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Practical Policy Applications of Uncertainty Analysis for National Greenhouse Gas Inventories

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Abstract International policy makers and climate researchers use greenhouse gas emissions inventory estimates in a variety of ways. Because of the varied uses of the inventory data, as well as the high uncertainty surrounding some of the source category estimates, considerable effort has been devoted to understanding the causes and magnitude of uncertainty in national emissions inventories. In this paper, we focus on two aspects of the rationale for quantifying uncertainty: (1) the possible uses of the quantified uncertainty estimates for policy (e.g., as a means of adjusting inventories used to determine compliance with international commitments); and (2) the direct benefits of the process of investigating uncertainties in terms of improving inventory quality. We find that there are particular characteristics that an inventory uncertainty estimate should have if it is to be used for policy purposes: (1) it should be comparable across countries; (2) it should be relatively objective, or at least subject to review and verification; (3) it should not be subject to gaming by countries acting in their own self-interest; (4) it should be administratively feasible to estimate and use; (5) the quality of the uncertainty estimate should be high enough to warrant the additional compliance costs that its use in an adjustment factor may impose on countries; and (6) it should attempt to address all types of inventory uncertainty. Currently, inventory uncertainty estimates for national greenhouse gas inventories do not have these characteristics. For example, the information used to develop quantitative uncertainty estimates for national inventories is often based on expert judgments, which are, by definition, subjective rather than objective, and therefore difficult to review and compare. Further, the practical design of a potential factor to adjust inventory estimates using uncertainty estimates would require policy makers to (1) identify clear environmental goals; (2) define these goals precisely in terms of relationships among important variables (such as emissions estimate, commitment level, or statistical confidence); and (3) develop a quantifiable adjustment mechanism that reflects these environmental goals. We recommend that countries implement an investigation-focused (i.e., qualitative) uncertainty analysis that will (1) provide the type of information necessary to develop more substantive, and potentially useful, quantitative uncertainty estimates-regardless of whether those quantitative estimates are used for policy purposes; and (2) provide information needed to understand the likely causes of uncertainty in inventory data and thereby point to ways to improve inventory quality (i.e., accuracy, transparency, completeness, and consistency).

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1 Introduction

The national greenhouse gas (GHG) emissions inventory is an estimate of a nation's total net emissions from all anthropogenic sources and sinks during the course of one year. Policy makers and climate researchers use this inventory information in myriad ways. For example, national inventory estimates provide a basis for gauging countries' progress toward national emission targets. More specifically, these estimates are used to measure compliance with countries' commitments to reduce emissions under the Kyoto Protocol, which came into full force and effect on 16 February 2005. In the research arena, inventory estimates are one input into global atmospheric and climate models used to project future levels of warming and associated climatic changes. Inventories are also a component of simplified decision-analytic models and integrated assessments that combine several types of models and can be used to evaluate the impacts of alternative policies or emission paths.

In the context of this paper, "uncertainty analysis" refers to the process of identifying and characterizing the causes of uncertainty in a national GHG emissions inventory and quantifying probability distributions for both the data underlying inventory estimates and the actual estimates per se. Like inventories, uncertainty analyses may have policy, analytical, and scientific applications. For example, some policy analysts, concerned about the particularly high uncertainty surrounding emissions estimates for some source and sink categories relative to other categories, have suggested adjusting inventory estimates or emissions trading ratios[1] to reflect the uncertainty in emissions estimates, and so provide a measure of safety for the environment.[2] As another example, Monte Carlo-type analysis, which relies on quantified uncertainty estimates for inventories and other data, is used by the research community to project climate and economic outcomes, and to evaluate the likelihood of outcomes under alternative policy scenarios. Finally, the process of investigating the causes and magnitude of uncertainties in a GHG emissions inventory can also provide information useful for reducing the uncertainty of the inventory itself. This information will also help to target resources to areas where improvement in either the inventory methods or the process of estimating uncertainty will be most constructive.

The characteristics that an uncertainty analysis should possess will depend on the intended use of the analysis. For example, to be adopted by forecasting models that evaluate the climate consequences of alternative policy decisions, uncertainty estimates should be quantitative and should, ideally, exist for all critical uncertain model structures and parameters (or inputs). Omitting key processes could change the probability distributions over the outcomes (Webster et al., 2003). In this circumstance, it may make sense to use expert elicitation to develop subjective probability assessments, in order to develop quantitative uncertainty measures for all key components of the analysis. Moreover, a failure of scientists to quantify uncertainty analysis using all the tools at their disposal may lead to the less desirable result that likelihood is still assessed, but in a less rigorous manner, by other scientists, policy makers, or the general public (Webster et al., 2003). The more closely linked to policy decisions the uncertainty analysis is, the more important it will be to quantify all relevant uncertainties.

Another policy use of quantified uncertainty estimates might be as the basis for adjusting national inventories that are used to make compliance determinations in a system of international emissions commitments. In this case, given that inventory adjustments may necessitate considerable expenditures by nations on additional GHG control, policy

[1] A trading ratio specifies the relative value of emission allowances from two different sources. As described subsequently in Section 3.1, the trading ratio is the number of units of emissions from one source that is equivalent to (offset by) one unit of emission allowances purchased from another source. If the ratio is 1, one purchased allowance can be used to increase emissions by one unit. A lower ratio is more protective of the environment (i.e., increases the likelihood that the trade will result in environmental improvement). For example, a ratio of 1:2 requires that two units of allowances (representing emission reductions elsewhere) must be purchased for each additional unit of emissions offset. In this paper, trading ratios can be either greater or less than 1.

[2] The Parties to the United Nations Framework Convention on Climate Change (UNFCCC) are in the process of adopting an inventory adjustment scheme that in some circumstances uses generic uncertainty estimates to adjust national inventory estimates that a UNFCCC expert review team finds to be deficient.

makers should be fairly confident that the uncertainty estimates are not systematically biased or otherwise likely to be far from actual uncertainty. It will also be important that the adjustment factor be applied equitably across all countries, in the sense that countries not be able to manipulate the mechanism in their own interests. These and other characteristics are discussed in the sections below.

In this paper, we will begin to explore some of these issues in the context of practical applications of uncertainty estimates in national GHG inventories. We examine three distinct applications of uncertainty information and quantified uncertainty estimates. The first two applications involve uncertainty estimates as inputs to the policy process. Specifically, we look at two types of adjustment mechanisms: (1) adjustments to emissions inventories for determining compliance with Kyoto-like commitments;[3] and (2) adjustments to emissions trading ratios. The third application involves the process of uncertainty analysis itself as a learning tool for improving the inventory estimates. The discussion of each application has two purposes: (1) to evaluate and define the benefits of uncertainty estimates and the estimation process in the context of the particular application; and (2) to identify the characteristics that an uncertainty analysis should have in order to be most productive in that application.

The results and approach in this paper can be viewed in the context of other papers in this volume. Nahorski, Horabik, and Jonas (2007) look at a compliance approach that uses uncertainty estimates for adjusting emissions inventory estimates in both the base year and the commitment year. Fewer allowances are then allocated to Parties with higher uncertainties. In turn, these adjustments imply that trading ratios will vary depending on the uncertainties of the inventories being trading. As in our paper, they do not assert that the market would naturally make these types of adjustment without additional regulatory changes. Bartoszczuk and Horabik (2007) discusses uncertainties in GHG emission abatement costs at the country level. Monni, Syri, Pipatti, and Savolainen (2007) look at the uncertainties associated with different emissions trading schemes due to the inclusion or exclusion of particular source (or sink) categories with varying assumed levels of uncertainty. Nilsson, Shvidenko, Jonas, and McCallum (2007) and Jonas and Nilsson (2007) advocate the inclusion of all GHG fluxes to the atmosphere, whether or not they are the direct result of anthropogenic activities, with the hope that future verification techniques (e.g., remote sensing) will be able to reduce uncertainties in a comprehensive flux emissions accounting approach.

All of these papers assume that unbiased and credible quantitative uncertainty estimates are available, or will be available in the future. In contrast, we conclude that quantitative uncertainty estimates at the national level currently do not have the necessary characteristics to be used for compliance purposes (e.g., unbiased in that they are comparable across categories and Parties). A similar conclusion was reached by Winiwarter (2007), although he did not discuss the reasons why he reached this conclusion. As in our paper, Rousse and Sévi (2007) make the critical distinction between uncertainties in emissions estimates (i.e., inventory uncertainty) and market uncertainties. We also make the equally critical distinction between inventory uncertainties and regulatory uncertainty.

2 Using Uncertainty Estimates to Adjust Inventories

National emissions inventories are the yardsticks by which progress in reducing national GHG emissions and compliance with international commitments (such as the requirements of the Kyoto Protocol) are measured. However, the uncertainty surrounding emissions inventory estimates is widely accepted to vary substantially by source category, by country, and over time. Uncertainties are particularly high for some source categories, such as nitrous oxide (N_2O) emissions from agricultural activities. As a result of this uncertainty, some analysts are concerned that compliance with commitments, as measured by inventory estimates, does not adequately measure progress toward meeting national commitments and may not adequately protect the environment.

In response to high uncertainty, some analysts and policy makers have proposed quantitatively adjusting

[3] Throughout this paper, unless otherwise specified, the discussion focuses on a potential adjustment approach that relies on national estimates of uncertainty data. At the 2005 COP/MOP 1 in Montreal, the Parties to the Kyoto Protocol agreed on a simplified adjustment process that does not use uncertainty estimates from individual Parties. That process is not the focus of this paper.

a country's overall estimated national emissions inventory (or the emissions estimates for a particular source category) according to the estimated uncertainty.[4] Consider the Kyoto Protocol, in which a country's quantitative emissions commitment in a given compliance period is based on estimated emissions in a base year. In this case, the emissions estimate for a current compliance period could be adjusted upward. The amount of adjustment should, ideally, reflect the uncertainty of the estimate and increase the probability that a country's actual emissions accord with the country's commitment. Alternatively, the emissions estimate for the base year could be adjusted downward to account for uncertainty. Either type of adjustment would increase the extent to which a country would need to reduce its estimated emissions in order to be in compliance.

