seek additional authority to address the functional mismatch stemming from an inefficient policy programing on Biodiversity at the European level.

In sum, these six hypotheses paint an introductory picture to how the Multilevel Governance of the LIFE Nature Programme affects conservation status. It is expected that the overall outcomes for the program are functionally connected, that the Commission selects projects responding to functional needs, that Type I and Type I related actors cooperate, that this cooperation is functional, but that it is not always optimal. Lastly, it is expected that the Commission is aware of the suboptimal outcomes related to Type I cooperation, and thus seeks projects in Natura 2000 sites which are not related to other forms of nationally or sub-nationally designated conservation and conservation adjacent areas.

Method

To explore the above hypotheses, a quantitative approach will be taken. This approach will be used to provide both internally and externally valid suggestions about underlying trends in the multilevel governance of the LIFE programme. Taken together, these trends should begin to provide some conclusion to the research question. The quantitative approach taken will rely on cross-sectional analysis to provide broad assumptions about the programme's governance across the EU (Kesmodel, 2018).

EU LIFE Nature Projects improve conservation outcomes.

Hypothesis one centers around the general functional element of the research question. To build a test around this hypothesis, data was first gathered for Conservation Trends. This data comes from the EEA's Article 12 and Article 17 Nature Directive reporting data (European Environmental Agency, 2022a; European Environmental Agency, 2022b). This reporting data is gathered by the EEA based on mandatory Member State reporting on the status of species and habitats listed in the Annexes to the Nature directives. For the Article 17 Habitats Directive reporting, the first reporting period was 2001 to 2006 and the second 2007 to 2012, whereas reporting for Article 12 Birds Directive was first 2008 to 2012 (European Environmental Agency, 2022a; European Environmental Agency, 2022b). Their latest reporting period was 2013 to 2018 each. This data covers the entirety of the EU, including the UK.

To address Hypothesis One, public data was used from the 2013-2018 reporting period. Of note in this data is that it does not include all species. This reflects a contentious topic in public conservation data, whereby some species may be made worse off with public disclosure on locations and trends of those species (Lindenmayer and Scheele, 2017). For this reason, the data is not complete.

Together, this data represents a monumental, European wide effort to accumulate conservation information into a single place. Because of its pan-European nature, especially in the data gathering, there are degrees of difference between the reporting methods between states, and within states between teams measuring each species and habitat. Fortunately, the data includes a 'use for statistics' column whereby the differences in techniques are evaluated by the EEA for their usability and comparability. This test thus used only data which was coded as usable for statistics. Before moving on however, it is worth noting that the gathering of conservation data is incredibly difficult, especially in the context of species and habitats in decline, as it becomes much more labor intensive to argue that a species no longer exists in an area (Daigle and Janicki, 2022).

This data is also reported at different levels between datasets. The Article 17 dataset reports conservation trends and information at the European and Member State Biogeographical regions (European Environmental Agency, 2022b). These biogeographical regions reflect the major biotic zones in Europe, including European Marine territory (European Environmental Agency, 2016). For this test, only the Member State Biogeographical region data was included. This allows for distinctions in trends between Member States, whereas the European Level data reflects only continent-wide trends. By including the Member State trends, the data can be made slightly more specific, and thus more operable and responsive to the independent variable.

Article 12 reports trends and information at the European and Member State levels (European Environmental Agency, 2022a). While the Member State level data was also chosen from this dataset, a question is raised by this distinction, namely whether the data between datasets are comparable. That is, whether the differentiation between reporting areas alters their relationship between one another enough that their inclusion in the same dataset renders test outputs nonsensical.

For the purposes of this paper, these data are assumed to be comparable, at least in their mutual responsiveness to the independent variable. This is assumed because the habitats, species, and birds each have different degrees of mobility. Herein, it is assumed that the generally wider mobility of bird species makes their relevant reporting range much wider, with that range descending to other species and habitats. Now, this is not to say that all birds have the same ranges. It would be ridiculous to assume that a year-round species of duck has the same range as a migratory stork (Bobek et al., 2008). With that said, the assumption is that, on average, terrestrial mobility descends between birds, other species, and habitats. Because of this assumed varying mobility between habitats and species, the range in which they are reported is assumed to reflect a generally comparable scale for that reporting. In this way, the responsiveness between both Article 12 and Article 17 data, and the independent variable, should be similar enough that their use together is justified.

Hypothesis One presents two potential areas of interest regarding the effectiveness of LIFE Nature projects in affecting conservation trends. First is the question of whether LIFE Nature projects may improve conservation trends in the reporting area, and whether they improve conservation trends in the sites in which they are undertaken. Given the data, the first is the most doable. However, this paper will attempt to do both.

The data from the reporting area trends will be bound based on that data's reporting area. In this way, the reporting area data will be reflective of the directives, and not a combination of the two. For the Birds Directive data, the reporting area will be the Member State, and for the Habitats Directive data, the reporting area will be the Member State biogeographical region.

The Article 12 and Article 17 data is also spatial, insofar as the EEA suggests how the reported conservation trends may be applied spatially to a 10 by 10 km European wide grid (European Environmental Agency, 2022a; European Environmental Agency, 2022b). The conservation data was then combined with the EEA's European wide 2018 Natura 2000 spatial data (European Environmental Agency, 2023a). The location of the sites was then validated against the biogeographical data in the Natura 2000 biogeographical tabular data (European Environmental Agency, 2023a) to ensure that the spatial data was appropriately joined. Lastly, the tabular Article 12 and Article 17 (European Environmental Agency, 2022a; European Environmental Agency, 2022b) reporting data was merged to select statistically valid (use for statistics column) Natura 2000 trend information for species and habitats within the reporting period. These two columns, Natura 2000 trends and reporting area trends, were then merged into a single column whereby each reporting area trend was substituted for the Natura 2000 trend where they did not match.

Lastly, to create a sort of fingerprint for each site, the 2018 Natura 2000 tabular species and habitats designation data was merged and used to select the conservation information for each site relating only to those species and habitats for which that site was designated. In this way, each site was then provided with the mean value corresponding to those conservation trends. This value reflects the coding given by the EEA for conservation trends, that they are experiencing either Deterioration, No Change, or Improvement. These categories were then given the values of -1, 0, and 1 accordingly.