For purposes of this discussion, we make two assumptions about uncertainty adjustments for compliance purposes: (1) that the approach taken to the adjustment depends on the environmental goal for which it is undertaken; and (2) that the means of adjustment should employ a statistically valid method. In Section 2.1, we explore the implications of two different environmental goals and the two different definitions of an adjustment they would suggest. In Section 2.2, we look at the criteria that the uncertainty adjustment would need to meet in order to be implemented (assuming that it would be politically feasible) and the implications of these criteria for the characteristics of the uncertainty analysis itself. Note that we do not advocate any of the approaches here, or claim to have presented all reasonable approaches. Instead, we propose two approaches as examples of how uncertainty could be used concretely to adjust inventory estimates and explore the characteristics of the uncertainty analysis that would be needed to make the adjustments viable and practical.

[4] The current adjustment approach under the Kyoto Protocol is based on the judgments of an expert review team and default uncertainty estimates (i.e., conservative factors). These default uncertainty estimates are based on expert judgment and *are not specific to a Party's inventory or related to the actual quality of a Party's inventory*. They are instead used as a justification for a conservative (i.e., punitive) adjustment to a Party's inventory estimate. Expert review teams are also given flexibility to apply adjustments and conservative factors. (See FCCC/SBSTA/2003/L.6/Add.3 and FCC/SBSTA/2005/2.) This approach will be revisited later in the paper.

2.1 Two Possible Adjustment Mechanisms

We start from the premise that any adjustments to inventory estimates should be designed to maintain the environmental integrity of any international compliance system of which the adjustments are a part. In the current context, environmental integrity can be broadly interpreted to mean ensuring that actions-including the estimation process for national emissions inventories, the level of emissions commitments, compliance requirements, and any adjustments made or enforcement actions-tend to further, and not erode, the goals of the United Nations Framework Convention on Climate Change (UNFCCC) and Kyoto Protocol in protecting the environment. More narrowly, we might choose to define environmental integrity as follows: we want to be confident that our policies have met our global climate change goals (i.e., that when we say that emissions have fallen globally, we can have confidence in that statement). Put differently, we care about increasing the confidence that we can have in our global emissions and removal estimates and the confidence that we have met our global goals or are in compliance with commitments.[5]

This type of definition is consistent with the views of a number of countries that are Parties to the Kyoto Protocol and that have stressed that maintaining environmental integrity requires a conservative approach.[6] In turn, they offer a number of different interpretations of adopting a conservative approach (e.g., that commitment period emissions estimates should be conservatively high rather than too low and that estimates and any adjustments should overestimate rather than underestimate emissions or that the emission baseline estimate should be conservatively low).[7] By extension, another interpretation could be that estimated reductions should be conservatively low rather than too high.

[5] Additional discussion of potential adjustments, particularly under a trading regime, can be found in Cohen, Sussman, and Jayaraman (1998).

[6] See, for example, submissions from Australia, Canada, China, New Zealand, Portugal, and the United States to the UNFCCC (2000).

[7] For ease of exposition, in this paper, we refer sometimes to commitment years and sometimes to commitment periods. The analysis is appropriate for either, but is easier to conceptualize in terms of years. The Kyoto Protocol uses commitment periods, which are summed over 5 years.

To develop an adjustment scheme, we must, however, develop a more specific definition of environmental integrity. It is reasonable to begin the discussion with the targets set by the Kyoto Protocol (Annex B) for each participating developed country for the first commitment period. Suppose we start by defining as our goal that we want to be confident that, when countries report emissions inventories that nominally are in agreement with their commitments under the Protocol, the countries truly are, if not in compliance, at least within a given tolerance of complying with their commitments. Thus, we might consider an adjustment based on uncertainty as described in Definition 1.

Definition 1 Compliance with Emission Targets: Attain a reasonable level of confidence that countries have actually achieved the target emissions levels stated in their commitments under the Kyoto Protocol and are in compliance.

To implement this definition, we ask three questions: (1) Would we consider it acceptable if actual emissions *exceeded* the target emissions commitment by some fractional or percentage amount? (2) How much is that amount? (3) How confident do we want to be in our result? If we assume that we know the magnitude of uncertainty surrounding the inventory estimate (an assumption we revisit later in the paper), this definition suggests that emissions inventory estimates would be adjusted upward to take into account the uncertainty of the estimate. In particular, the assumption would be that we want to ensure that, given a reasonable level of confidence, actual emissions do not exceed estimated emissions by more than a specified amount.[8]

Table 1 illustrates the types of adjustment that this definition might imply, based on various quantified levels of uncertainty in an inventory estimate, on the amount of confidence we want to have in our results, and on the percentage amount by which actual emissions could exceed the emissions commitment (i.e., the target level of emissions) before we were uncomfortable with the result.[9] For example, if emissions estimates are 50% uncertain and we want to be 90% certain we have not exceeded our emission target by more than 10%, we would adjust the emissions inventory estimate upward by 20%. The adjusted emissions value would then be compared against the target emissions value to determine compliance. This adjustment provides a margin of safety; that is, a country would effectively need to reduce emissions by more than its commitment in the Kyoto Protocol to remain in compliance with commitments. The higher the level of uncertainty surrounding the emissions inventory estimate, the greater the adjustment that would be required. Similarly, the greater the degree of confidence we require, the greater the adjustment.

The analyses in Table 1 and later in this paper make the simple assumption that the uncertainty distributions are normal and symmetric about the inventory estimate (i.e., there is no bias). In theory, a normal distribution cannot be exactly correct, because negative emissions values are impossible, but this error will be negligible if the probability of a negative value is sufficiently small. For GHG emissions inventories, normal, log-normal, uniform, triangular, and beta distributions have been used to model uncertainty distributions, often truncated to force the values to be within a plausible range. While we could carry out exactly the same analyses for other choices of uncertainty distributions, the normal distribution is a sufficient choice to illustrate our conclusions. In principle, by using a Monte Carlo simulation, all of the numerical approaches described in this paper could be applied to any given uncertainty distributions for a national GHG inventory.

[8] Throughout this discussion, we assume that probability distributions for estimated emissions or emission reductions are normal and that the shape of the probability distribution of emissions for each country or source does not change significantly as emissions are reduced. This entire analysis also ignores the possibility that we might *underestimate* actual emission reductions (i.e., this analysis assumes that the purpose of investigating uncertainty is to ensure that we do not *overestimate* actual emission reductions).

[9] Given the uncertainty ($u\%$) range (assumed to be the end points of a 95% confidence interval) around estimated emissions (E), and assuming a normal distribution, the standard deviation of the distribution equals approximately $u\%\ E/$ (1.96). If we are willing to accept that our emissions could be up to $p\%$ higher than the nominal emissions commitment, then the probability that the actual value lies *below* an upper bound of $(100+p)\%\ E$ can be calculated from the table for a normal error integral found in standard statistics textbooks or using standard statistical software (including Excel). See, for example, Appendix A in Taylor (1997).

Table 1 Ratio of adjusted emissions to estimated emissions

	Uncertainty of emissions inventory		
Confidence[a]	20%	50%	80%
95%	1.06	1.30	1.52
90%	1.03	1.20	1.39
85%	1.01	1.15	1.30
80%	n/a	1.10	1.22

[a] Confidence that actual emissions will not exceed emissions estimate by more than 10%.

Sources: Sussman, 1998, and Sussman, Cohen, and Jayaraman, 1998

The definition of environmental integrity proposed above focuses on only one aspect of emissions uncertainty: the uncertainty of current-year emissions estimates as they are reported for compliance purposes. However, the emissions estimate for the base year – from which the commitment level for a country is calculated under the Kyoto Protocol – is subject to uncertainty that is likely to be of similar or greater magnitude than the uncertainty of the emissions estimate for a commitment period.[10] The uncertainty in a country's base-year emissions or removal[11] estimates may be greater than that during the commitment period because countries will, hopefully, have made improvements in their inventory over time, some of which cannot be fully implemented by recalculating the base – year estimate.

We can broaden the definition of environmental integrity to take into account the influence of uncertainty in both the base year and the current inventory year. Focusing on emission reductions, rather than on emissions, is one way of accomplishing this. In particular, we can argue that it is more important to ask whether or not we have reduced emissions (and, in the case of the Kyoto Protocol, achieved the emission reductions to which countries have committed) than to ask whether emissions are actually what we think they are. Moreover, as the uncertainty surrounding the level of emissions is not identical to the uncertainty surrounding the absolute (or relative) level of emission reductions, we can develop a second definition.

Suppose that a country has agreed to reduce emissions to a target level in a given year (or set of years). If estimated emissions in that time period equal the target level, how confident can we be that emissions have actually been reduced by an amount equal to the difference between base-year emissions and estimated emissions in the target period? Put another way, how confident can we be that estimated emission reductions are not smaller than we think they are or, at least, that they are not "off" by more than a certain amount. Following this line of reasoning, we might choose to define environmental integrity along the lines of Definition 2.

Definition 2 Achieving Emission Reductions: Achieve a reasonable level of confidence that countries have actually achieved the emission reductions, measured relative to base-year emissions, stated in their commitments under the Kyoto Protocol and are in compliance.

To implement this definition, we need to ask, (1) Would we consider it acceptable if actual emission reductions were to fall *below* the committed level of reductions by some fractional or percentage amount? (2) How much is that amount? (3) How confident do we want to be in our result? If we assume that we know the magnitude of uncertainty surrounding the estimated emission *reductions*, this definition suggests that estimated emission reductions would be adjusted downward to take into account the uncertainty of the estimate. However, the result can be compared more easily with the results in Table 1 if we ask how the *emissions estimate* for the commitment period would have to be adjusted upward to ensure that, given a reasonable level of confidence, actual emission reductions do not fall below estimated reductions by more than a specified amount (which could be zero). Again, the conclusion is that emissions estimates would be more heavily increased for more uncertain inventories.

We can construct Table 2 in a manner analogous to Table 1, but this time beginning by looking at

[10] Uncertainty may also differ (and in fact may be lower) in the base year because of policy and political changes over time, including the effects of economic reforms. These changes can affect the definition of what types of sources and sinks are included in the emissions estimate.

[11] A reviewer pointed out that removals are not normally accounted for in the base year under the Kyoto Protocol, except for some 3.4 activities.

uncertainty in emission reductions.[12] Our goal is to provide a level of confidence that our emission reductions have actually been achieved. Given that goal, we can ask what adjustment should be made to the nominal emissions inventory for the commitment period in order to compensate for the uncertainty of emission *reductions*. Suppose that emissions in a commitment year must be 7% below emissions in the base year for compliance (a number that translates into a target absolute quantity of emission reductions). Then, if quantified emission reductions are 50% uncertain and we want to be 90% confident that we have achieved at least 90% of the target quantity of emission reductions, the emissions inventory estimate should be adjusted upward by 3%. The adjusted emissions estimate is then compared with the target level to determine compliance.[13]

The two approaches have some similarities. Both approaches focus on ensuring a reasonable level of confidence with which we achieve externally defined goals; that is, quantified emissions or emission reductions for a target year or period, such as the first commitment period under the Kyoto Protocol. By adjusting emissions estimates to account for uncertainty, both approaches provide a concrete incentive for countries to reduce estimated emissions below nominal emission requirements. Thus, both approaches increase the confidence that we can have in our global emissions estimates, by adjusting the estimated emissions to account for uncertainty. They also provide an incentive for countries to reduce the uncertainty of their emissions estimates over time, in order to reduce the magnitude of the adjustment and so move estimated emissions closer to the nominal commitment level.

Which approach is more stringent? Assuming for the moment that the uncertainty of the emissions estimate

Table 2 Ratio of adjusted emissions to estimated emissions

	Uncertainty of emission reductions[a]		
Confidence[b]	20%	50%	80%
95%	1.01	1.04	1.15
90%	1.00	1.03	1.08
85%	1.00	1.02	1.04

[a] Emission reductions for compliance assumed to be 7% below baseline level.

[b] Confidence that actual emission reductions equal at least 90% of estimated reductions.

Source: Sussman et al., 1998

(in Table 1) is the same as the uncertainty of the emission reductions (in Table 2), then the fractional adjustments are much larger under Definition 1 (in Table 1) than under Definition 2 (Table 2), because in the former case the definition focuses on the absolute level of emissions, which is a much larger number than the absolute reduction in emissions (the focus of the latter definition). Whether or not this is a legitimate assumption, however, and the relationship between uncertainties in emissions relative to uncertainties in emission reductions, are not addressed here.

These are only two of many different possible environmental goals and associated statistical adjustments that could be performed. Other environmental goals could be employed that would result in larger (or smaller) adjustment factors. For example, if our environmental goal were to have confidence in the environmental impact of meeting the target commitments, we would want to apply the adjustment factor to the base-year estimate on which the target commitments are based, and so indirectly adjust the actual target commitment (downward, in this case, to reflect uncertainty). To the extent that inventory uncertainty is likely to decline over time as inventory methods improve, this type of adjustment may make sense. We might, therefore, also want to look at the uncertainty in trends and determine if adjustments to actual emissions estimates are also warranted, or if the adjustment to the base year is sufficient to meet our environmental goals.[14] Another alternative approach might be to

[12] It may not be immediately obvious how to calculate the uncertainty of emission reductions, as it will depend not only on uncertainty in the base and current year, but also on correlations between the two uncertainty estimates, since the factors that produce bias in one year may produce bias in another year. Winiwarter and Rypdal (2001) have looked at trend uncertainties for the Austrian inventory.

[13] Constructing Table 2 requires two steps: (1) making necessary assumptions (e.g., about the uncertainty of emission reductions and the required level of confidence) and calculating the necessary adjustment in emission reductions to provide that level of confidence, and (2) translating the adjustment to emission reductions into an adjustment to the emissions estimate.

[14] The UNFCCC approach uses adjustments to both the base and current year. Again, these adjustments are primarily designed to encourage the use of good practice inventory methods (while also providing some environmental benefit) and are not related to the uncertainty of the overall inventory or of a specific source category for a particular country.

focus on the commitment level rather than the inventory estimate (i.e., how country emission targets would need to be adjusted *downward* in order to ensure that we are confident that we are meeting the current emissions limits specified in the Kyoto Protocol).

2.2 Characteristics of the Adjustment Factor and Implications for the Uncertainty Analysis

The approaches described above result in potentially large adjustments to the emissions inventory. Given the political debate that raged over the targets for commitments under the Kyoto Protocol – with the average across all countries for the first commitment period finally settling at around 5% below base-year emissions-additional reductions of even 1% could have serious political ramifications.[15] An adjustment factor could also have significant associated control cost implications for countries that face additional reductions. Thus, for an adjustment factor to warrant the additional economic costs, we suggest that the factor should possess the following characteristics (many of which are the same characteristics that the national inventory should possess):

- It should meet clear environmental goals and be statistically justifiable given those goals (as described in Section 2.1).
- It should be applied fairly and objectively across countries and source categories (i.e., the method for calculating the factor should rely on data that can be reviewed and verified).
- It should be comparable across countries (i.e., not be subject to inherent variability based on unexplained or unexamined differences in methodology, expert judgment, or expenditures).
- It should be administratively feasible and not burdensome, so that it is practical to calculate and apply the factor.
- It should not be easily manipulated by countries acting in their own self-interest.
- It should not influence market values in a way that (unintentionally) impedes allowance trades between countries.

In large part, the answer to whether or not the adjustment factor can meet these criteria will depend on the characteristics of the uncertainty analysis. In the context of adjustment factors, the uncertainty estimates for the GHG emissions inventory will face several challenges.

Box 1: Sources of Uncertainty in GHG Inventories

Uncertainties associated with GHG inventories can be broadly categorized into *scientific* uncertainty and *estimation* uncertainty. Scientific uncertainty arises when the science of the actual emission and/or removal process is not completely understood. For example, the process of indirect N_2O emissions associated with nitrogen-containing compounds that are first emitted to the atmosphere and then deposited on soils involves significant scientific uncertainty.

Estimation uncertainty arises any time GHG emissions are quantified. Therefore, all emission or removal estimates are associated with estimation uncertainty. Estimation uncertainty can be further classified into two types: *model* uncertainty and *parameter* uncertainty. Model uncertainty refers to the uncertainty associated with the mathematical equations (i.e., models) used to characterize the relationships between various parameters and emission processes. For example, model uncertainty may arise either from the use of an incorrect mathematical model or from the use of an inappropriate input in the model.

Parameter uncertainty refers to the uncertainty associated with quantifying the parameters used as inputs (e.g., activity data and emission factors) to estimation models. Parameter uncertainties can be evaluated through statistical analysis, determinations of the precision of measurement equipment, and expert judgment. Quantifying parameter uncertainties and then estimating source category uncertainties based on these parameter uncertainties is typically the primary focus of most national inventory agencies.

Box 2: Inventory Uncertainty versus Regulatory and Market Uncertainties

Inventory uncertainty relates to the uncertainties in the quantified emissions (or removals) reported in GHG inventories (see Box 1). In contrast, regulatory uncertainties relate to uncertainties in how current or future regulatory rules will affect compliance determinations, and market uncertainties relate to the uncertainties in future allowance prices, mitigation costs, and transaction costs. Both regulatory and market uncertainties are largely independent of inventory uncertainties.

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[15] In addition to general political considerations and the feasibility of negotiating an international system of adjustments that would require reductions beyond those already agreed to (as in the Kyoto Protocol), such a system could raise equity concerns if poorer nations were also those with greater uncertainty, especially if this were primarily due to the source composition of their inventory. In particular, nations with inventories that have a large component of non-energy sources will tend to have greater uncertainties that would be relatively expensive to reduce.

For example, the rules for an emissions trading scheme specify the methodologies that are acceptable for estimating emissions. Emission allowances of a quantity equivalent to these emissions must then be surrendered for compliance purposes. Markets only respond to uncertainty in the value of the traded item-whether it is what it says it is. Thus, an allowance will be worth the price of one ton of emissions, if the rules of the trading scheme say it is worth 1 t of emissions. There will be no regulatory uncertainty about its value on the market if the rules are clear, regardless of the uncertainty in the emissions inventory estimate. If, however, there is uncertainty about the rules that define the quantity of emissions for which an allowance must be surrendered, then regulatory uncertainty will affect the value of an allowance in the market.

Regulatory and market uncertainties have enormous impacts on emissions trading markets. However, these markets are relatively ambivalent about inventory uncertainties, unless they are perceived to have an impact on emissions trading rules (e.g., they are the basis of emissions trading ratios). Policy makers and the public, however, may show concern about inventory uncertainties if they perceive them to be high enough to cause the compliance process to lack credibility or environmental efficacy. Thus the rules of an emissions trading scheme are the conduit though which inventory uncertainties can affect regulatory and market uncertainty.

The subjectivity of uncertainty estimates First, for some source categories, the uncertainty estimates possess a strong subjective component. The inventory is subject to several types of uncertainty (see Box 1 for a discussion of inventory uncertainty and Box 2 for the distinction between inventory uncertainty and regulatory and market uncertainties). Of these, scientific uncertainty and model uncertainty are particularly difficult to quantify, because they must rely heavily on expert judgment regarding inherent uncertainties and potential biases in the estimation methodology. Moreover, expert judgment will be a significant and unavoidable component of uncertainty estimates for national inventories, since the measurements and sample data needed to produce probability distributions will rarely exist for the emission factors and activity data used to produce GHG emissions inventories.

While for some scientific exercises it is possible to collect rigorous statistical data that can be used to estimate the statistical uncertainty of a parameter,[16] it is often impossible to collect similar sample data for many of the national statistics used in inventories. Often only a single data point will be available for a parameter (e.g., tons of coal purchased). It is not meaningful to repeatedly collect independent sets of national statistics for the same year. Instead, we are often given a single emission value or activity factor that supposedly is a census of the entire population rather than a statistical sample, and so is unrepeatable. Our uncertainty estimate in this case represents an assessment by one or more experts of the probabilities that the estimate differs from the true value by "*x*," partly based on the experts' general experiences of similar estimation problems and inventory data and partly based on the experts' understanding about the country-specific inventory, such as possible double- or undercounting of emissions.

The subjectivity of the estimates for some source categories (and, hence, for the inventory overall) has several important consequences. Because of the subjectivity of inputs to the uncertainty estimation process and the reliance on expert judgment, it will be difficult and time-consuming to prepare a detailed uncertainty estimate that is totally transparent and reproducible, and that thoroughly documents all the expert judgment necessary to produce a comprehensive analysis. The analysis, therefore, will not be easily verified by the international community. The difficulty of verifying uncertainty estimates also raises the potential problem that countries may manipulate the uncertainty estimates for their inventories to their own advantage.