There are two things to note here. First these categories are assumed to represent distinct, hierarchical, and non-overlapping values, such that No Change is better than Deterioration, and worse than Improvement. In the context of the actual status of the species and habitats, it could be the case that No Change in population size for instance is the most desirable outcome, reflecting a healthy population. However, the protected status of these species and habitats seems to reject the idea that their populations and integrity are in fact healthy. Furthermore, the question of status is not applicable to trends, such that No Change is neither a Deterioration nor Improvement.

The other feature of note is that the categorical classifications are not necessarily comparable between species and habitats, such that the deterioration experienced by one species may be much more severe than that experienced by another. Yet, this classification reduces them to an indistinguishable degree. This is more problematic than the prior question, as it potentially obscures the true effect of LIFE Projects on real conservation trends. It is thus assumed that the true degree of difference between each species and habitat of the same category that their true conservation trends are equally under and overrepresented, such that on average, the classification of each conservation trend is appropriate.

By adapting these categories into numerical values, the trend information can be used to provide a little more statistical utility in additional analysis. Thus, the mean value for each site reflects the unique overall basket of conservation trends of species and habitats for which each site is designated. This basket is then comparable to other Natura 2000 sites. This method of 'fingerprinting', or reverse engineering, the conservation trend for each site raises additional questions. Most obviously is the question of whether the underlying biogeographical region and member state wide can be reduced to a single geographical space.

This is addressed in two parts. First, sites are likely designated around species and habitats which they can protect, that is, species and habitats which are either present in that site, or will, by some project, be reintroduced to it. Therefore, the conservation trend data is most applicable to sites which are designated for those same species and habitats.

Secondly, species, and to a lesser degree habitats, are not bound to sites. They can and do move between and from them (Sunblad et al., 2011; Kail et al, 2023; Opermanis et al, 2012). N2K is a *network* of protected sites, such that the positive influence on conservation from a site is not strictly bound to it. This reflects the idea that N2K exhibits Network Effects, whereby each additional site adds more conservation value than the previous (Swann, 2002). In this way, the true effect size of a Natura 2000 site project extends beyond the site itself. Therefore, underlying data reflecting an area appropriate for the mobility of such habitats and species is thus likely to reflect on the true trends experienced 'at' that site. In this way, the fingerprinting method seems at least somewhat appropriate in approximating true site conservation trends.

Before approaching the LIFE Nature project data, one last question deserves attention. The selection of the 2018 Natura 2000 information raises questions about the applicability of that data's use for LIFE Nature project data which extends from 1992 (European Climate, Infrastructure, and Environment Executive Agency, 2024b). Within this time, sites have been added, and site boundaries have been updated (Evans, 2012). Because of this, there may be a problem in capturing the true effects of projects. However, because the underlying data is based on the reporting area, and because the projects can only be undertaken in a Natura 2000 site, the boundary changes and additions of new sites should not need to be considered in the question of whether a site or reporting area has had a project.

The independent variable, LIFE Nature projects, have been gathered from the LIFE Public Database (European Climate, Infrastructure, and Environment Executive Agency, 2024b). To address this hypothesis, only projects which have been completed prior to the end of 2012 are considered. This reflects the assumption that there is a time delay between a project's conclusion and its effect being felt (Batini and Nelson, 2003). Given that the 2013-2018 reporting period data treats the data collected in the start of 2013 to that at the end of 2018 as equally comparable, projects completed near the end of 2012 are considered to also have an equal distance in time to both period start and conclusion. While this is clearly not reflective of reality, it nonetheless reflects the assumptions present in the underlying data, and furthermore, reflects a gap in time for the effects of such end 2012 projects to be felt.

As the LIFE Nature projects are meant to improve conservation outcomes, it is then expected that LIFE Nature projects will be shown to affect conservation trends positively. The independent variable will then be the number of projects in a site for the site-based elements of the hypothesis, and in the reporting area for the reporting area elements. This variable is obtained by simply adding the total number of projects per reporting area/site.

Graph One shows the distribution of conservation trends among sites. The Histogram demonstrates a relatively clear normal distribution of the data, with the mean of conservation trends falling below 0, or slightly deteriorating. This normal distribution suggests the appropriateness of an Ordinary Least Squares regression model (Long, 2008). While OLS models require normal distribution in their assumptions, they also require equal variance between error terms, or homoskedasticity (Long, 2008). To check heteroskedasticity, an initial OLS with no control variables was performed and measured with a Breusch-Pagan test (Breusch and Pagan, 1979). The test returned a p-value of 2.2e-16. With such a small p-value, the OLS model demonstrates high heteroscedasticity, violating a principal assumption of the OLS model. This is likely due to N2K sites which had one observation, overestimating the mean value around zero, and overestimating the true incidence of sites with values at 1 and -1, as can be seen on Graph One. The initial OLS and an OLS with the Member State control variable can be seen in 'Test 1' and 'Test 2' of Table One.

To address this, a Type 3 Heteroskedasticity-Consistent Standard Errors (HCSE) modifier was applied to the OLS model. HCSE estimators are additional statistical steps to help make inference in the face of heteroskedasticity more reliable (Hayes and Cai, 2007). The Type 3 HCSE estimators attempt to correct for high leverage points in the data and are considered the most reliable (Hayes and Cai, 2007). The output of the estimator regressions in Table One do not vary significantly from the output for the non-corrected OLS model, supporting the idea that the heteroskedasticity does not really make much of a difference, and that the model is a decent fit (Freedman, 2006). For this test, the Member States were used as controls. Because the five drivers of biodiversity loss (UNEP, 2023) are well within the purview of Member States to regulate and control, Member States are expected to have a significant influence on conservation trends. Similarly, Member States are expected to still influence, albeit to a much smaller degree, the likelihood of a site receiving a project. With the authority to designate a Natura 2000 site, the Member States have influence over how long a site exists, and therefore, how many opportunities a site must receive a project. Similarly, Member States may write and lead projects themselves.

Like Graph One, Graph Two also shows a relatively normal distribution around a slightly negative mean for conservation trends in reporting areas. This validates the true reflective value of Graph One despite its potential inherent bias, as the reporting areas have many more incidences than one. Of note, Graph Two has no observations of reporting area trends improving for all relevant species and habitats yet maintains a similar distribution towards a full -1 value.

This reflects on the general conclusion of more professional studies (Sundseth, 2021) that the status of Nature in the EU is generally deteriorating.

As before, the reporting area trend model also has a small P-value for a Breusch-Pagan heteroskedasticity test, though not as small, at .01667. As the P-value is less than 0.05, the model is assumed to also be heteroskedastic. Therefore, the model is also modified with HCSE estimators. Once more, the HSCE estimators had little difference between the modified and non-modified OLS.

Lastly, to see if there were any major differences in how additional projects may influence conservation outcomes, a Quantile Regression was performed. Unlike OLS models, Quantile Regressions do not assume an underlying normal or homoscedastic distribution of the data and residuals (Huang et al, 2017). Therefore, no additional testing is needed to try and correct model error. Quantile Regressions also have the advantage of demonstrating how changes in the independent variable affect different quantiles of the dependent variable. In this case, how changes in the number of projects affect conservation outcomes in areas with the worst and best conservation trends.

The Quantile Regression was used to look at the relationship between trends in Reporting Areas, and the number of projects in those areas. Unlike the OLS model, the Directive was also included as a variable of interest. The Directive is justified as a control because the Directives were implemented 20 years apart, and there may be differences in conservation trends reflecting this gap. Furthermore, it might be the case that the targets of one Directive are more prioritized by LIFE Nature Projects, resulting in differences in project numbers and conservation outcomes. As a variable of interest, it communicates information about how Directive targets perform by target with additional projects.

EU LIFE Nature projects are selected to target worse conservation trends.

This Hypothesis has a time-dimensional element to it, suggesting that the behavior in time n+1 is dependent on information in time n. Therefore, testing for Hypothesis Two was done by essentially inverting the variables for Hypothesis One. Instead of looking at the conservation trends in 2013-2018, Article 12 and 17 reporting area conservation trends were gathered from 2007/2008 - 2012 (European Environmental Agency, 2022a; European Environmental Agency

2022b). Additionally, the dependent variable, LIFE Nature projects, were selected from the LIFE Nature project web app (European Climate, Infrastructure, and Environment Executive Agency, n.d.) for 2013-2018.

Unlike testing for Hypothesis Two, Hypothesis One does not require a ‘fingerprinting’ of conservation trends. This is because it is assumed that the data to which the Commission and Member States may be responding is that which is submitted to the Commission and gathered by the Member States. While the ‘fingerprint’ data does come from the same dataset, it requires additional work on the part of the Commission and Member States, making it less likely to be impactful. Furthermore, the required reporting area by the Commission for the Nature Directives is assumed to be the desired reporting area. Therefore, the Commission is less likely to be interested in potential site-based changes in conservation trends. This also reflects the idea that the Natura 2000 Network is more than the sum of its parts, and the most important thing for the Commission is the area in which that network sits. Like Hypothesis One, the conservation trend data was also agglomerated to the reporting area mean. Similarly, the Member State is used as a control variable for the same reasons that Member States are assumed to influence project likelihood and conservation trends in Hypothesis One.

Graph 3 shows a histogram of 2013-2018 project counts in reporting areas. In the 2013 period, of the 93 reporting areas, only 4 had no LIFE Nature projects. While the seemingly normal distribution of the dependent variable might suggest another OLS regression, this would be inappropriate as the measured data is count rather than continuous. Therefore, a Generalized Linear Model Poisson regression is most appropriate (Hayat and Higgins, 2014). Poisson regressions reflect counts rather than continuous data and are particularly useful in the presence of non-normal data distribution. Instead, Poisson regressions assume that the model variance is equal to the mean, such that an increase in value is related to an increase in variance (Hayat and Higgins, 2014). This means that the Poisson regression does not rely on assumptions of homoskedasticity, making its testing unnecessary. However, because most GLMs do not require the assumption of homoskedasticity, it is necessary to link the error terms to the model so that it can provide some predictability (Hayat and Higgins, 2014). In this case, a standard log-link Poisson regression was used (Hayat and Higgins, 2014).

The test results can be seen in Table 4. The models had a close relationship between the residual deviance and degrees of freedom (157 rd and 91 df for Test 1, and 50.923 rd and 66 df for Test 2), suggesting a goodness of fit between the model and the underlying data (Hayat and Higgins, 2014). Test 2 has a lower AIC relative to Test 1 (362 vs 419), suggesting that it is the better model (Hayat and Higgins, 2014).

The Commission is likely to demonstrate preference for projects led by more local authorities.

Testing for Hypothesis three requires some assumptions about how preference is expressed in the LIFE Program funding relationship. Ideally, a test for understanding Commission preference would look at differences in project types which are submitted to the Commission and those which are approved. Unfortunately, this type of public project submission data does not exist. Therefore this test relies on the assumption drawn from Principal-Agent theory that both the projects selected represent the Commission's preference, and that the degree of autonomy granted to them by the Commission, in this case a measure of resources, represents the degree to which the Commission has a preference for that project (Miller, 2005). In this case, it is assumed that the types of projects are similar regardless of authority level, and therefore the largest difference for which an effect can be measured is in the coordinator type. This assumption comes in reference to the fact that projects can receive co-funding depending on whether they are meant to target species or habitats at risk of disappearance.

One problem with this assumption is that project costs may reflect the project type, such that projects with an overall higher budget may correspond to projects meant to address species and habitats at risk of disappearance. Furthermore, the overall project costs may correspond to the total resources a coordinator type has at their disposal, disproportionately favoring National authorities. Yet, because there is no limitation on which type of actor can target species and habitats at risk of disappearance, the coordinator types themselves is not likely to influence this outcome. To address this assumption problem then, the total budget will be controlled.

Even though the Commission does not limit the project applicants along project type, the Member States themselves may still influence the application process. Because the Member States have for each site and species a management plan may need to be considered in LIFE Nature projects, the sites may be more restricted in the types of projects that they may receive. In turn, this means that the higher-level authorities have access to the most sites within their territory, and thus, the largest possible number of project types. Vice versa, the smallest authorities may only have a few if any sites in their territory, and therefore may be limited by the site's management plan to the type of projects that they may engage in. In this way, the Member States' approaches to management may produce variation in the project types of various authorities, and in turn, affect the funding which is available to them for such projects. For this reason, the Member States will also be included as a control variable for Hypothesis Three.