Moreover, because expert judgment will vary with the expert and according to his or her familiarity with the inventory data, it will vary from country to country, and even among source (or sink) categories within a country. Therefore, uncertainty estimates will not be comparable across countries, raising the question of whether the adjustment mechanism is an equitable one. Reliance on experts can produce considerable variability in the uncertainty estimates across countries using different experts. Rypdal and Winiwarter (2000) report that, for N_2O, uncertainty estimates vary dramatically – by two orders of magnitude – across existing country estimates. While differences in data and methods account for a portion of the difference, a large part of the difference is attributable to differences in the subjective assessments provided by expert judgment (Morgan & Henrion, 1990).

[16] Statistical uncertainty results from natural variations (e.g., random human errors in the measurement process and fluctuations in measurement equipment). Statistical uncertainty can be detected through repeated experiments or sampling of data.

Of particular concern are cases where the expert's uncertainty estimate is high. In these cases, the available information and data are likely to be extremely limited, and therefore an expert may not be able to quantify the uncertainty much beyond an assertion that the estimate of the parameter is unreliable. For example, an estimated uncertainty of 80% by expert judgment might mean the same as an estimate of 150% or more. (An estimate of 80% by one expert might be the same as one of 150% by another.) In practice, applications of uncertainty analyses should probably be limited to cases where the uncertainty is reasonably low (e.g., less than 80%), or where expert judgment plays a small role.

Systematic bias in uncertainty estimates Second, for some source and sink categories, systematic biases[17] may be the primary cause of uncertainty, especially for activity data (e.g., underreporting by companies or black market activities).[18] Therefore, countries will usually have to rely on expert judgment for the majority of their parameter uncertainty estimates.[19] Even with the most rigorous expert elicitation protocol, it is difficult to obtain judgments in a comparable (i.e., unbiased) and consistent manner across parameters, source categories, countries, and inventory reporting years. Some experts will inherently tend to be optimistic about the quality of data, and others will tend to be pessimistic.[20] Thus, there may not only be a wide uncertainty band around the mean estimate of uncertainty, but the mean estimate itself may be inaccurate (i.e., subject to bias).

Availability of uncertainty estimates Finally, as most countries have not, thus far, undertaken detailed and rigorous uncertainty analyses, reliable estimates of inventory uncertainty are not generally available. An adjustment factor based on country-level uncertainty estimates would require considerable additional resource expenditures for each country that is party to the Kyoto Protocol, as well as considerable resources expended in verifying the estimates internationally. The additional expenditure would be much less, however, than has been expended in producing the inventory itself, at least once the basic structure of the analysis has been developed and implemented. Setting up these initial structures could, however, be time-consuming as well as resource intensive, which would certainly delay trading between countries and impede compliance activities (since countries do not know their actual inventories until they have calculated the adjustment factors). Such a situation could also increase the potential for disputes between an expert review team and a Party because of the subjective elements of the uncertainty analysis.

The Kyoto Protocol process The adjustment process that is under development under the Kyoto Protocol avoids some of these problems. While essentially punitive in nature (i.e., designed to encourage countries to follow "good practice"), it also acts to produce environmental benefits (i.e., the adjustment factor works to increase current-year emissions estimates or reduce base-year emissions). The Kyoto

[17] Systematic parameter uncertainty occurs if data are systematically biased. In other words, the average of the measured or estimated value is always less or greater than the true value. Biases arise, for example, because emission factors are constructed from non-representative samples, not all relevant source activities or categories have been identified, or incorrect or incomplete estimation methods or faulty measurement equipment have been used. Because the true value is unknown, such systematic biases cannot be detected through repeated experiments and, therefore, cannot be quantified through statistical analysis. However, it is possible to identify biases and, sometimes, quantify them through data quality investigations and expert judgments.

[18] There are cases where cause and direction of a specific systematic bias may be known for a national statistical dataset, but for reasons of resource and time limitations or political constraints they cannot be quantified or corrected for in the official national statistics. Therefore, arguing that known systematic biases can be corrected for ignores the real complexities of collecting national statistical data.

[19] The role of expert judgment can be twofold: First, expert judgment can be the source of the data that are necessary to estimate the parameter. Second, expert judgment can help (in combination with data quality investigations) to identify, explain, and quantify both statistical and systematic uncertainties. It is also important to recognize that it is difficult for experts to distinguish between statistical uncertainty and systematic biases. Therefore, elicited estimates of uncertainty tend to incorporate both.

[20] For example, in the United States an early estimate of the uncertainty in methane emissions from manure management based on expert judgment was ±15%. The following year, improvements were made to the methodology to account for more regional differences and corrections were made to some activity data. The resulting change in the overall emissions estimate was 60%.

Protocol process uses an approach similar to that of Definition 1 and Table 1 in this paper, but with different parameters. For emissions commitments, rather than applying the adjustment to all inventories, they apply the adjustment only in cases where an expert review panel finds enough problems with a country's inventory estimate to justify an adjustment to the inventory value. In such cases, after replacing a Party's estimate for an individual source or sink category with one from a review team (representing the central tendency, such as a mean or median), the review team makes a further, punitive adjustment to account for uncertainty. The adjustment uses the uncertainty estimates from the IPCC Guidelines, rather than country-specific uncertainty distributions.

Instead of the 10% leeway illustrated in this paper, the Kyoto Protocol approach allows 0% leeway – no leeway – so that the confidence level equals the probability of not exceeding the target. For our approach, with an illustrative 10% leeway, the confidence level equals the probability of not exceeding the target plus 10%. In contrast to illustrative 80–95% confidence levels used in this paper, the UNFCCC approach prescribes a 75% confidence level for upward adjustments of the commitment period emissions estimates (and 25% for downwardly adjusting base-year emissions estimates). Finally, the Kyoto Protocol approach assumes that emissions are log-normally distributed; the calculations in this paper assume normal distributions.

The Kyoto Protocol process avoids several of the problems of the country-specific adjustment factor described above. In particular, because adjustment factors are uniform across countries (if applied), the process avoids some of the issues of comparability, subjectivity, and gaming that could occur. The process also involves lower administrative costs, because fewer resources are expended on calculating country-specific uncertainty and the adjustment is only applied in select cases. The approach can be implemented more rapidly, so that countries will know more quickly what their inventories for a given year are. The environmental improvements of such a system are relatively low, however, since adjustments are only applied in specific, limited circumstances, and the process for deciding when adjustments are needed is itself extremely subjective and potentially political. Further, because the process is not designed with a clear and stated environmental goal in mind, and because it does not use country-specific factors, it is unclear whether it is the most cost-effective means of obtaining environmental improvement.

Implications for adjustment factors in practice Where does this discussion leave us? Clearly, some system of adjustment factors would provide environmental improvement and increase our confidence that emission targets were being met. A country-specific set of adjustment factors that is applied across all countries would provide more confidence that targets were being met and would be statistically justifiable. Such a system would require a new and rigorous international system for reviewing and officially certifying uncertainty estimates. An adjustment factor applied to all countries, regardless of whether it is country specific or uniform, could also result in the largest environmental improvements. The choice between country-specific adjustment factors, which can be difficult to administer fairly, and uniform adjustment factors, which fail to reflect differences across national inventories, depends both on the environmental improvements each offers (i.e., how well the factors meet environmental goals or stated policy goals) and on the strengths and drawbacks of each approach.

3 Adjustments to Emissions Trading Ratios Based on the Uncertainty of Emissions

Now that the Kyoto Protocol has entered into force, developed (i.e., "Annex B") countries – excluding the United States and Australia, which have not ratified the Protocol – are legally committed to reduce their GHG emissions to specific negotiated target levels during the first 5-year commitment period (2008 through 2012).

In addition to meeting their commitments by reducing domestic emissions, Annex B countries can engage in three alternative market mechanisms that allow Parties to the Kyoto Protocol to purchase emission reductions from other Parties. The three mechanisms are (1) emissions trading, which permits buying and selling emission allowances among Annex B countries; (2) the Clean Development Mechanism (CDM), under which developed countries can undertake emission reduction projects in developing countries and use the emission reductions to offset their commitments; and (3) Joint Implementation (JI), which allows for project-related emission reductions within Annex B countries (e.g., mitigation projects in EIT[21] countries can produce emission reductions for developed nations). While the focus of this section is on international emissions trading, there are potential lessons for sales of project-related emission reductions under the CDM and JI.

In an emissions trading system, an administering authority generally sets quantified limits (referred to as rights, obligations, or permits) on the emissions of participants in the system. Participants can then transfer these rights, obligations, or permits from one participant to another (generally by buying and selling), subject to any restrictions set by the administering authority. Emissions trading systems are frequently referred to as "tradable allowance" systems, because participants must hold emission allowances, which give the owner the right to emit a specified physical unit of emissions, in sufficient quantity to cover actual emissions. Many consider emissions trading systems to be an attractive alternative to fixed emission limits because, in appropriate circumstances, they can reduce the overall cost of achieving an environmental goal. In a trading system, participants have flexibility in how they meet their obligations: they may choose either to take actions to reduce emissions or to purchase additional permits (if it is cheaper to do so). Thus, participants with lower costs of reducing emissions can undertake additional reductions and sell excess allowances to entities for whom the cost of reducing emissions is higher.

Under the Kyoto Protocol, Annex B countries can engage in emissions trading. The quantified limit for a country is its assigned amount (AA), and the instrument that is traded between countries is an assigned amount unit (AAU). As with other emissions trading programs, much of the debate in international GHG trading has centered on the impacts of trading on mitigation costs and the cost-effectiveness of emission reductions; on issues of the equitable division of responsibility for emission reductions; and on the design of domestic, facility-level emissions trading programs to support international trading.

Another issue in GHG emissions trading is the uncertainty of emissions data on which trades are based. Some analysts have suggested using trading ratios that are adjusted to reflect uncertainty, or even excluding highly uncertain emission sources from trading altogether, on the grounds of potential harm to the environment. Arguments have been made along these lines not only for the allowances traded between Parties (AAUs), but also for the instruments utilized by the project-based mechanisms of the Kyoto Protocol, the CDM (in which the tradable instrument is a certified emission reduction, [CER]) or JI (in which the tradable instrument is an emission reduction unit [ERU]).