Furthermore, the funding period in question is likely to matter, insofar as the LIFE Programme directives, and more importantly, the personnel overseeing project selection at the EU level, change between periods. Herein, the Commission's role as selector affects both whose projects are selected, and the funding apportioned to each project budget. Thus, the funding program period will be controlled.

The data for this test comes from the LIFE Public Database (European Climate, Infrastructure, and Environment Executive Agency, 2024b). From this database, information on the coordinator authority type, total budget, project year, and amount of the budget co-funded by the EU were gathered. This data reflects projects applied for in all call years between 1992 and 2020. This data was selected as it represents the maximum number of cases measurable. This is because the LIFE Public Database is no longer updated, in favor of the newer CINEA project portfolio (European Climate, Infrastructure, and Environment Executive Agency, 2024a). Unfortunately, the CINEA project portfolio does not include information on projects prior to 2014, nor does it include the coordinator type beyond their legal status as public or private.

As can be seen in the Table 5 summary statistics, there are 578 observations of projects with coordinators as authorities. These authorities include National Ministries, like the Danish Nature Ministry (LIFE Public Database, n.d.a), Regional authorities like the Polish Regional Directorate for Environmental Protection in Szczecin (LIFE Public Database, n.d.b), and local municipalities like Swedish Västerbotten County administration (LIFE Public Database, n.d.c). While the offhand that a sample size of 30 represents external generalizability is not technically true (Islam, 2018), the number of observations in each category seems sufficiently large that generalization is appropriate.

The dependent variable, EU Funding, is preferred over the percent of funding, as the percentage obscures the actual amount of resources apportioned to the project coordinator by the EU. This is to say, the percentage of funding is more reflective of the total budget and project priority than it is the degree of autonomy granted to the project coordinator. Graph 4 shows the distribution of this funding in 10000 EUR. The funding is right-skewed, meaning that most of the funding is on the low-end of the funding range, between 17000 EUR on the low end and 20.9 million EUR on the high end. This maximum value received more than 7 million EUR in funding than the next highest value, at 13.4 million EUR. This maximum value is a 2010 project in Spain aimed at restoring the natural range of the Iberian Lynx (LIFE Public Database, n.d. d).

Because the data is right skewed, an OLS regression is not the most appropriate. While funding is technically a count variable, insofar as certain values are not represented by money (fractions between cents for instance), it will be treated as a continuous variable instead. Given that the data is also positive, continuous, and does not contain any zeros, a Gamma Regression is appropriate (Elliot et al., 2015). Gamma Regressions are a type of GLM model which does not assume normal distribution or homoskedasticity (Elliot et al., 2015). Similarly to Poisson regressions, this non-assumption of homoskedasticity means a link function is required to pull predictive value from the variance in error terms. For this Gamma regression, a standard log-link function was also used (Elliot et al., 2015). As Test 1 with no controls was seen to have a low difference in degrees of freedom and residual deviation (residual deviance 525.51 and 575 degrees of freedom), this regression is considered to have a good model fit. The lowest relative AIC value between tests was Test 4, suggesting it is the best model tested to describe the relationship between EU Funding and Coordinator Type.

LIFE Nature programme projects led by more-local authorities have the greatest positive impact on conservation trends.

Hypothesis four, engaging in conservation trends, uses the same conservation data as tests in Hypothesis one. Engaging in authority types, hypothesis four also uses the same coordinator data as hypothesis three, though with some clarification. Like hypothesis one, responses to hypothesis four recognize the same assumptions in project effect delay. Furthermore, the fact that the latest Article 12 and 17 conservation trend information is from the 2013-2018 reporting period means that only projects which had ended prior to 2013 were considered.

The fingerprinted site data was judged the most appropriate for exploring this hypothesis as the PFMLG framework suggests that the reason for such difference in trend outcomes is local stakeholder buy-in. In this case, trend information more reflective of local trends is then considered to be the better choice. As the distribution of the trend data is the same as for hypothesis one, an OLS regression is suitable. Additionally, like hypothesis one, the Member State will be controlled for its potential effect on conservation, on project types through management plan development, and on project counts through timing of designation. Furthermore, as total project budgets can influence perhaps the quality of a project, and the number of tangential sites a project can address, total funding will also be controlled. Because the area of focus is on sites, and because a project can be undertaken in multiple sites, many of the projects appear more than once. This is not a problem because the interest is in the status of sites which have received a project by which type of coordinator authority. This duplication is reflected in the Table 7 summary statistics for hypothesis four as compared to those for hypothesis three. Notably, sites which have not had a project have been left out.

The number of projects a site has had might seem like a good candidate for a control variable, as the number of projects is likely to influence conservation trends. However, the same cannot be said for the degree to which the number of projects a site has had can influence whether a site receives a project by a coordinator of a specific type. It could be that the number of sites can be tied to the likelihood that a coordinator returns to a site, that is, projects may increase the capacity of those working on them to identify next steps for future projects. However, this cannot be said to bias a specific type of coordinator. For this reason, the number of projects a site has had will not be controlled.

Like hypothesis one, the initial OLS demonstrates a high degree of heteroskedasticity with a Breusch-Pagan score of 1.883e-11 in the model, suggesting the need for Robust Standard Errors. Table 8 demonstrates the test outcomes with the various controls before the Robust Standard Error modification, Table 9 repeats those same tests but with the Standard Error modification. Test 3 has the lowest relative AIC between the model tests, making it the most reflective.

Authorities are more likely to engage with the LIFE Programme in sites intersecting with multiple designated areas.

Hypothesis five introduces an additional spatial element to the overall research question, looking at how the differences in the overlapping legal boundaries on a site influence the selection of project sites. The dependent variable is then the number of boundaries which overlap a selected site, and the independent variable is the type of authority coordinating the project. The independent variable was constructed using data from the LIFE Public Database and the LIFE Nature project web app. From the Public Database, projects were filtered by coordinator type, local, regional, and national, and then merged with the overall LIFE Nature project site data from the project viewer which includes the Natura 2000 site codes for each site worked on in a project. Projects which were not sorted as an authority were then coded as non-authorities.