The argument commonly made for prohibiting, or at the least placing a lower value on, emissions trades involving emissions from source categories for which the emissions estimates are highly uncertain is based primarily on the environmental harm that can be caused. If the uncertainty of the emissions estimate is high-or poorly understood-for some source or sink category, then emissions between certain and uncertain sources should not be traded (i.e., bought and sold) on an equal basis. For example, if society allows increased emissions from a source category with very low uncertainty in its emissions estimate to be offset by an equal quantity of emission reductions from a source category for which the emissions estimate is highly uncertain, we may not be sure that we have actually reduced emissions. Thus, the argument goes, any emission reductions or excess emission allowances from uncertain sources should be sold more cheaply (i.e., be worth less) than emission allowances or reductions from more certain sources. The trading ratio between allowances for certain and uncertain sources, therefore, is essentially less than 1: a given quantity of uncertain allowances will be equivalent to fewer certain allowances.

This approach of adjusting trading ratios to account for uncertainty has generally been adopted by watershed nutrient credit trading programs in the United States (King & Kuch, 2003). In these programs, nutrient discharges by diverse sources do not trade on an equal pound-for-pound basis. Rather, the trading ratio is based on the expected "risk-adjusted" outcome of trades; that is, on the certainty that a trade will actually result in decreased nutrient discharges and improved water quality.

[21] Economy in transition (EIT) is a term used under the UNFCCC to refer to the countries of the former Soviet Union and related East European satellite nations that are now undergoing a transition to a market-based economic system.

For example, the impact on nutrient levels, in the receiving water body, of changes in end-of-pipe nutrient discharges by a point source, such as a wastewater treatment facility, will be relatively certain. However, the impact on water quality of changes in land-management practices by non-point sources, such as farms, will be far less certain.[22] Thus, a wastewater treatment facility seeking to offset existing levels of nitrogen discharge may need to buy three or four pounds of non-point discharge reduction credits to offset each pound of nitrogen they are allowed to discharge, implying a trading ratio of 1:3 or 1:4. Note that, in this type of nutrient trading program, there is no accepted trading ratio for point and non-point trades. Each trade must be evaluated on its individual merits and approved by the regulatory authority, a process that can increase administrative costs (for both the traders and the administering authority) and the time required to finalize a trade (King & Kuch, 2003).

In this section we examine two alternative approaches to defining GHG trading ratios to reflect uncertainty in emissions inventories and maintain the environmental integrity of trades. As in Section 2.1, we start from the premise that any adjustments to the trading ratios should be designed so that allowance trading does not diminish environmental quality. There are at least two possible ways to interpret this:

- In the first situation, countries have emissions commitments, and a country is found to be in compliance with its commitment if its estimated emissions inventory is less than or equal to its commitment. In this case, the trading ratio is defined so that the upper bound of a confidence interval (say, 95%) around their estimated combined emissions is unchanged by trading, relative to a system of binding commitments (that are met). Thus, we can be confident that the upper bound of the uncertainty band around total combined emissions does not rise as a result of trading. Note that we do not know whether estimated total combined emissions rise or fall.
- In the second situation, countries have emissions commitments, but these have been converted into what are referred to as *targets*; that is, country commitments are adjusted to reflect uncertainty in a manner similar to that in Section 2.1. A country is assumed to be in compliance if its emissions inventory is less than or equal to its *target*. In this case, the trading ratio is determined so that the probability that two countries exceed their aggregate (i.e., combined) emissions *commitment* is the same before and after trading. Thus, we want to be confident that actual combined emissions do not rise as a result of trading. Again, estimated total combined emissions may rise or fall.

Sections 3.1 and 3.2 below address each of these situations in turn. It turns out that, given reasonable assumptions about uncertainty and environmental goals, the intuition behind the nutrient trading program – that uncertain emissions should be less valuable than certain emissions – is not necessarily justifiable from an environmental and statistical perspective. Whether, and how, trading ratios should be adjusted to account for uncertainty depends, in fact, on the characteristics of the uncertainty estimate. In Section 3.3 we look at the characteristics that the uncertainty estimate would need to possess to be viable in these applications, building on the discussion in Section 2.2. We also discuss some additional issues in the practical application of trading ratios.

3.1 Trading Ratios: Upper Bound Emissions are Unchanged

The approach developed below ("upper bound") begins with the idea that we want to be confident that emissions do not rise as a result of trades. The starting point for this approach is the idea that, given an environmental goal, the purpose of both national commitments and the trading system is to ensure – with a reasonable amount of certainty – that this goal is not exceeded. In this case, the assumed "goal" is an upper bound of a probability distribution around mean estimated emissions. For example, the upper bound might be defined as the upper end of a 95% confidence interval around the mean; that is, the

[22] The impact of altered management practices at a farm, for example, will depend on the effectiveness of practices at the farm in reducing "edge of farm" nutrient discharges (which is highly site specific), on weather, on how spatially removed the farm is from an adjacent water body, and on conditions in adjacent receiving water (King & Kuch, 2003).

97.5 percentile value.[23] In this case, we can be very confident that actual emissions will not exceed the upper bound value, given the mean emissions estimate. Thus, one possible approach to designing a trading system is to define trading ratios such that trades *do not change the upper bound*. Thus, trading will not change the likelihood that we achieve our desired environmental goal (measured in terms of actual emissions), even if the mean emissions estimate changes.

Suppose there are two countries, A and B. Without loss of generality, choose emissions units so that Country B anticipates reducing emissions by one unit below its commitment. Country A has committed to reduce emissions to an amount A, and Country B has committed to reduce emissions to an amount B. Thus A and B are the emissions commitments of Countries A and B, respectively, divided by the anticipated additional emissions reduction by Country B. Suppose we have good information on the percentage (or fractional) uncertainty (denoted u) range associated with a 95% confidence interval for the emissions estimates for two countries. Thus, if B achieves its goal, its upper bound (97.5th percentile) emissions will equal $(1+u_B)B$. Similarly, if A achieves its goal, its upper bound emissions will equal $(1+u_A)A$.[24]

Suppose further that B anticipates reducing emissions below its commitment, and that A anticipates being unable to meet its commitment. The question then becomes, If B anticipates reducing emissions by one unit below its commitment, so that emissions in Country B equal $(B-1)$, by how much could Country A increase its emissions without violating the upper bound constraint? If the amount that Country A could increase its emissions is called x, then x also gives the trading ratio between the two countries; one unit of emission reductions in Country B is worth x units of extra emissions in Country A. Thus, Country A will be willing to pay to B, for each unit B sells, an amount equal to the amount it would cost Country A to reduce emissions by x units.

Assuming approximate normality, the estimate of total emissions represented by the commitments has mean $A+B$ and adjusted standard deviation[25] given by

$$\mathrm{SD} = \sqrt{u_A{}^2A^2 + u_B{}^2B^2}, \tag{1}$$

so that the upper bound for the total emissions represented by the commitments is given by

$$\mathrm{BOUND} = A + B + \mathrm{SD}. \tag{2}$$

The post-trading total for the relevant sectors has mean $A + x + B - 1$ and upper bound

$$\mathrm{PBOUND} = A + x + B - 1 + \sqrt{u_A{}^2(A+x)^2 + u_B{}^2(B-1)^2}. \tag{3}$$

A reasonable argument requires that trading should not change the upper bound (although the mean does change), so that we are just as confident as before of not exceeding the given upper bound. We therefore choose x to be the solution of BOUND = PBOUND. To solve this equation, first write it in the form

$$\begin{aligned} x - 1 &= \mathrm{SD} - \sqrt{u_A{}^2(A+x)^2 + u_B{}^2(B-1)^2} \\ &= \mathrm{SD} - \mathrm{SD2}. \end{aligned} \tag{4}$$

Next, subtract SD from both sides and square the resulting equation to obtain

$$\mathrm{SD2}^2 - \mathrm{SD}^2 = (x-1)^2 - 2(\mathrm{SD})(x-1). \tag{5}$$

[23] A "95% confidence interval" is an interval calculated from observational data such that the interval would be expected to include the unknown true value (e.g., total GHG emissions) for 95% of possible data sets, although we generally will not know whether or not this is true for a given data set. Since emissions inventory estimation often uses non-statistical methods (e.g., expert judgment) and methods not based on observational data, the term 95% confidence interval is here extended to mean any interval that in some sense is assumed to have a 0.95 probability of including the unknown true value. The upper bound is typically assumed to be the 97.5th percentile, and the lower bound the 2.5th percentile, so that the same 2.5% of the values lie above and below the confidence interval.

[24] For simplicity, we assume that the uncertainty (expressed as a percentage) is unchanged for the sector or country by activities that increase or decrease emissions.

[25] Strictly, this equation represents the standard deviation of the sum of emissions from A and B, multiplied by a scalar. The magnitude of the scalar (which may equal 1) depends on the width of the confidence interval for which the uncertainties are calculated and on the shape of the distribution of emissions. The scalar would equal 1.96 for a 95% confidence interval if emissions were normally distributed. It is assumed for this equation that the uncertainties represent the same level of confidence for both A and B.

This gives a quadratic equation for x

$$u_A^2(2Ax + x^2) + u_B^2(-2B + 1) = (x-1)^2 - 2(\mathrm{SD})(x-1), \text{ giving}$$
$$x = \frac{-\beta + \sqrt{\beta^2 - 4\alpha\chi}}{2\alpha}, \text{ where}$$
$$\alpha = u_A^2 - 1,$$
$$\beta = 2Au_A^2 + 2 + 2(\mathrm{SD}),$$
$$\chi = u_B^2(1-2B) - 1 - 2(\mathrm{SD}). \tag{6}$$

If A and B are large (relative to 1, the quantity of emissions to be sold), then the solution for x is approximately

$$x = \frac{\mathrm{SD} + B \cdot u_B^2}{\mathrm{SD} + A \cdot u_A^2}. \tag{7}$$

(This can be shown using Taylor series expansions. Note that x is a dimensionless ratio, as are the values of SD, A, and B, since everything is relative to the additional one unit emissions reduction by Country B.) Thus, unless a country is selling or buying a large portion of its emissions, the simpler equation is a reasonable approximation of the trading ratio. In this equation, x could be bigger or smaller than 1 depending on the relative sizes of the means (A and B) and of the uncertainties (u_A and u_B).

The equation for x above illustrates that the trading ratio that satisfies this approach is not simply a function of the uncertainty of each country's inventory. Rather, the trading ratio depends on (1) the magnitude of estimated emissions in Countries A and B; (2) the absolute uncertainty (i.e., the standard deviation) of total emissions from the two countries; and (3) the relative uncertainties surrounding emissions in Countries A and B.

In particular, consider the numerator of the equation defining x in Eq. 7. All else being equal, x will be higher if the second term, $B\,u_B^2$, is greater. In other words, if any three out of A, B, u_A, u_B, and SD are held fixed, then x increases with the second term, $B\,u_B^2$, because of the bound condition BOUND = PBOUND.

This second term combines the uncertainty of Country B's emissions with the magnitude of its emissions. (Formally, it is proportional to the variance in B's emissions divided by B's mean estimated emissions.) Thus, as this term rises, Country A should (from a global perspective) pay *more* to reduce emissions from B, or, equivalently, emission reductions purchased from B should translate into fewer emissions by Country A. The rationale is that emission reductions in Country B contribute more to reducing the upper bound of the combined emissions from A and B than would the same quantity of emission reductions by Country A. Similarly, as the analogous term for Country A (the second term in the denominator) rises, Country A should pay *less* (from a global perspective) for emission reductions from Country B, because the emission reductions from Country A would do more to reduce the upper bound than would emission reductions from Country B.[26]

A simple example may help clarify how the equation for x of Eq. 7. would work. Suppose that Countries A and B have both committed to emissions of 100 t. The uncertainty in the emissions estimate is 40% for Country A and 50% for Country B.[27] Country B finds that it is cheaper to reduce its emissions than it anticipated, and Country A finds that it is more difficult to meet its commitment than anticipated. Thus, Country A finds that its estimated emissions inventory equals 110 t, and it needs to purchase 10 t of emission reductions (emission allowances) from another country. Country B has estimated emissions of 90 t, and so it has 10 t to sell. Using the above equation, x equals 1.11. Country A then purchases about 9 t of Country B's excess reductions to offset A's excess of 10 t of emissions, and so meet its own commitments. Note that, whenever x is greater than 1, estimated total emissions between the countries will *rise* as a result of the trade. This and other examples are illustrated in Table 3.

The trading ratio formula may seem counter to expectations, because it implies that the emissions with the greatest uncertainty are the *most* valuable to buyers. The intuitive explanation is that if Country A has relatively certain emissions and Country B has relatively uncertain emissions, then A's contribution to the overall upper bound (to $A+B$) is small compared with the reduction in the upper bound caused by a one-unit reduction in B's emissions. Effectively, Country A is given a bonus because each reduction in B's uncertain emissions is being swapped for more certain emissions from A. We value reductions in uncertain sources more highly because such reductions essentially begin to remove the emissions from uncertain source categories

[26] The impact of the SD term depends on the ratio of the uncertainty products.

[27] While this large uncertainty between countries is unlikely for developed countries, it is certainly possible for trades between source categories. Moreover, the large uncertainty serves to illustrate the workings of the trading ratio.

Table 3 Illustrative trading ratios

Country A (buyer)		Country B (seller)		X (trading ratio)
Emissions commitment (t)	Uncertainty (%)	Emissions commitment (t)	Uncertainty (%)	
100	40	50	5	0.72
100	40	50	20	0.76
100	5	50	40	1.37
100	20	50	40	1.12
100	30	50	40	0.98
50	40	100	5	0.73
50	40	100	20	0.89
50	5	100	40	1.39
50	20	100	40	1.32
50	30	100	40	1.24
100	40	100	10	0.74
100	40	100	30	0.89
100	40	100	50	1.11

from the inventory and from the environmental system; for a given emissions estimate, the environment would be better off if those emissions came only from the most certain source categories, because then we would have the best idea of what emissions really look like. This does not (as some suppose) argue for removing uncertain emissions from the *inventory*, but rather places a higher premium on removing more uncertain emissions from the *environment*.

In Section 3.2, we explore a variant of this approach, using a slightly different environmental goal. In Section 3.3 we return to the question of whether this approach makes sense from the perspective of the uncertainty characteristics of the GHG inventory, and discuss some possible implications for nutrient trading as well.

3.2 Trading Ratios: Probabilities of Exceeding Emissions Commitments are Unchanged

An alternative trading ratio can be developed based on limiting the probability of exceeding the emissions commitment (i.e., combining some of the ideas in Sections 2.1 and 3.1). Suppose Countries A and B have emissions commitments under the Kyoto Protocol of E_A and E_B, respectively. Instead of adjusting inventory estimates to reflect uncertainty (as in Section 3.1), each country is assumed to have an emissions target A or B, where the target is derived by adjusting the emissions commitment level to reflect uncertainty. Specifically, the target is determined so that if a country has an estimated emissions inventory that equals the emission target, then the probability is 95% that actual emissions do not exceed the emissions commitment by more than 10% (for that country). As in Section 3.1, we choose the emissions units such that Country A wants to buy one unit of emissions from Country B. Instead of defining trading ratios to preserve the upper bound (as in Section 3.1), this here we define trading ratios to preserve the probability that total estimated emissions from the two countries sum to less than their combined emissions commitments.

Let the two countries have fractional uncertainties u_A and u_B. Assume that the emissions targets are defined so that, at the targets, the probability of not exceeding the emissions commitment by more than 10% is 95%. Assume that emissions are (approximately) normally distributed. For Country A, with estimated emissions meeting the adjusted target (A), the 95% confidence interval for actual emissions is

$$\text{Emissions} = A \pm Au_A = A \pm 1.96 \text{ Std. Dev. (emissions)}. \tag{8}$$

Thus the probability of not exceeding the emissions commitment by more than 10% equals

$$\Phi\left(\frac{E_A(1.1) - A}{Au_A/1.96}\right) = 0.95, \tag{9}$$

where Φ denotes the cumulative distribution function of a standard normal random variable. This equation has the solution

$$1.1\ E_A = A\left(1 + \frac{1.64}{1.96}u_A\right). \tag{10}$$

A similar equation applies for Country B.

Before trading, the probability that the estimated combined emissions for the two countries will not exceed the combined emissions commitment by more than 10% equals

$$\Phi\left(\frac{(E_A + E_B)(1.1) - (A + B)}{\sqrt{A^2{u_A}^2 + B^2{u_B}^2}\Big/1.96}\right). \tag{11}$$

If B sells one unit and A is allowed x units for that trade, then the mean combined emissions after trading will be $A + x + B - 1$, and the standard deviation will be adjusted accordingly. Thus, after trading, the probability that total emissions will not exceed the total emissions commitment by more than 10% equals

$$\Phi\left(\frac{(E_A+E_B)(1.1)-(A+x+B-1)}{\sqrt{(A+x)^2{u_A}^2+(B-1)^2{u_B}^2}\Big/1.96}\right). \qquad (12)$$

The trading ratio is the value of x that makes these two probabilities equal, which is the same as solving the equation

$$\frac{1.1(E_A+E_B)-(A+B)}{\text{SD}} = \frac{1.1(E_A+E_B)-(A+x+B-1)}{\text{SD2}}, \qquad (13)$$

using the notation of the previous section. This gives the equation

$$(x-1)K = \text{SD} - \text{SD2}, \qquad (14)$$

where K is the expression

$$K = \text{SD}/\{1.1(E_A+E_B)-(A+B)\}, \qquad (15)$$

$$K = \frac{1.96\ \text{SD}}{1.64(Au_A+Bu_B)}. \qquad (16)$$

(The second expression for K follows from the above relationship between targets and limits.) Note that K *is not a constant, but instead depends upon the emissions targets and the uncertainties*. It is not difficult to show that K is bounded below by $(1.96/1.64)\big/\sqrt{2}$ and above by (1.96/1.64).

As before, we obtain a quadratic equation for x:

$$u_A^2\left(2Ax+x^2\right)+u_B^2(-2B+1) = K^2(x-1)^2-2K(\text{SD})(x-1),\ \text{giving} \qquad (17)$$

$$x = \frac{-\beta+\sqrt{\beta^2-4\alpha\chi}}{2\alpha},\ \text{where} \qquad (18)$$

$$\alpha = u_A^2 - K^2, \qquad (19)$$

$$\beta = 2Au_A^2 + 2K^2 + 2K(\text{SD}), \qquad (20)$$

$$\chi = u_B^2(1-2B) - K^2 - 2K(\text{SD}). \qquad (21)$$

If A and B are large (relative to 1, the quantity of emissions to be sold), then the solution for x is approximately

$$x = \frac{K(\text{SD})+B\cdot u_B^2}{K(\text{SD})+A\cdot u_A^2}. \qquad (22)$$

This alternative trading ratio is in a form very similar to the ratio developed in Section 3.1 and can be used in exactly the same manner. This trading ratio, which is based on inventories that are adjusted in the manner described in Section 2.1, will be quantitatively different from the ratio in Section 3.1.

3.3 Characteristics of the Trading Ratio and Uncertainty Analysis

The trading ratio should have characteristics similar to those of the adjustment factor described in Section 2: It should meet clear environmental goals and be statistically justifiable given those goals. (Specifically, for a trading ratio, we would expect that allowance trading would not reduce environmental quality, relative to a system of binding commitments without trading.) The trading ratio should be equitably applied and comparable across countries or source and sink categories. Finally, it should be administratively feasible and practical to apply, and it should not be easily manipulated by countries attempting to act in their own self-interest.

For the most part, these criteria suggest the same characteristics for the uncertainty estimate that were identified in Section 2, and the subjectivity and variability of the uncertainty estimate pose similar problems for the operation of a trading ratio as they did for the adjustment factor. The trading ratio, however, is more complex to calculate and administer than the inventory adjustment discussed in Section 2. This section, therefore, addresses in more detail two aspects of the trading ratio: (1) the impacts of trading on environmental quality and (2) a method for simplifying the trading ratio without sacrificing environmental integrity.

3.3.1 Fulfilling Environmental Goals: The Impacts of Trading on Environmental Quality

The discussions in Sections 3.1 and 3.2 define specific (and plausible) environmental goals and then derive trading ratios that are statistically justifiable, given those goals. For simplicity, the discussions assume that trades are being made between two countries and that the uncertainty of the emissions inventory estimate for each country is known. In both cases, the trading ratio depends not only on the relative uncertainty of the emissions inventories in the two countries, but also on a number of factors. These factors, such as the relative magnitude of emissions from the two countries, are important because they influence inventory uncertainty and the uncertainty of total combined emissions estimates.

In both the situations in Section 3.1 and 3.2, however, one important conclusion emerges regarding the trading ratio, which is defined as the number of allowances, or tons of allowable emissions, that the buying country receives in return for purchasing one excess allowance, or ton of emission reductions, from the selling country. All else being equal, the trading ratio, x, will rise as the uncertainty of the tons sold rises. Thus, the price of sold allowances will rise as their uncertainty rises (i.e., they will become more valuable to the purchasing country); hence a given quantity of sold allowances will offset a greater number of emissions in the purchasing country.