The dependent variable was constructed with the Natura 2000 spatial data for 2014, 2015, and 2016 (European Environmental Agency, 2023a), and matched with the European Environmental Agency's Nationally Designated Protected Areas database for those same years (European Environmental Agency, 2023b). The multiple years were selected to avoid the problem of boundary change. Judging the number of intersecting boundaries with a site in a year that is not the same can lead to an incorrect count of site boundaries. However, spatially joining the boundaries for over 27000 Natura 2000 sites to more than 100,000 Nationally Designated Areas is computationally demanding, so the number of years were kept at a minimum that was deemed appropriate. This appropriateness reflects the number of site-projects undertaken by an authority. The Table 10 summary statistics for these years show 74 site projects led by local authorities, 85 by national authorities, and 89 by regional authorities. The overwhelming majority of projects were led by non-authorities. With the smallest category count of 74 observations, the tests are expected to be externally valid.

The summary statistics include two more variables. The Degree of Privatization reflects the number of project participants coded as legally private divided by the total number of participants. This data is taken from the CINEA Project Portfolio (European Climate, Infrastructure, and Environment Executive Agency, 2024a), which only starts in 2014, and is treated as a control variable. Project privatization is assumed to influence the site selection because public authorities are assumed to have more knowledge about the legal frameworks surrounding each type of site overlapping a Natura 2000 area. In this way, projects which are more private are assumed to select sites with less intersections. Additionally, projects which are more private are less likely to have public authority as the coordinator, simply because it is impossible for a project to be fully private and to have a public coordinator. Because privatization is assumed to influence both independent and dependent variables, it is controlled. For this test, the median degree of privatization is 40 percent, with the interquartile range between 20 and 64 percent. For these projects, the majority are decidedly more public than private.

The second variable is the percent of a Natura 2000 site within a National Park. While the variable refers to National Park, the more appropriate name for it might be the percent of a Natura 2000 site within a protected area of IUCN category Ia, Ib, or II. The variable is referred to as National Park for simplicity, as the National Park designation corresponds to IUCN category II (Dudley, 2008). The IUCN category system is an international standardization of accounting for the legal frameworks around a protected area, where categories Ia, Ib, and II are the most restrictive types (Dudley, 2008). Whereas Category II refers to National Parks, Category Ia refers to 'Strict Nature Reserve' and Ib refers to 'Wilderness Area' (Dudley, 2008). These categories of protected areas correspond to a class of legal protection which is stricter than a Natura 2000 designation, or for other types of international protections, like a Ramsar site (Evans, 2012). This means that these classifications must be reflective of a National (or subnational depending on the State) interest, within which other site types can be designated, rather than an international agreement.

Because these special types of overlapped sites represent the interest of an authority, those same authorities may be overrepresented in such sites. Furthermore, as these site types differ from the Natura 2000 network, they necessarily add to the number of intersections that a site might have. Therefore, whether the site overlaps protected areas of these categories should be controlled. By identifying the percent of the Natura 2000 site which is overlapped by these types of protected areas, more information can be captured by the model. There is a difference between sites which overlap the border of a category Ia, Ib, or II, and those which are fully nested within them. In creating a continuous variable, the effect of this degree and changes in it is also captured.

Creating this variable involved selecting nationally designated sites of these categorizations and merging them together as a single ‘mega site’. This was then overlaid on the Natura 2000 sites, and the area of overlap was then divided by the Natura 2000 site area. As can be seen in the summary statistics, the overwhelming majority of sites do not overlap at all with a protected area of IUCN category Ia, Ib, or II. The interquartile range of observations is completely contained at zero percent overlap. Despite this, the mean overlap is about 12 percent.

Graph 5 shows the spread of site intersections, the majority of which are contained below 10. Because the distribution is right skewed and the dependent variable is selected site intersection counts, a Poisson regression is a good match to the data. However, an initial Poisson log-link regression indicates a poor fit to the data, with a large difference in the residual deviance of 7688.1 on 941 degrees of freedom. This difference, indicative of overdispersion, suggested an alternative model be used (Hayat and Higgins, 2014). Therefore, a Negative Binomial regression is performed instead (Hayat and Higgins, 2014). The initial Negative Binomial was performed and produced a much more fitting residual deviance of 959.18 on 941 degrees of freedom. Table 11 shows the results of the five tests. Test five has the lowest relative AIC suggesting that it is the most appropriate model for the data.

The Commission has preferences for projects in areas which have only the minimum legal protection as a Natura 2000 site.

Hypothesis six builds on the assumption that the use of LIFE Nature project funds in sites which are more fully protected by additional layers of legal frameworks is a sub-optimal for the Natura 2000 network. The testing for this hypothesis will reflect the same approach taken for Hypothesis three, whereby the total EU contribution to a project is assumed to reflect the preference of the Commission in the project’s design. The independent variable will attempt to see the relationship between this funding and the percent of a site within a protected area of IUCN category Ia, Ib, and II.

As the testing for this hypothesis relies on the IUCN category variable as used in hypothesis five, those same projects reflected in that testing will be used. Graph 6 shows the distribution of the funding for the three-year period from 2014 to 2016. The data for this period is a little right skewed, but it is not entirely clear. However, because the underlying contribution data between 1992 and 2020 is right skewed, a similar gamma regression as was used in hypothesis three will be used for hypothesis six. Similarly, testing for hypothesis six will include controls for the Member State and Total budget. Furthermore, as increases in the total number of protected area intersections are likely negatively correlated with a project being in an area protected as strictly or more as a national park, the number of intersections could influence the independent variable. Likewise, as the number of intersections increases, projects may require additional resources and time to account for potentially different legal frameworks. Therefore, the number of intersections will be controlled. Table 12 shows the results to the Gamma regressions.

Discussion

Hypothesis One

Tests 1 and 3 in Tables 1 and 2 suggest that LIFE Nature projects do have a positive effect on conservation trends, though only a modest one. While statistically significant with a small p-value, an additional project is only correlated with a three percent increase in conservation status. Now, in a context where the conservation status itself is relative, and not completely comparable between species and habitats, a statistically significant increase, no matter how modest, is noteworthy. However, Tests 2 and 4 of Tables 1 and 2 suggest that when controlling for the Member States, the positive impact on conservation trends becomes statistically insignificant. Interestingly, the Member States are shown to have varying relationships with added projects, such that some Member States like Denmark, Germany, and Sweden have poorer conservation outcomes in Natura 2000 sites with each added project.