The mathematical intuition behind this conclusion is understandable: The higher the uncertainty surrounding a country's emissions estimate, the more those emissions contribute to the overall uncertainty of aggregate (total across all countries) emissions. Thus, reducing those uncertain emissions reduces overall uncertainty more than does reducing emissions from a more certain inventory. Reducing those relatively uncertain emissions also reduces the upper bound, or confidence interval, around aggregate estimated emissions. Consequently, the environment will be better served by eliminating emissions from uncertain sources, as we then will be left with estimated emissions from certain sources and will know with more certainty what actual emissions might be, and may even reduce the upper bound. In addition, whether a trading ratio is greater or less than 1 depends not only on the relative uncertainties of emissions estimates from the two sources, but also on the magnitude of emissions, which affects the extent to which estimation uncertainty contributes to the overall upper bound.

The foregoing intuition is directly opposite to the conventional wisdom regarding how a trading ratio should operate. Moreover, in the nutrient program described earlier, the principles underlying the types of trades that have been allowed by regulators operate so that emission reductions from a source with uncertain emissions are worth *less* in a trade with a source with more certain emissions (point source). Thus, a point source must buy *more* emission reductions from a non-point source – the point source can offset less than one ton of emission increases in return for purchasing 1 t of non-point source emission reductions. As long as the uncertainty of the emissions estimate for the non-point source is greater than the uncertainty of the emissions estimate for the point source, x will be less than 1, and it will be lower as the uncertainty of the non-point source is higher. This approach is also appealing from an intuitive perspective, because it ensures that each trade will likely improve environmental quality; by removing extra "uncertain" emissions from the environment when we increase "certain" emissions, we ensure that the estimated (and actual) emissions have not risen as a result of the trade, that is, we ensure that we really have removed enough emissions to offset the extra emissions elsewhere.

Which of the two views, then, is correct from an environmental (and statistically justifiable) perspective? Are they compatible? Fundamentally, they are not inconsistent with each other and may be appropriate under different circumstances. Which view is appropriate from an environmental perspective depends on two distinct considerations: (1) how much protection is required from an emissions trade (i.e., what our environmental goal is) and (2) the nature of uncertainty and uncertainty estimates on which the decisions are based.

The amount of protection and the environmental goal The first consideration is how much protection is required when an environmental trade is made. In the formulae derived in Section 3.1, the only protection afforded the environment is an assurance that the upper bound on the combined emissions estimate cannot rise; in fact, depending on the nature of the trade, aggregate estimated emissions might rise or fall. In Table 3, total measured emissions from the two countries may rise in cases where x exceeds 1. Similarly, in the formulae derived in Section 3.2, the only protection afforded the environment is an assurance that the probability that the combined emissions estimate exceeds the target commitment cannot rise; again, depending on the nature of the trade, aggregate estimated emissions might rise or fall. Thus, although the upper bound or exceedance probability is unchanged, the combined emission inventories of the two countries (relative to pre-trading emissions or relative to the sum of commitments) may rise. Such trades represent environmentally adequate emission controls, as judged against the goal of maintaining the upper bound or exceedance probability.

In contrast, in the nutrient case, the trade is designed to protect the environment in two additional ways: (1) the ratio of sold-to-bought emission allowances/reductions is greater than 1 (and so emissions are being "retired"), and (2) uncertain emissions are being removed more than proportionately when emissions that are relatively more certain increase. In the interests of making continuous progress in reducing emissions, or to provide an extra measure of protection because our uncertainty estimates may not be entirely correct (as we know from Section 2.2), it may be desirable to restrict trading ratios for GHG emissions to a maximum of 1, or even a lower number, as what is sometimes called an "environmental dividend." This restriction would also provide some of the additional environmental benefits of the nutrient trading situation, although any limit would essentially be ad hoc in nature.

The nature of uncertainty and uncertainty estimates The applicability of the two different approaches – that followed for the nutrient program and that described in Section 3 – depends in part also on the characteristics of the uncertainty estimates for the inventory. The methodology followed in Sections 3.1 and 3.2 relies critically on the assumption that our uncertainty estimate adequately represents the statistical properties of the emissions estimate.

Regardless of whether the uncertainty estimate comes from objective or subjective data, we are relying on our estimate to obey certain statistical properties: that our emissions estimate is the "best estimate" of the value of actual emissions and that all components of uncertainty, whether objectively or subjectively obtained, contribute to our estimate of uncertainty (ISO, 1993). Up to a point, this is reasonable. In some cases, the inventory activity data and emission factors come from physical measurements (such as facility-level emissions data) or surveys (such as the number of cows). In these cases, the raw data have objective statistical properties that can be used to derive similarly objective uncertainty measures for the inventory estimates. In other cases, as discussed in Section 2, objective measures of uncertainty for the activity data and emission factors are unavailable, and so we must rely on subjective assessments; that is, on assumed probability distributions based on experience and expert judgment. While these latter assessments may vary in rigor and quality, there are nonetheless accepted procedures for eliciting subjective assessments from experts, and many scientists accept the legitimacy of these procedures.

However, national GHG inventories are unlike many other estimation processes because, for a number of source categories, little is known about the processes that produce emissions, or about the effects of changes in activity levels on emissions. Thus, the assumption that all sources of uncertainty are included in the assessment is questionable. For some sources, the legitimacy of subjective assessments of uncertainty is also questionable. For example, so little is known about the processes by which emissions are produced (and reduced) for some source categories – such as nitrogen from soils – that inventory estimates are highly suspect, and so uncertainty estimates by experts are equally or more suspect.

For some emission sources, uncertainty estimates may be less in the category of risk and more in the nature of "Knightian uncertainty." While the term uncertainty has come to be used in a general way to represent a situation wherein variables are not known with certainty, Frank Knight, in his seminal 1921 book (Knight, 1921), made a distinction between risk – which describes situations where an explicit probability distribution of outcomes can be calculated-and "true" uncertainty – where the randomness of an uncertain event cannot be adequately described by a probability measure. Thus, risk can be reduced to a single distribution with known parameters, whereas in the case of uncertainty, the information is too imprecise to be summarized by a single probability measure; that is, probabilities are unknown and are impossible to calculate with any confidence because of the uniqueness or specificity of the situation (Kasa, 2000).

The distinction between risk and uncertainty is particularly instructive for national GHG inventories and adjustments such as trading ratios. For some emission sources – in particular, those for which understanding of the processes by which emissions are generated is still evolving – our understanding of the magnitude of emissions is closer to uncertainty than to risk.

If we have confidence in our assessments of uncertainty – and believe that we actually have good uncertainty estimates – then environmental integrity can be served by a trading ratio that is derived in the manner described in Section 3. We may not, however, necessarily believe our uncertainty estimates, or believe that we have captured all sources of uncertainty. If that is the case, then we may be in a situation of true uncertainty; we do not know how good our inventory estimates are, what we have missed, or what the uncertainty is surrounding the inventory estimate. In this case, the best policy alternative may be to follow a precautionary approach: to protect the environment more than our statistical assessments might suggest is necessary. We might, therefore, choose to cap the trading ratio at 1, or even a lower number. An economics paper looking at labor markets found a similar result: "...when people lose confidence in their forecast about what happens in the future, they generally prefer certainty to uncertainty" (Nishimura & Ozaki, 2001). Where emissions trading is concerned, we may prefer the devil we know to the devil we don't know.

3.3.2 Administrative Complexity

The administrative requirements of the system affect costs to participants and administrators of the system. Participants in the system face the costs of finding and completing trades (sometimes referred to as transaction costs) and complying with the administrative requirements of the system. The administering authority faces the costs of review and verification (of emissions inventory and uncertainty estimates) and making compliance determinations. Both sets of parties face the costs of tracking emissions, allowances, and allowance trades. In the context of trading ratios, several aspects of the adjustment mechanism will be key to tractability. In particular, the method of calculating trading ratios should involve clearly defined formulae or relationships, so that all countries can understand and easily implement the system. The trading ratio should also be relatively stable; that is, it should not change so frequently that trading parties cannot anticipate what the ratio will be when planning how to reach compliance.

A bilateral trading system, such as is described in Sections 3.1 and 3.2, could be relatively difficult to administer and to participate in. The system could be simplified, however, by setting up a clearinghouse to which countries sell, and from which countries purchase, emissions. The administrative complexity of the system would be greatly reduced if, for example, the clearinghouse could be designed in such a way that the formula determining the value of emission reductions purchased was based on the buyer's inventory, without reference to the origin of the reduction (i.e., the seller). Such a clearinghouse would greatly facilitate trades among diverse sources, which otherwise would require the calculation of individual trading ratios for each separate combination of possible sources or sub-sources that are traded between or within countries.

One way to set up the clearinghouse might be to consider the process of trading between two countries as a two-stage process. In the first stage, Country B sells one unit of emissions to the group of all other countries except for A and B. The emissions and associated uncertainty for the group of countries are easily computed: the mean and variance of the total are the sums of the countries' means and variances. Treating this group of countries as if it were a single Country A, the trading ratio between B and the group is computed using the above formula. In the second stage, the single Country A buys emissions from the group of countries using the appropriate trading ratio. Unless A and B have very large emissions compared with other countries, this calculation should give almost the same answer as if the group of countries included A and B. Thus in stage one, B is selling emissions to the clearinghouse (consisting of every country), and in stage two, A is buying from the clearinghouse.

This approach neatly converts a trade between two countries into trades between countries and clearinghouses. However, such a system could be controversial if it is perceived as inequitable. In particular, the trading ratio at which a country sells emissions to the clearinghouse will be different for each country (as well as for each source and gas, if that is the level at which trading ratios are calculated), so that a unit of one country's emissions may be more valuable than a unit sold by another country. Similar issues arise for the different buying ratios.

3.3.3 Implications for Trading Ratios in Practice

The discussion in Section 3 has followed a systematic approach of defining an environmental goal and then identifying the statistical implications of that goal. The discussion suggests that, as in the discussion of the adjustment factor in Section 2, there is no unique method for calculating trading ratios, but rather the appropriate ratio depends on the environmental goal. Moreover, for the weak environmental goal examined here, the conventional wisdom (that uncertain emissions should be valued less in a trade than certain emissions) is not borne out. Rather, the trading ratio depends on both the uncertainty and the magnitude of emissions. Further, because uncertain emissions contribute more to increasing the upper bound on the emissions estimate than do certain emissions, reductions in uncertain emissions tend to be valued more highly than reductions in certain emissions (given the definition examined here). Consequently, we should not *assume* that a trading ratio less than 1 (i.e., that a one-unit reduction in uncertain emissions offsets less than one unit of increased certain emissions) is necessitated by the uncertainty of emission reductions.

The trading ratio developed here assumes that we have valid estimates of statistical uncertainty and that we believe that our measures of statistical uncertainty adequately capture and represent all significant sources of uncertainty. Thus, the requisite characteristics of an uncertainty estimate described in Section 2 – for example, that it be objective and verifiable – are even more crucial to a trading ratio. In addition, because a trading ratio that includes an adjustment for country- or source-specific uncertainty involves calculating the ratio each time a trade is made, the system could be administratively intractable or at least very costly to participate in and administer. One solution might be to develop a clearinghouse so that trades occur only through the central authority, and so bilateral trades are not examined on a case-by-case, individual basis.

4 Uncertainty Analysis as a Tool for Inventory Improvement

In the context of national GHG inventories, the process of producing an uncertainty analysis can be divided into four parts: (1) the rigorous investigation of the likely causes of data uncertainty and quality; (2) the creation of quantitative uncertainty estimates and parameter correlations; (3) the mathematical combination of those estimates when used as inputs to a statistical model (e.g., first-order error propagation or Monte Carlo method); and (4) the selection of inventory improvement actions to take in response to the results of the uncertainty analysis. There has been a tendency in much uncertainty work associated with national GHG inventories to focus on the second and third parts, with less effort expended on the first and fourth.

Although the process of modeling the interactions between the uncertainties in parameter values can be instructive, in isolation it does not provide the type of specific information needed to isolate the causes of data quality problems so that they can be corrected or lessened. We refer to any approach to uncertainty analysis that puts an intense focus on the first and fourth parts of this process as *investigation focused*. An investigation-focused approach to uncertainty analysis can both provide the kind of rigorous information needed to more credibly quantify the uncertainties in parameters for use in modeling and simultaneously lead to a system focused on achieving real data and inventory quality improvements.

An investigation-focused approach to uncertainty analysis requires that inventory developers work closely with data suppliers and researchers to (1) exchange information on the inventory's data quality requirements and actual data collection practices; (2) identify activity data reporting or collection problems; (3) identify situations where there is a lack of empirical data for emission factors or other parameters; (4) identify situations where the variability in an inventory parameter is high; (5) identify situations where there is a lack of scientific consensus of the appropriate estimation method for an inventory parameter or category; and (6) identify specific actions that can be taken to correct or mitigate each problem.

The process of analyzing uncertainties can provide a systematic approach for the thorough investigation of the data underlying an inventory and a basis for a more formal understanding of data quality. By jointly identifying specific causes of uncertainty and approximating the magnitude of their effect on data quality, inventory practitioners and data collection agencies can generate better quantitative uncertainty estimates and hopefully also produce better arguments for investments in data quality improvements (e.g., expanded data collection or more research).

This process of implementing an uncertainty analysis effort that is *investigation focused* has been found to be helpful to the authors in the process of preparing inventories at an individual facility (i.e., project), for a corporation, and at the national level. These benefits of this type of approach can be summarized as follows:

- Promoting a broader learning and quality feedback process within the national inventory process.
- Supporting efforts to qualitatively understand and document the causes of uncertainty and help identify ways of improving inventory quality. For example, collecting the information needed to determine the statistical properties of activity data and emission factors forces researchers to ask hard questions and to carefully and systematically investigate data quality.
- Establishing lines of communication and feedback with national statistical agencies, researchers, and other data suppliers, in order to identify specific opportunities to improve the quality of the data and methods used.
- Providing valuable information to reviewers, stakeholders, and policy makers for setting priorities for investments aimed at improving data sources and methodologies.
- Informing policy makers engaged in negotiating future climate change treaties regarding the possible range of confidence they can have in the monitoring of future targets.

It should be obvious that an investigation-focused approach to uncertainty is one that should be tightly integrated with an inventory agency's quality control and quality assurance (QA/QC) processes. In many ways, an investigation-focused approach to uncertainty is simply a more in-depth approach to quality management in that it is a process to rigorously identify the causes of data quality problems, especially ones that the general quality control processes already in place in a country are unlikely to catch. These problems will often involve issues of incomplete data or other systematic biases in the data, which also happen to be key issues for developing a quantitative uncertainty analysis.

An investigation-focused uncertainty analysis can be performed solely on a qualitative basis and still provide useful information for inventory improvements. However, it can provide more useful information for prioritizing the allocation of scarce resources to inventory improvements if it also produces rough quantitative uncertainty estimates. These rough quantitative uncertainty estimates can then be combined with estimates of how much each data quality improvement investment is expected to lower the uncertainty in a particular parameter.

The required characteristics of quantitative uncertainty estimates are obviously less strict if they are only to be used as input for deciding how to prioritize inventory improvements than if they are to be used for a particular policy purpose. For example, it is less critical that rigorous expert elicitation protocols be utilized to increase the comparability of uncertainty estimates across parameters, source categories, and countries. Moreover, because with an investigation-focused approach quantitative uncertainty estimates are only used internally by an inventory agency for allocating resources, the manipulation (i.e., gaming) of uncertainty estimates for the benefit of a particular party is less of a concern. However, particular experts engaged in inventory work within a country may still have an incentive to exaggerate the magnitude of particular uncertainties or the benefits of particular actions in terms of lowering that uncertainty in order to obtain greater budget allocations.

In summary, the purpose of an investigation-focused approach to uncertainty analysis is to improve inventory quality, not just to assess inventory uncertainty. Inventory agencies do not have to choose between an investigation-focused and Monte Carlo-type uncertainty analysis. The former should be seen as a way of obtaining better results than can be obtained from the latter. However, for an inventory agency with limited resources for uncertainty analysis, the quality of its inventory will likely benefit the most if those resources are shifted to the first and fourth parts of the process. Instead of expending resources on quantification and developing models to combine subjective (i.e., expert-judgment-based) estimates, limited resources can be expended on identifying and correcting real data quality problems.

5 Conclusions

Information on the uncertainties in a national GHG inventory – including quantitative estimates of uncertainty – can have a variety of different applications that in turn can satisfy a variety of different goals. For uncertainty information to have practical applications, however, it needs to have characteristics that match the application. These characteristics are particularly restrictive for applications of quantitative inventory uncertainty estimates for policy purposes, such as adjusting emissions for determining compliance.

Consider, for example, a policy that involves an adjustment to an inventory or an emissions trading ratio that is designed to capture uncertainty. Such an adjustment mechanism can, at a minimum, be evaluated against the same types of criteria that we would require of other environmental policies, such as cost-effectiveness, fairness, and administrative feasibility, among others. In turn, these criteria suggest key characteristics that an uncertainty estimate should have if it is to be the basis for an adjustment mechanism, namely, (1) it should be comparable across countries; (2) it should be relatively objective, or at least subject to review and verification; (3) it should not be subject to gaming by countries acting in their own self-interest; (4) it should be administratively feasible to estimate and use; (5) the quality of the inventory uncertainty estimate should be high enough to warrant the additional compliance costs its use in an adjustment factor may impose on countries; and (6) in order to fully secure environmental benefits, it should attempt to address all types of inventory uncertainty, particularly in the case of trading ratios.

In the context of the current state of national GHG inventories, uncertainty estimates do not have the characteristics outlined above. For example, the information used to develop quantitative uncertainty estimates for national inventories is quite often based on expert judgments, which are, by definition, subjective rather than objective. These expert judgments do not undergo any rigorous type of review or verification and are unlikely to be comparable across countries, source and sink categories, parameters, and time, because of differences across the experts producing the judgments.

Over time, however, the authors hope that uncertainty estimates will come closer to possessing these characteristics. As national inventories improve, so should our ability to (1) objectively estimate uncertainties (i.e., by linking uncertainty estimates to specific measurement techniques); (2) review country-specific uncertainty estimates; and (3) elaborate detailed guidance for conducting uncertainty analyses. Whether country-specific quantitative uncertainty estimates of national GHG inventories will ever be "good enough" to base adjustment policies on is highly debatable and depends not only on having the political will to accomplish these changes, but also on the potential technical limits in uncertainty analysis.

Assuming that we can develop quantitative uncertainty estimates for GHG inventories with the characteristics necessary to apply them to policy applications, policy makers still must design an appropriate adjustment mechanism. In turn, the appropriate design of inventory adjustments or trading ratios depends, at least in part, on what type of adjustment is statistically valid; this in turn depends on how the policy goal is defined. Consequently, the design of adjustment mechanisms can benefit from a systematic approach in which policy makers (1) identify clear environmental goals; (2) define these goals precisely in terms of relationships among important variables (such as emissions estimate, commitment level, or statistical confidence); and (3) develop quantifiable adjustment mechanisms that reflect these environmental goals as they are defined. In some cases, a systematic approach may suggest that the statistically valid approach is not the one that is commonly accepted by the conventional wisdom.

An investigation-focused (i.e., qualitative) uncertainty analysis can (1) provide the type of information necessary to develop more substantive, and potentially useful, quantitative uncertainty estimates-regardless of whether those quantitative estimates are used for policy purposes-and (2) provide information needed to understand the likely causes of uncertainty in inventory data and thereby point to ways to improve inventory quality (i.e., accuracy, transparency, completeness, and consistency). Too often, analysts simply assume that uncertainty estimation will provide quality improvements, rather than structuring a process of investigation, analysis, and feedback that is designed to obtain real quality benefits.

Implementing a process of investigating the uncertainty of the emissions inventory may require resolving potentially competing priorities. A process that is intended to derive quantitative uncertainty estimates should involve a different emphasis than a process that is focused on producing inventory improvements. Similarly, deriving uncertainty estimates for use in a policy context may require a very different emphasis than if the estimates are for use in scientific or modeling applications. This paper has begun to explore these issues by identifying how the expected use or application of the uncertainty estimates influences the characteristics that the uncertainty estimate should have. The focus in the paper is on two particular uses: policy (adjustment schemes for emissions inventories and for trading ratios) and inventory improvement. We find that, indeed, identifying the application or applications of the results of an uncertainty analysis is critical to how it should be designed and implemented.

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