While the PFMLG framework suggests that the countries which had experiences with compliance issues would have sub-optimal outcomes, a negative correlation is still surprising. Now, it deserves to be said here that a negative outcome is strictly in relation to the baseline category, which in this case is Austria. Therefore, while this seems like an indication of sub-optimal functionality, it could also be the case that Austria is above average in its project implementation abilities. This relationship changes at the Reporting Area level however, and only Slovenia, and Portugal are suggested to experience a statistically significant negative effect of added projects relative to Austria in both cases, and neither of these countries were those predicted.

The outputs in Table 3 for the Quantile Regression also reflect the importance of Member States in the effects of whether a project will have a positive impact on conservation trends. This is however modified by the varying measured quantiles, such that in Member States like Germany, additional projects in Reporting Areas in the bottom 25th quantile of conservation trends seems to have a negative impact on those trends. This negative relationship becomes positive however when the trends are positive in the upper 75th quantile. Again still, the most appropriate way of interpreting the values of additional projects on conservation trends is always in comparison to the additional trend values for Austrian projects.

The Quantile Regression in Table 3 also suggests that the Directive has a statistically significant effect on the conservation trend, such that in the 50th and 75th quantiles, the Habitats Directive species and habitats perform consistently worse when compared to Birds Directive species.

Hypothesis Two

Test One in Table 4 shows an Incident Rate Ratio of .78. In a Poisson regression, the IRR cannot be interpreted as a beta coefficient in an OLS regression, meaning that this does not represent a .78 increase per additional count (Hayat and Higgins, 2014; Coxe et al., 2009). For Poisson regressions, an IRR below 1 indicates a negative relationship between independent and dependent variables, and an IRR above 1 indicates a positive relationship (Hayat and Higgins, 2014). The IRR is not necessarily a percentage but is multiplicative to the dependent variable (Hayat and Higgins, 2014; Coxe et al., 2009). In this context, the IRR of .78 suggests that for each increase in reporting area conservation trend, the project count decreases by a factor of .78. Now, the p-value for this result is just above the typical .05 threshold at .057. This suggests that 94.43% of the time, there is confidence that the model would include the real-life IRR.

Because the p-value is so close, it seems justifiable to say that the model for Test 1 tends to suggest the Hypothesis is correct, that projects are undertaken where conservation trends are lower. However, the model for Test 2 which includes the Member State as a control finds that, like Hypothesis One, Hypothesis Two is very dependent on the Member States. Tellingly, this model shows that Finland, Bulgaria, Hungary, and Slovakia all have a statistically significant positive relationship with the general conservation trends and the IRR of additional projects. These four Member States are characterized by some of the most intact forests in the EU (Eurostat, 2023). Therefore, it may be tempting to suggest that this is what is being captured by the data. However, an alternative explanation recognizes Denmark’s near inclusion in this club, with a small p-value of .055, which puzzle’s this outlook given Denmark’s limited natural area (Eurostat, 2023). Instead, the differences in Member State outcomes may come down to different approaches to conservation, such that some Member States may prioritize achieving across the board Nature Directives Article 2 favorable status in specific areas, while others may favor the more broad and reactive approach assumed by this model (Council Directive 92/43/EEC).

For this reason, the model is likely demonstrating that these Member States are less likely to have poor generalized conservation trends in the first place. A different model might include the generalized conservation status of each reporting area to account for this fact.

Hypothesis Three

Test one with no controls shows a statistically insignificant relationship between coordinator authority type and funding. When including the LIFE Funding period controls however, this changes, and both Regional and National authorities are suggested to receive significantly more in EU Funding, controlling for the funding period. Because the beta coefficient of a Gamma regression represents a log, the coefficients have been exponentiated to make them easier to read (Nelder and Wedderburn, 1972; Elliott et al., 2015). Like the IRR in a Poisson regression, this means that when the beta value is less than 1, the model is demonstrating a decrease. In this case, being a Regional or National authority is associated with a 40 to 53 percent increase in project funding, controlling for the reporting period. Similarly, the model shows a steady increase in EU funding contribution size between funding periods, corroborating the steady increase in the LIFE Program budget since its first programming period from 1992-1995 (European Climate, Infrastructure, and Environment Executive Agency, n.d. a).

Test three shows the statistical significance of the total budget as a control variable, with not too big of a change between Test one with no controls. Test four, with the lowest relative AIC, shows a statistically significant positive 10 percent difference in funding for regional authorities and a 20 percent difference for national authorities, compared to local authorities. Furthermore, the program period has its effect size moderated when it is controlled together with project budget, providing a more accurate reflection of the increase in the EU's LIFE Nature budget.

Test five controls the authority type with the Member States. While the difference between authority type funding is shown to be statistically insignificant in model five, it does suggest differences in Member State funding, most notably that being Italy, Portugal, or Romania is associated with a statistically significant decrease in funding compared to Austria, to the tune of 58, 62, and 71 percent respectively. This negative difference remains for Romania in Test six, though falls away in statistical significance for Italy and Portugal. Notably, Latvia is also expected to experience a 38 percent decrease in funding as well, when controlling for authority type, total budget, and funding period. Lastly, being a national authority in Test six is suggested to come with a 22 percent increase in funding compared to a local authority.

These tests mark a firm refusion of hypothesis three. They suggest not only do local authorities among authorities receive the least in funding, all else equal, but the degree of funding increases as the coordinator's level increases. This could suggest several things in the context of the PFMLG framework. First, it could be that projects led by local authorities do not have a better impact on conservation trends than projects led by other authorities, and this funding reflects that. This remains in line with PFMLG interpretations. Alternatively, this could reflect on the capacities of different levels of governance to write a good project proposal. Because ascending authorities may offer better pay and more prestigious work, they may also attract the better candidates, leaving those less capable or those with less specialized backgrounds to write proposals at lower levels. Another possibility is that the Commission has a preference for projects led by national authorities among authority types. This could reflect on the connections which the Commission has to those national authorities, and that maintaining those connections are more cost effective for the Commission than maintaining those with subnational authorities. On this last point, this could also play into where the personnel for the Commission come from, where they are likely recruited from those same national authorities (Trondal, 2006, Trondal et al., 2008).

Hypothesis Four

Test 1 in the non-robust adjusted model suggests that local authorities perform the worst in improving conservation outcomes, with national and regional authority projects improving conservation trends 5 to 14 percent more respectively than those led by local authorities. With a p-value under .05, these are considered significant. In the Robust Standard Error adjusted model however, only the regional authority retains statistical significance. However, with a new adjusted p-value at .061, the national authorities are still close to statistically significant in their impact on conservation trends.

Test 2 in the unadjusted model introduces the total project budget as a control variable. This is accompanied by a decrease in p-value for the national authority as compared to Test 1 in the unadjusted model, and an increase in effect on site conservation trends. Likewise, in the robust adjusted model, the national authority now retains its statistical significance in its difference with local authorities. The overall project budget is suggested to not have an influence on site conservation trends.

Test 3 introduces the Member States as a control, which sees in the unadjusted model an increase once more, to 8 percent, in the statistically significant effect of national authority led projects on conservation outcomes. On the other hand, the control for Member States more than halves the still statistically significant difference between local and regional authority led projects. This remains true in the adjusted model as well. As seen in the results for hypothesis one tests, the conservation trends vary positively and negatively depending on the Member State. The fact that controlling for Member States reduces the positive impact on conservation trends of regional authority led projects likely reflects the difference in the relationships between Member States and their subnational levels of authority. Some Member States have overrepresented regions, like Spain, which are more or less autonomous, and which are active participants in EU led programs (Benedikter, 2009). Other Member States are far more unitary. By controlling for the Member State, the differences between European sub-national regions are also controlled.

Test 4 maintains the statistically significant relationship between national, regional, and local levels of authority. This is true also for the adjusted model. Once more, the model similarities suggest that controlling heteroskedasticity is unnecessary and that the models themselves generally capture the relationship between coordinator authority types and site conservation trends.

The results from these tables completely refute the hypothesis, and suggest a re-evaluation is necessary. Not only is the role of local authorities once again not as predicted, the tables also show once more the structural opposite of what was expected. The effect of regional authorities as project coordinators, moving from Test 2 to Test 3 however provides some pause. In context, the clear advantage of higher level authorities may be a function of the advantages in capacity that those authorities might have. Furthermore, because these advantages seem to favor national authorities only when the Member States are controlled, does it seem that the differences in capacity are themselves a measure of the relationship between those sub-national units and their State.

Hypothesis Five

Like Poisson regressions, Negative Binomial regressions have their coefficients reported in Incidence Rate Ratios. Test one suggests that the selected site boundary overlaps increase by a factor of 1.43 and 1.41 when the project coordinator is a local or national authority respectively, compared to a non-authority. Furthermore, these increases are statistically significant with small p-values. Regional authorities meanwhile have a small p-value just above the .05 threshold at .053, suggesting that the relationship is still meaningful. For regional authorities, the selected site boundary overlaps decrease by a factor of .77 compared to non-authorities. The divergence between authorities in preference for additional intersecting boundaries suggests that regional authorities are involved in leading projects in Natura 2000 sites which are more disjointed from other protected sites and areas, in effect developing conservation islands.

Test two introduces the degree of project privatization as a control variable. When project privatization is controlled for the regional authority IRR becomes positive at a statistically significant level, suggesting that the negative trend seen in Test one reflects more of a difference in the relationship authorities have with project privatization than with site intersections themselves. For non-authorities, the inclusion of private actors may correspond with a greater interest in conservation projects as a function of proximity, reflecting an aesthetic or ecosystem service benefit to themselves. On the other hand, the inclusion by authorities may reflect gaps in technical capacity to undertake certain elements of a conservation project.

Furthermore, controlling for project privatization leads to an increase in IRR and statistical significance for national and local authorities, with the rate of intersections increasing by a factor of 1.82 for national authorities, and a full 3.19 for local authorities. This dramatic increase in incidence rate of local authorities reflects back on the idea that authorities in general engage in projects which correspond to a balancing of interest between goals for protected areas of different types, and limited resources. In this context, local authorities are suggested to have the most limited resources, and therefore, are most likely to coordinate a project in sites with the most complimentary location to those other interests.

The dramatic change in IRR with the introduction of privatization as a control suggests that privatization plays an important role in mediating site selection. However, on its own, this effect is not so large, as Test two suggests that an increase in project privatization is correlated with a statistically significant increase in the rate of overlaps by a factor of 1.02. This increase in the presence of added likelihood for public authorities compared to non-authorities to lead projects in sites with a higher intersection rate may seem somewhat contradictory, especially in the context of regional authority IRR changes. However, the inclusion of private actors in a project may reflect the need for added specialized technical skills, and when the coordinator is an authority, that need for increased technical specialization is greater. The fact that the positive IRR is so small reflects that this greater technical specialization, while statistically relevant, can coincide with other private interests.

Test three introduces the percentage of selected sites within national parks (IUCN category protected areas Ia, Ib, and II). This is suggested to have virtually no influence on site selection compared to Test one with no controls. Test four controls for the Member States. While controlling for Member states does not substantially change the results from Test one, aside from moving a decreased though still positive national authority IRR from a strict threshold of statistical significance to .055 and inflating from Test one the statistically significant IRR for local authorities, the most interesting results come from the Member State tables. Whereas some Member States like Bulgaria, Estonia, Slovenia, and the UK are likely to have projects with a higher rate of intersecting boundaries than Austria, most Member States are less likely, with statistical significance for Spain, Finland, Hungary, Ireland, Italy, Latvia, the Netherlands, Portugal, and Sweden. This large spread of results, compared to Austria as the baseline, suggests a wide difference in how and where Member States designate Natura 2000 sites. That the IRR for local authorities, and to a lesser extent national authorities, remains significant when this variation is controlled, suggests robustness in the theory.

Test five evaluates the model in the context of all control variables, and finds that, compared to Test one, the closeness in statistical significance for regional authorities falls off, that national authorities have the same IRR with a somewhat larger but still statistically significant p-value, and that local authorities have a much larger IRR than Test one, with an intersection rate of 1.94 greater than non-authorities. This local authority IRR is accompanied by a much smaller p-value of less than .001.

Meanwhile, the degree of privatization is suggested to have a smaller IRR of 1.01 than in Test two. Additionally, the presence of other controls leads the percent of site area in national parks variable to have an IRR between 1 and 1.01 which is statistically significant. Lastly, the controls lead to an across-the-board reduction in statistically IRRs for Member States compared to Austria, except in the case of Ireland, which saw its IRR increase from Test four to .17 from .16. Test five has the lowest relative AIC, suggesting that it is the most appropriate model.

In all, these tests suggest an acceptance of the hypothesis that authorities are more likely to engage in LIFE Nature projects where they intersect with multiple protected areas. However, this does not seem to apply to regional authorities. The suggested relationship between regional authorities and the number of selected site intersections is not statistically significant, indicating that regional authorities and non-authorities behave similarly in site selection. This presents a small problem with the theory which would expect regional authorities to have a greater incidence rate than national authorities due to constraining resources and complementary interests. One interpretation of this difference is that regional authorities simply do not have complementary interests. This is hard to accept however, as regional authorities can and do designate their own protected areas (Dudley, 2008). In this context, perhaps regional authorities have an interest in sites which are completely within the boundaries of self-designated sites of lower IUCN protected classification of II. In such a case, there may be fewer intersecting boundaries, but still relevant overlap in interest. This suggests the need for further testing, controlling for the percent of overlap with all other types of protected areas.

Hypothesis Six

The tests for hypothesis six show no effect of project site overlap with protected areas on the total project budget. With an exponentiated beta coefficient of 1 for each test, the models fail to reject the null hypothesis that there is a difference in outcomes. Furthermore, with a residual deviance of 60.208 on 942 degrees of freedom for test one, the models demonstrate a poor fit to the data.

The outcomes for the hypothesis six tests likely reflect two elements. The first element is a lack of data at a level which would correspond to a better fit to the distribution of the data. As this test was only done with a cross-section of the 1992-2020 funding data, which itself was not representative of the funding distribution, the true effect a project site's overlap might have on EU funding preference is less likely to be known, and this is corroborated by the model's poor fit. Secondly, the funding in highly protected areas is suggested to not be related to the EU's thinking in approving a project. This is likely true as the integrity of these sites can be considered to still impact the broader conservation trends in Natura 2000 by providing for a large space for nature (Bruner et al., 2001; Valente et al, 2022). While this does not necessarily pose a problem to the PFMLG framework, it does suggest a rethinking concerning EU LIFE Nature project location preference.

Limitations

These tests have several very important limitations. First, as has already been mentioned, much of the conclusions around conservation trends have been taken from underlying data which is at once both intentionally incomplete, collected in methodologically distinct ways, and extrapolated based on a series of underlying assumptions which may prove to be false. Furthermore, the conservation data also is presented with a designation problem. That is, sites may be designated around species and habitats which are in relatively poorer shape than non-designated areas. This is less likely a problem as the underlying data is region and Member State wide, and as the sites are meant to protect habitats and species listed as in poor condition to begin with. Lastly, there may be control variables that would have been made for better models but were nonetheless excluded. One such variable might be the year in which a site was designated, potentially affecting the long-term conservation trends, and the likelihood of a site receiving a project. Another could be the conservation status of species and habitats. 2012 conservation status is likely correlated to whether a species is experiencing a conservation trend change in 2013-2018. Moreover, the conservation status could very well predict whether a site is selected or not. The lack of inclusion on conservation status seems to be the biggest limitation to the trend data and raises doubt about the robustness of its conclusions.

Limitations concerning the spatial data are also present. Merging the spatial data, and doing additional spatial transformations required aligning coordinate reference systems. By doing so, true information may have been lost or distorted (Seeger, 2005). Like the conservation data, much of the spatial data is reported by Member States, which may or may not be of the same level of robustness.

Additionally, the focus has been almost entirely restricted to the behavior of authorities, with no real added assumptions about internal political dynamics beyond a strict functional relationship. While this is consistent with the framework developed, and perhaps leaves the door open for more study, it is nonetheless a restricted methodological scope for what the theory could offer or suggest exploring.

Conclusion

Despite the limitations, a few generalizations can be made concerning how the multilevel governance of the LIFE Nature program affects conservation trends. There seems to be a positive correlation between LIFE Nature program projects and long-term conservation trends both in the wider reporting areas of Article 12 and 17, and in the sites themselves. However, the degree of this relationship is highly dependent on the Member States in which the project is implemented. Similarly, while the question of whether a site receives a project appears to be correlated with the reporting area conservation trends in the previous reporting period, this also remains dependent on the Member States.

Among the different types of authorities to coordinate projects, it is the national authority which tends to receive preference from the Commission. In turn, projects led by national authorities seem to be the most capable of affecting conservation trends. Once more, this outcome is heavily dependent on the Member State, suggesting that the ability to affect conservation trends is correlated with the dispersion of authority within Member States and between their constituent Type I nodes.

Between those constituent authorities, the local and the national demonstrate a strong preference for selecting sites with multiple intersecting protected area boundaries. This suggests that it is in these intersecting points that the conservation effects of authority led projects are likely to be felt the most. In effect, this likely means that despite the ‘incompletion’ of the Natura 2000 network, the series of international, national, regional, and local protected areas in the EU create a more complete patchwork network with higher conservation value in areas of overlap generated by authority participation in the LIFE Nature programme.

Lastly, the Commission does not have a preference between the location of projects as such. However, the clear numerical distinction between projects led by non-authorities, and those led by authorities suggest perhaps a simpler way in which the Commission expresses preference for project selection type.

Paired together, results for hypotheses one and five suggest an important role played by authorities in the multilevel governance of the LIFE Nature programme. This suggests that local and national authorities tend to engage in conservation in areas which may lead to the production of conservation ‘hot-spots’, or places of more uniquely favorable conservation status. This relationship would seem stronger among national authorities, as they seem to outperform local authorities in the degree to which their projects affect conservation. The surprising near inverse relationship among regional authorities however gives pause and suggests greater focus ought to be paid in the mechanisms linking project implementation and long-term success to authority type. Additional analyses may focus on the role played by regional and local autonomy or authority.

In the context of PFMLG, this suggests that the multilevel interests associated with Type-1 nodes influence where and how conservation efforts are done within the LIFE Nature framework, and that these conservation efforts generally are associated with positive conservation outcomes. The next round of EEA Article 12 and 17 data will mean the wealth of data in the CINEA Project Portfolio database beginning in 2014 will be compatible with additional testing. This means potentially looking at factors related to the locality of all project participants, and how they affect conservation trends.

In the context of PFMLG, these conclusions neither suggest vindication nor rebuttal. A future project might be interested in exploring the dynamic of party power in specific nodes as an addendum to a similar experiment. In this way, a clearer picture could be parsed out of the actual influence the varying constituent parties of the EU have on its multilevel governance.

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